A CONCEPTUAL FRAMEWORK
FOR LIFE-CYCLE IMPACT ASSESSMENT

Published by
Society of Environmental Toxicology and Chemistry
and
SETAC Foundation for Environmental Education, Inc.

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U.S.A.

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PREFACE

This book documents the proceedings of a technical workshop on life-cycle assessment (LCA) organized and conducted by the Society of Environmental Toxicology and Chemistry (SETAC) and the SETAC Foundation for Environmental Education, Inc. The workshop was the twelfth in a series of Pellston workshops (so called because the first was held in Pellston, Michigan) convened to evaluate current and prospective environmental issues. Each workshop has focused on a relevant environmental topic, and the proceedings of each have been published as a peer-reviewed or informal report. These proceedings are widely distributed and are valued by environmental scientists, engineers, regulators, and managers because they are technically based and they reflect comprehensive, state-of-the-science reviews.

The Life-Cycle Impact Assessment Workshop marked the first time that the SETAC offices in Europe and the United States shared responsibility in identifying and bringing together international experts for a Pellston workshop. Evidence of the success of the cooperation is shown in the number of countries represented at the workshop.

Special thanks go to the SETAC/SETAC Foundation office staff for logistical support and to the Steering Committee for extraordinary efforts before, during, and after the workshop. The financial support of individual SETAC members, businesses, and governmental agencies made the workshop possible.

Rodney Parrish
Executive Director
SETAC/SETAC Foundation
Pensacola, Florida, USA
March 1993
ACKNOWLEDGEMENTS

The editors would like to acknowledge the workshop participants who contributed a week of their time to provide valuable insights, ideas, and suggestions to advance the science, practice, and application of Life-Cycle Assessment. Special recognition should be given to Linda Longsworth, Gail Kummers, and Cheri Mertins for providing excellent logistics and support during the workshop. In addition, we would like to acknowledge the editorial and word processing support from Roy F. Weston, Inc.
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Environmental professionals, policymakers, and the general public continue to be intensely interested in having a means to look holistically at the cradle-to-grave life cycle of a product, package, process, or activity. In response to this interest, the Society of Environmental Toxicology and Chemistry (SETAC) established an Advisory Group to advance the science, practice, and application of life-cycle assessments (LCAs) to reduce the resource consumption and environmental burdens associated with products, packaging, processes, and activities along their entire life cycle. The SETAC LCA Advisory Group serves as a focal point to provide a broad-based forum for the identification, resolution, and communication of LCA issues. One of the group's activities is to organize workshops for representatives of government, academia, industry, and public interest organizations to address LCA issues.

During the week of February 1-7, 1992, SETAC held a workshop on the role of the life-cycle impact assessment. This workshop was the second in a series to facilitate the development of the science, practice, and application of LCAs and is a continuation of the successful Pellston Workshop Series. The first such workshop, conducted in August 1990, resulted in the workshop proceedings entitled *A Technical Framework for Life-Cycle Assessment.*

Since 1977, SETAC members have organized and conducted workshops to address a number of relevant environmental topics. The workshops include:


Society of Environmental Toxicology and Chemistry

SETAC is a professional society of 2,500 members founded in 1979 to provide a forum for individuals and institutions engaged in the study of environmental problems; the management and regulation of natural resources; education, research, and development; and manufacturing and distribution. It is the only professional society that specifically brings together environmental scientists and engineers from academia, government, industry, and public interest groups to provide research, education, and training in environmental problem solving. The goals of SETAC are pursued through activities such as:

• An annual scientific meeting consisting of workshops and scientific paper and poster presentations on topics related to environmental toxicology and chemistry.
• A monthly journal, Environmental Toxicology and Chemistry; a bimonthly newsletter, SETAC News; and special publications (e.g., Multispecies Toxicity Testing and Hazard Assessment of Effluents).

• Organizing and sponsoring chapters to provide a forum for the presentation of scientific data and for the interchange and study of information of more local concern.

• Advice and counsel to technical and nontechnical people, groups, and institutions about scientific issues through a number of standing and ad hoc committees.

SETAC Foundation for Environmental Education, Inc.

The purpose of the SETAC Foundation for Environmental Education is to encourage, support, and promote educational, charitable, and research endeavors that enhance a scientific approach to assessing the risks and benefits of chemicals in the environment.

The goals of the Foundation are (1) to enhance public understanding of the risks and benefits associated with the chemicals used in everyday life, and (2) to strengthen the technical basis for environmental risk assessment by supporting the purposes of SETAC.

Being associated with SETAC and its 2,500 members from around the world ensures a balanced view of environmental issues (SETAC members come from academic, industrial, and governmental backgrounds), provides accessibility to multidisciplinary technical expertise unmatched in any other professional society dealing with environmental issues, and establishes immediate linkage with a network of environmental professionals on a global basis.

The idea for the SETAC Foundation for Environmental Education came out of the Strategic Planning Meeting held in Pensacola, Florida, in January 1989. There, past presidents of the Society, the Board of Directors, and the Committee Chairs agreed that a foundation was needed from the perspective of both enhancing environmental education and securing sufficient funding to achieve other Society aims and goals.

The SETAC Foundation for Environmental Education was incorporated in the District of Columbia, May 11, 1990, and was declared a 501(c)(3), tax-exempt organization by the Internal Revenue Service on January 4, 1991.

SETAC LCA Advisory Group

The Sandestin workshop was organized and coordinated through the SETAC LCA Advisory Group. The mission of this group is to advance the science, practice, and
application of LCAs to reduce the resource consumption and environmental burden associated with products, packaging, processes, or activities. To achieve this mission, the SETAC LCA Advisory Group will:

- Serve as a focal point to provide a broad-based forum for the identification, resolution, and communication of issues regarding LCAs.
- Facilitate, coordinate, and provide guidance for the development and implementation of LCAs.

The LCA Advisory Group reports to the Board of Directors of both SETAC and the SETAC Foundation for Environmental Education. Specific activities such as workshops, conferences, and educational materials development are approved by the Boards of Directors. Technical activities are conducted through the SETAC Technical Committee, educational activities through the SETAC Education Committee, and governmental activities through the SETAC Government Affairs Committee. Funding for the various activities is provided by the SETAC Foundation.

The LCA Advisory Group is committed to fulfill the purposes of SETAC, particularly the study of concepts and the implementation of programs that can be used for the development of ecologically acceptable practices and principles. The group represents a balance among government, industry, public interest organizations, and academia; is multidisciplinary in representation; has an international structure; and recognizes current SETAC activities in product LCAs. The members of the LCA Advisory Group are shown in Table 1.

The workshop and this report are part of a multiyear program to advance the science, practice, and application of LCAs.

Sponsorship

This workshop, A Conceptual Framework for Life-Cycle Impact Assessment, was sponsored by SETAC and the SETAC Foundation for Environmental Education to assess a critical need to enhance our understanding and capabilities to conduct impact assessments as part of LCAs and to identify future research needs. SETAC gratefully acknowledges the financial contributions of the following workshop sponsors:

- 3M
- Battelle
- Canadian Standards Association
- Dow Chemical Company
- Monsanto
- Scott Paper Company
- The Procter & Gamble Company
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<td>The Open University</td>
<td>London, United Kingdom</td>
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<td>Mary Ann Curran</td>
<td>U.S. EPA</td>
<td>Cincinnati, Ohio</td>
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<td>Nico T. deOude</td>
<td>The Procter &amp; Gamble Company</td>
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<td>James A. Fava, Chair</td>
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<td>Allan A. Jensen</td>
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<td>Tim Mohin</td>
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<td>Research Triangle Park</td>
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<td>Kenneth Reckhow</td>
<td>Duke University</td>
<td>Durham, North Carolina</td>
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<td>Bruce Vigon</td>
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<td>Richard Denison</td>
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<td>Washington, DC</td>
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<td>Christine Ervin</td>
<td>Oregon Department of Energy</td>
<td>Salem, Oregon</td>
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<tr>
<td>Jeffrey A. Foran</td>
<td>George Washington University</td>
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<td>Richard Kimerle</td>
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<tr>
<td>H.J. Poremski</td>
<td>Federal Environment Agency</td>
<td>Berlin, Germany</td>
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<tr>
<td>John Rodgers</td>
<td>University of Mississippi</td>
<td>Oxford, Mississippi</td>
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An Approach to Consensus on a Conceptual Framework for Life-Cycle Impact Assessments

To develop a consensus on the state of the practice and research needs for conducting life-cycle impact assessments, approximately 50 experts in LCA and environmental impact assessment assembled for a 1-week workshop. The workshop was held February 1-7, 1992, in Sandestin, Florida, USA. The participants represented state and federal agencies, industry, universities, public interest groups, and research laboratories in the United States, Canada, United Kingdom, Belgium, Denmark, France, Germany, and the Netherlands.

The workshop objectives were to define impact assessment in the context of an LCA, to discuss and develop a consensus on how impact assessments could be applied to LCAs, and to assess the overall need for developing feasible impact assessment methods for LCAs. An additional objective was to identify research needs to improve the impact assessment component of LCAs.

The workshop followed a three-phase format. During the initial phase, discussion initiation papers were presented covering three general areas: LCA background, life-cycle impact assessment approaches, and impact assessment methodology. During Phase 2, small work group sessions were used to identify and discuss impact categories and work group boundaries on assignments. Based on the impact categories recommended by the work groups, the individual work groups identified and discussed, during Phase 3, what impact assessment methods existed, their potential applications to LCAs, and research needs. The six work groups were Human Health, Ecological (Chemical Stressor), Ecological (Nonchemical Stressor), Resource Depletion, Valuation, and Integration (Table 2). A complete list of workshops attendees is provided in Appendix A. Prior to the workshop, each participant was asked to prepare a list of issues and thoughts related to improving our understanding and developing the life-cycle impact assessment component. The specific questions included:

1. What are the key or important impacts to be considered in an impact assessment?
2. From your experience base, are there existing impact assessment methods that could be evaluated for potential application to LCA?
3. Could those methods be applied directly?
4. If not, how might the methods be modified?
# Table 2. Workshop Participants by Work Group

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<td><strong>Ecological Health Assessment — Chemical Stressors</strong>&lt;br&gt;Ken Dickson, University of North Texas, Chair</td>
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<td>S. Alkinson &lt;br&gt;S. Nasir Ali &lt;br&gt;Larry Barnthouse &lt;br&gt;Mary Ann Curran &lt;br&gt;Nico deOude &lt;br&gt;Richard Kimerle &lt;br&gt;Ed Price</td>
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<tr>
<td><strong>Ecological Health Assessment — Nonchemical Stressors</strong>&lt;br&gt;Helias Udo de Haes, CML, Chair</td>
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<td>Derek Augood &lt;br&gt;Jim Chiles &lt;br&gt;Gary Davis &lt;br&gt;Charles Pittinger &lt;br&gt;Kathy Saterson &lt;br&gt;Frank Vincent &lt;br&gt;Linda Wynn</td>
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<tr>
<td><strong>Human Health Impact Assessment</strong>&lt;br&gt;Adam Finkel, Resources for the Future, Chair</td>
<td></td>
<td>Norm Dean &lt;br&gt;Allan Jensen &lt;br&gt;Gordon Loewengart &lt;br&gt;Tim Mohin &lt;br&gt;Robert Moolenaar &lt;br&gt;Karen Shapiro &lt;br&gt;Jeanette Wiltse &lt;br&gt;David Richardson</td>
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<p>| Valuation                                    |</p>
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<td>Richard Denison</td>
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<td>Eun Sook Goidel</td>
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<td>H. Joachim Poremski</td>
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<td>Thomas Saaty</td>
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<p>| Integration                                  |</p>
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<td>James Fava</td>
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<td>Päivi Julkunen</td>
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5. How can the impacts from multisite operations, multireleases or perturbations, or different categories of impacts, be integrated — or should they be?

6. Based on the above evaluation, what is the value or implications for impact assessment conducted as part of an LCA?

Each work group was responsible for developing an initial written summary of the findings discussed during the week. The Steering Committee was responsible for synthesizing the findings of the work groups into a unified report. This report presents the outcome of that workshop.

Chapter 1 presents an overview of the technical framework for an LCA, including the elements added during the workshop. Chapter 2 presents the developed conceptual framework for the impact assessment component. Specific discussions on the three major impact categories are presented for ecological health (Chapter 3), human health (Chapter 4), and resource depletion (Chapter 5). Chapter 6 presents various approaches to valuate impacts across categories. Chapter 7 summarizes the result of an open forum on the impact assessment component of LCAs held in Washington, DC, March 17, 1992. The forum not only summarized the results of the Impact Assessment Workshop, but also solicited additional ideas and comments on impact assessment and LCAs. Research directions and technical considerations necessary to advance LCAs and the impact assessment component are discussed in Chapter 8. Appendix A is a list of workshop participants. A general discussion on various approaches to impact assessment is presented in Appendix B. This document was used to provide workshop participants with background prior to the workshop. Supplementary information is provided in Appendices C through E. References cited in this book are listed in Appendix F.

This workshop report presents a conceptual framework for life-cycle impact assessment from which a technical framework and specific methods and procedures can be developed.
A CONCEPTUAL FRAMEWORK FOR THE LIFE-CYCLE IMPACT ASSESSMENT: A SETAC WORKSHOP REPORT

NATURE AND SCOPE

Environmental professionals, policymakers, and the general public continue to be intensely interested in having a means to holistically examine and reduce the cradle-to-grave environmental impacts of products, packages, processes, and activities. In response to this interest, the Society of Environmental Toxicology and Chemistry (SETAC) established an Advisory Group to advance the science, practice, and application of life-cycle assessments to reduce the resource consumption and environmental burdens associated with products, packaging, processes, and activities along their entire life cycle. A life-cycle assessment (LCA) is an analytical tool that can help in understanding these impacts from the acquisition of raw materials to final disposition. LCAs have been used in the United States, Europe, Canada, and a few other countries on behalf of industries, governmental agencies, and public interest organizations. The SETAC LCA Advisory Group serves as a focal point to provide a broad-based forum for the identification, resolution, and communication of issues regarding LCAs. One of the activities of the group is to organize workshops for representatives of government, academia, industry, and public interest organizations to advance the science by addressing LCA issues.

During the week of February 1-7, 1992, SETAC sponsored a workshop on the role of impact assessment in LCAs. This workshop was the second in a series to facilitate the development of the science, practice, and application of LCAs. The first such workshop, conducted in August 1990, resulted in the workshop proceedings entitled *A Technical Framework for [Product] Life-Cycle Assessments* in which a three-component model for LCAs was developed:

1. An **inventory** of materials and energy used and environmental releases arising from all stages in the life of a product or process, from raw material acquisition to ultimate disposal.

2. An **impact assessment** examining potential and actual environmental and human health effects related to the use of resources (energy and materials) and environmental releases.

3. An **improvement assessment** of the changes needed to bring about environmental improvements in the product or process.

The majority of *A Technical Framework for Life-Cycle Assessment* focuses on defining concepts and developing a framework for the inventory component of an LCA; however, it also identifies the need to conduct other workshops to evaluate other LCA components.
The Impact Assessment Workshop held in February 1992 provided a forum to begin discussing the second LCA component.

OBJECTIVES AND CONTEXT

The workshop participants were charged with defining impact assessment in the context of life-cycle assessment. Additionally, they were asked to discuss and to develop a consensus on whether and how existing impact assessment tools could be applied to LCAs. For those areas where consensus could not be reached, the participants were asked to identify research needs to improve the impact assessment component of LCAs.

Although the impact assessment component is still in an early stage of development, a number of existing impact assessment tools were identified that might be applied to LCAs. This workshop report presents a general conceptual framework for impact assessment from which a technical framework and specific methods and procedures can be developed.

MAJOR FINDINGS

Technical Framework Reaffirmed. The workshop participants reaffirmed the value of the three-component model for LCAs. The improvement component was included in the technical framework to ensure that products, packages, processes, and activities are evaluated with the underlying premise that opportunities should be systematically pursued at each stage in the life cycle that could reduce the environmental burden. Although it was recognized that improvement is only one application of an LCA, it was concluded that this underlying premise was critical and as such the improvement component should continue to be an integral part of the LCA technical framework.

Technical Framework Expanded to Include Goal Definition and Scoping. The workshop participants agreed that to help focus and define the scope of an LCA, goal definition and scoping steps should be added to the technical framework (Figure 1).

![Figure 1. Incorporation of Goal Definition and Scoping Into the LCA Technical Framework](image-url)
The goal definition step serves to define the purpose and the expected products of the study. With a clear goal definition, the scoping step will serve to select time and spatial boundaries, define boundary conditions and assumptions, and determine what is to be included and excluded in the study in a manner appropriate for the goal of the study. These steps should be conducted through an interdisciplinary approach in a manner that is transparent to the intended audiences. It was also agreed that because of the potential complexity of impact assessment, scoping can occur at any stage throughout a study (e.g., after data are collected, or after the initial classification of impacts).

**Feedback to Inventory Analysis.** Because of the expanded nature of some of the data needed to conduct impact assessments, it was recommended that modifications be made, as needed, to the types and extent of data collected in the inventory. The data needs should be defined during the goal definition and scoping portions of the study and refined as the study proceeds. It was agreed that conducting an LCA is not a linear process but one that incorporates feedback loops and requires interaction among the LCA components. For example, the improvement component overlaps with other components because it is recognized that opportunities for improvement to the product, package, process, or activity could be identified throughout the study.

**Impact Categories Identified.** One of the first challenges the participants addressed was to define the major impact categories. The primary impact categories were **human health**, **ecological health**, and **resource depletion**. Another impact category the group thought should be considered was human health, ecological health, and resource depletion impacts associated with changes in **social welfare** aspects. Occupational health considerations were included within the human health category.

**Impact Assessment Component was Further Defined in Three Phases.** Based on discussions at the workshop and during the SETAC-Europe workshop held in Leiden, Netherlands, in December 1991, the workshop participants agreed to a three-step conceptual framework for impact assessment as follows:

1. **Classification** — The process of assignment and initial aggregation of data from inventory studies to relatively homogenous stressor categories (e.g., greenhouse gases or ozone depletion compounds) within the larger impact categories (i.e., human and ecological health, and resource depletion).

2. **Characterization** — The analysis and estimation of the magnitude of impacts on the ecological health, human health, or resource depletion for each of the stressor categories, derived through application of specific impact assessment tools.

*This three-step impact assessment model further developed the two-step impact assessment model discussed during the Leiden Workshop. This workshop separated the assigning of stressors to stressor categories (i.e., classification) and the analysis (weighting) of stressors within stressor categories (i.e., characterization). The output of the Leiden Workshop combined assigning and weighting to the classification step, which represents a minor procedural change for added clarity but does not result in conceptual or scientific conflict.*
3. Valuation — The assignment of relative values or weights to different impacts and their integration across impact categories to allow decisionmakers to assimilate and consider the full range of relevant impacts across impact categories. Use of formal valuation methods should make this process explicit and collective, rather than one based on implicit, individual value judgments.

The relationship among these phases and the LCA technical framework is presented in Figure 2.

The Stressor Concept Links the Inventory and Impact Components of an LCA. One of the significant findings was the importance of the stressor concept to bridge the gap between the inventory and impact assessment components. A stressor was defined as a set of conditions that may lead to impacts. For example, typical inventory data will quantify the amount of sulfur dioxide (SO₂) released, which might produce acid rain and in turn affect the acidification in a lake, which in turn might change the species composition in the lake or eventually create a loss of biodiversity in the lake. For this example, the quantity of SO₂ released would be reported in the results of an inventory and would be classified in the acid emission stressor group and thus linked, through other stressor groups, to the impacts mentioned, such as acid rain or loss of biodiversity.
EXECUTIVE SUMMARY

Establishment of Cause-and-Effect Relationships Not Part of an LCA. One of the generic findings of the workshop was that establishing the linkage between the occurrence of an actual impact and a category of stressors is not necessarily a component of an LCA. Using the example of SO₂ emissions leading to acid rain and ultimately other impacts, such as a loss of biodiversity, it is not the purpose of an LCA to definitively prove that the release of SO₂ is responsible for a loss of biodiversity. In other words, the group recognized that cause-and-effect relationships are sometimes difficult, if not impossible, to prove. Moreover, it was recognized that quantification of the specific actual impacts associated with any product or process is extremely difficult in the context of an LCA and might better be addressed by other impact tools, such as risk assessments. As a result, the participants focused their efforts on surrogate methods for assessing impacts (e.g., stressor concept). These methods tend to link inventory data (e.g., emissions of SO₂ in air) with impacts (e.g., a loss of biodiversity) and assess the potential magnitude of the impact by the quantity of release, the potency of the release, the expected environmental concentrations, and the probable exposure regime. It must be emphasized that these methods of analysis do not indicate that actual impacts will be observed in the environment because of the life cycle of the product or process under study, but only that there is a potential linkage between the product or process life cycle and the impacts.

Impact Assessment Tools. Many of the workshop participants entered the workshop skeptical that impact assessment was feasible as part of an LCA. After discussions among individuals experienced with general impact assessment tools [e.g., environmental (human and ecological risk) assessments and environmental impact statements (EA/EIS) under the National Environmental Policy Act (NEPA)], there was a realization that some of these tools may have practical application to LCAs. Research is needed to determine whether or not methods developed for human health and environmental risk assessments, particularly at the generic or programmatic level, could be adapted to the practice of LCA. Similarly, conventional resource analysis methods may be adapted for interpreting resource usage data in an LCA, but require practical demonstrations to assess their feasibility.

In the context of LCA, impact assessment will usually be comparative in nature. This places a different requirement on life-cycle impact assessment from what would be required for an absolute "stand alone" impact assessment of a single product or process. Comparative analysis can more readily use stressors as surrogates for impacts. Thus, for the purposes of life-cycle impact assessment there will likely be less need for detailed assessments, such as site-specific risk assessments.

Valuation Phase is Subjective. The valuation phase, which assigns value or relative weights to the various impact categories, was judged to be inherently subjective and value laden. An individual’s or a group’s view of the relative importance of one impact category compared to others was recognized as fundamentally subjective.

A variety of tools (often referred to as decision theory techniques) that offer the potential to make valuation a rational, explicit process were described. These techniques can use both expert judgment and input from interested or effected parties (publics).
Although it was recognized that not all applications of LCAs would require the use of decision theory techniques, the tools described appear to hold some promise for application to LCAs.

**SUMMARY AND FUTURE WORK**

In summary, the participants left the workshop encouraged that some impact assessment tools or processes are being developed that could be eventually applied to LCAs. However, the participants also recognized that impact assessment is still in an early stage of development and identified a number of research initiatives to enhance the science, practice, and application of LCAs. These initiatives, stated below, relate to the specific actions needed to improve the impact assessment component, as well as actions that will further develop the overall LCA method.

- A multiyear research initiative is needed to ensure the development of effective life-cycle impact assessment tools and LCA methods in general.

- Case studies should be developed demonstrating the usefulness of the impact assessment steps (i.e., classification, characterization, and valuation), either individually or combined, when applied to a wide range of products, packages, processes, and activities.

- Scoping processes used in other applications should be critically evaluated for their application to LCAs.

- Research is needed to evaluate the cause-and-effect relationship between pairs of stressor-impact linkages relevant to ecological and human health impacts.

- Evaluation of methods to quantify the resource depletion impact category is needed.

- Approaches to applying various decision theory methods to LCAs should be examined.

- The role of social activities and their influence on ecological, human health, and resource depletion impacts should be further considered and approaches to incorporating these impacts in LCAs should be evaluated.
CHAPTER 1.0

A FRAMEWORK FOR LIFE-CYCLE ASSESSMENT

1.1 OVERVIEW

A life-cycle assessment (LCA) is a process to evaluate the resource consumption and environmental burdens associated with a product, process, package, or activity. The process encompasses the identification and quantification of energy and material usage, as well as environmental releases across all stages of the life cycle; the assessment of the impact of those energy and material uses and releases on the environment; and the evaluation and implementation of opportunities to effect environmental improvement. To assist in the development of LCAs, the Society of Environmental Toxicology and Chemistry (SETAC) established an LCA Advisory Group to advance the science, practice, and application of LCAs. One of the ways the Advisory Group advances LCA development is by conducting expert workshops for representatives of government, academia, industry, and public interest organizations. The proceedings presented in this book summarize the findings from the second in a series of LCA workshops entitled *A Conceptual Framework for Life-Cycle Impact Assessment*, held in Sandestin, Florida, USA, February 1-7, 1992.

This chapter briefly reviews the findings from the initial LCA workshop held in August 1990 and further reinforces and expands the original concepts developed there. Chapter 2 broadly outlines the conceptual framework for an LCA developed during the Sandestin workshop. Chapters 3 through 6 elaborate on various elements of the conceptual framework for impact assessment. Chapter 7 summarizes the results of an open forum held to discuss the outcome of the Sandestin workshop. Chapter 8 outlines ongoing research needs. Appendix A lists the workshop participants. Supplementary information is provided in Appendices B through E. References cited in this document are provided in Appendix F.

1.2 A TECHNICAL FRAMEWORK FOR LIFE-CYCLE ASSESSMENT: A REVIEW

This subsection presents a brief overview of *A Technical Framework for Life-Cycle Assessment* published by SETAC (1991) to place the results of the life-cycle impact assessment workshop in perspective.

One of the most significant findings presented in the Technical Framework is the recognition that the complexity of environmental issues requires a more sophisticated approach than just quantifying materials and energy use, along with airborne, waterborne, solid waste, and other environmental releases, as has been done in previous LCAs. The Technical Framework identifies a three-component model for LCAs, which is illustrated in Figure 1-1 and described below:
A Framework for Life-Cycle Assessment

1. An inventory of materials and energy used and environmental releases (e.g., air, water, and solid waste) arising from all stages in the life of a product or process, from raw material acquisition to ultimate disposal.

2. An impact assessment of potential and actual environmental and human health effects related to the use of resources (energy and materials) and environmental releases.

3. An improvement assessment of the changes needed to bring about environmental improvements in the product or process under study.

An implication of the Technical Framework is the understanding that an LCA addresses energy and material use and environmental releases and disturbances along the entire life cycle of a product, including raw materials acquisition; manufacturing, processing and formulation, distribution and transportation; use, reuse, and maintenance; recycling; and disposal. The three-component model for LCAs, developed and agreed to during the Vermont workshop, evolved from some earlier work done in the area of environmental assessment and from work done by EPEA (1990).
In general, there is a recognition that the technical framework for LCAs is still in an early stage. The framework outlined in the SETAC (1991) report presents a baseline on which procedures and methods can be further developed. Individual components can be useful in addressing specific questions regarding a process or product. For example, the life-cycle inventory (LCI) is valuable because it can assist in identifying steps to improve environmental quality by:

- Establishing a baseline of information on a system's overall resource requirements, energy consumption, and environmental releases.
- Identifying points within the life cycle as a whole, or within a given phase, where the greatest reduction in resources, energy, and releases might be achieved.
- Comparing the system inputs and outputs associated with alternative products and processes.
- Guiding the development of new products or processes toward a net reduction of resources, energy, and environmental releases.
- Identifying needs and providing information to the life-cycle impact and improvement analysis components.

LCAs completed to date have typically included only the first component, the inventory of environmental releases and energy and material uses.

Three of the major research needs identified at this workshop were: (1) refinement in the LCI component; (2) development of approaches to bridge the gap between the inventory and assessment components of an LCA; and (3) development of methods to perform the life-cycle impact and improvement analyses.

1.3 EXPANDED TECHNICAL FRAMEWORK FOR LIFE-CYCLE ASSESSMENT

This workshop reaffirmed the basic technical framework for an LCA and added some refinements. The workshop discussed the relationships among the three LCA components identified in the Technical Framework for Life-Cycle Assessment. Considerable debate led to the incorporation of the goal definition and scoping process into the LCA, as illustrated in Figure 1-2. The SETAC-Europe workshop held in Leiden, Netherlands, in December 1991 had initially pointed out the importance of including goal definition and scoping.
The goal definition and scoping discussion centered on the importance of defining the interrelationships among the three LCA components: inventory, impact, and improvement. The participants believe that the intended purpose should be clearly articulated and agreed upon right at the start of an LCA study. The agreed-upon purpose would then drive the scope, boundary conditions, data categories, and data needs. Similarly, as initial data are gathered and reviewed during the inventory component, they may suggest additional or reduced data needs relevant to some category of impacts. Further discussion of what is included in the goal definition and scoping process is provided below.

1.3.1 Goal Definition

The goal definition element of an LCA identifies the purpose for the study and its intended application(s). This step will present reasons why the study is being conducted and how the results will be used.

LCAs can achieve different goals for different user groups. Their varied purposes potentially encompass comparisons of products, processes, packages, or activities; site and technology selection; product development or improvement; pollution prevention; use as a
problem-solving or optimization tool; use as a strategic management tool; and labeling. They may range in scope from site specific to global in nature.

Consideration should be given at the beginning of the study to assess whether the results will be used only internally within a company, or whether they will be released to the public or used in a public policy context. The intended use will influence the subsequent scope and content of the LCA study.

1.3.2 Scoping: Purpose and Definition

Scoping defines the boundaries, assumptions, and limitations of a particular LCA. It defines what activities and impacts are included or excluded and why.

An LCA practitioner must decide at the outset the nature of the system to be studied in relation to the study's goal, what data must be gathered, and what impact areas will be assessed. Resource capabilities (time, money, laboratories, technically qualified people) limit the practitioner in developing every data element for every exposure potential for each possible impact. As a result, decisions will have to be made and priorities set as they are in other assessment techniques (e.g., risk assessments). Stated simply, the scope of the LCA must be bounded.

Scoping should be attempted before any LCA is conducted to ensure that:

- The breadth and depth of analysis are compatible with and sufficient to address the goal of the LCA.
- All boundaries, methodologies, data categories, and assumptions are clearly stated, comprehensible, and visible.

As analysis proceeds beyond an inventory into the impact assessment component, factors considered become both more numerous and less quantifiable in absolute terms. The potential extent may be infinite, requiring that boundaries and pathways be realistic, practical, and clearly articulated; otherwise, there is a real danger of confusion and of producing an outcome that is unmanageable and difficult to interpret. At the same time, it is important to recognize that restrictions placed on the scope of the LCA may also limit the range of appropriate applications of the results. For example, a decision to limit analysis to human health impacts and exclude ecological effects will necessarily limit the uses and nature of conclusions that may be legitimately drawn from the study. Similarly, a scoping decision that, in order to reduce a study's costs, directs the analyst not to collect actual release data but to assume environmental releases are within regulatory limits must reflect that assumption in any interpretation or communication of the study's results.

Scoping explains what major decisions have been made throughout the LCA and why. As such, scoping along with goal definition plays a critical and fundamental role in shaping
the depth and extent of an assessment. A discussion of the role of scoping in the field of Environmental Impact Assessment (EIA) is presented in Text Box 1.

Examining the Stages in Scoping Within an LCA Context. The concept of scoping as discussed in Text Box 1 applies directly to LCAs. In a product’s life, the information derived from the inventory for each of the categories of stressors (see Chapter 2) should be initially considered. Some of the stressors may be easily eliminated from analysis, some may quickly emerge as potentially critical, and some may require more study before the level of assessment required can be determined.

Proper scoping also addresses tradeoffs within a life-cycle system and their relevance to an assessment. For example, suppose the purpose of an LCA is to evaluate a process change that will reduce energy usage but increase solid waste production. The LCA will evaluate reductions in emissions of greenhouse gases, acidification precursors, and photochemical oxidants. These benefits will be balanced against the land use requirements. The composition of the waste will be screened for potentially hazardous contaminants using one of the methods described in this workshop proceedings. Other aspects of the product life cycle not affected by the process change being considered (e.g., nutrient loadings or hazardous air emissions, impacts of product disposal) will not be addressed, i.e., everything else remains the same.

1.3.3 Issues Relevant to Goal Definition and Scoping

Goal definition and scoping are generally the first activities undertaken in the LCA process. Both these activities should draw upon a range of disciplines and perspectives rather than a singular or exclusively scientific approach.

The questions of system definitions, exclusions, priorities, and breadth and depth of analysis are likely to continue to be hotly debated topics. However, regardless of where limits are drawn, the reasons for choosing specific goals and boundaries must be transparent. This transparency is essential to withstand scrutiny by clients, the public, and peers. For public studies and perhaps even for internal ones, the use of a peer review process is encouraged. Peer review participants should represent a diverse mix of backgrounds and sectors.

Experience has shown that answering key questions prior to commencing an LCA is often helpful in establishing the initial goal and scope for the study. Some helpful questions (not an exhaustive list) follow:

1. After defining a desired study goal and scope, are key inventory data inaccessible, nonexistent, or of a high degree of uncertainty? If so, can these barriers be overcome or do they seriously impede the utility of the LCA?

2. Does the breadth of the impact assessment generally match the overriding goal of the LCA?
TEXT BOX 1
Scoping within the EIA Context

The term *scoping* is borrowed from a 14-year tradition within the field of Environmental Impact Assessments (EIAs). In 1978, the U.S. Council on Environmental Quality (CEQ) issued federally mandated regulations in response to the National Environmental Policy Act of 1969 (NEPA). Some of the regulations dealt with the proliferation of items, often insignificant, showing up in Environmental Impact Statements (EISs). The utility of EISs for decisionmakers was losing effectiveness because the documents were becoming too laden with what many considered trivial considerations, and the important facts were, in essence, being hidden or overlooked.

In an attempt to reduce paperwork and time delays and improve the use of the EIS as a decisionmaking tool, CEQ regulations required the use of scoping early in the process of EIA. Scoping refers to a process of initially considering a long, perhaps exhaustive, list of potential impacts in relation to a proposed project. For purposes of this discussion, let us refer to the long list of potential impacts as simply the "Long List." By understanding the proposed project in adequate detail, many of the potential impacts on the Long List can be ruled out quickly. Those impacts can then be eliminated from further assessment, saving time and resources for both impact analysts and decisionmakers. For example, a project that neither requires nor produces a product that generates ionizing radiation should not require the consideration of the impact category "human health considerations of radiation releases." Thus, scoping allows the elimination of many potential impacts, but only after having examined the relationship of those impacts to the product, package, process, or activity.

Scoping is also tied to the term *tiering* in the CEQ regulations. This comes into play when impact categories of fairly insignificant consequence on the Long List are potentially impacted by a project. These impacts are tiered to a lower level of importance and are not initially considered in any great detail, if at all. If in the overall scheme of things the impact means little or nothing, the consideration of that impact category should not consume many resources. For example, when building a short road from Point A to Point B, a relatively small amount of water is required. If the only source of water in the area is groundwater, impact to groundwater resources may seem to be an impact category requiring analysis. However, the design engineer says only 20 acre-feet, a relatively small amount of water, will be required to build the road. The groundwater expert in the area says 20 acre-feet of water is less than 1% of the total groundwater flowing through the aquifer on a daily basis. Without any further analysis, the process of scoping indicates that the groundwater depletion impact category can be eliminated from initial analysis.

On the other hand, if the groundwater expert had said that 20 acre-feet of water represents approximately 10% of the total groundwater left in that particular aquifer, a more detailed but still lower tier of analysis may be required. If 20 acre-feet represented 40% of total groundwater left in that particular aquifer, serious or higher tier analyses would be required. If we knew nothing of the groundwater resources in the area, we must consider this impact category further. The point is had we simply relied on the design engineer's bias that 20 acre-feet is a relatively small amount of water, we may have missed this important category of impact for this particular project.

Thus, scoping requires an early analysis of the Long List of potential impacts with reference to a particular project to: (1) eliminate unnecessary impact categories from analysis; (2) tier relatively minor impact categories to a low level of analysis; and (3) identify the critical impacts that must be addressed.
3. Have limits been imposed or have assumed weights been assigned to certain impacts due to
   - time constraints
   - financial limitations
   - the request of the client
   - existing facility constraints
   - technical limitations?

4. Is the aim to optimize a product or develop a new one?

5. How comprehensive is the approach intended to be?
   - Have any major impact categories relating to the natural sciences been omitted?
   - What impacts are included and excluded relating to other sciences (e.g., economic and social)?
   - Has consideration been given to secondary environmental impacts arising from social and economic impacts?

6. What limits have been placed on the extent of analysis of impacts?

7. Has consideration been given to the audiences and their interests, priorities, and level of scientific expertise in identifying and communicating the limitations of the LCA?

One of the areas of discussion relevant to the scoping element of a life-cycle impact assessment was criteria for selection of impacts that are important to carry forward. The selection criteria should be identified and transparent. It was argued that "rigid" selection criteria could be established because certain impacts have been judged unacceptable by the practitioner/user from an ecological point of view. It was counterargued that in some circumstances there may be overriding considerations, such as widespread prevention of human or animal suffering, which might give rise to a use that, under normal circumstances, could be banned. Consensus on this issue could not be reached. It was recommended that research is needed to evaluate the use of these criteria.

1.4 INTERRELATIONSHIPS AND FEEDBACK LOOPS AMONG LCA COMPONENTS

Conducting an LCA is not a linear process, but rather one that entails frequent interactions among the major components and incorporates feedback loops, as the three-component triangle for an LCA (Figure 1-1) is intended to demonstrate. Within the
A Framework for Life-Cycle Assessment

expansion to include scoping and goal definition, there are interrelationships of the scoping step not only at the goal definition step but also at the inventory analysis and impact assessment steps (Figure 1-3).

![Figure 1-3. Interrelationships Among the Major LCA Components](image)

Expansion of the LCA concept to include impact assessment also means expansion in the range and nature of data needed to conduct such an analysis. Workshop participants recognized and recommended that the process of collecting data for inventories be expanded to reflect the needs of impact assessment. The data needs should be defined as thoroughly as possible during goal definition and scoping activities and refined as the study proceeds.

Similarly, the improvement component is linked to the other components because it is recognized that opportunities for improvement may be encountered throughout the study. Clearly, not all aspects of an LCA need to be completed before improvement opportunities can be identified.

An additional point illustrated in Figure 1-3 is the iterative use of scoping at different points during the conduct of LCAs. It is conceivable, and indeed highly likely, that modifications to the study scope can and will occur in the course of conducting an LCA. This arises because as one conducts a study, new information will be uncovered and previous assumptions and boundary conditions may require upgrading. It is important to be prepared to change the scope and perhaps even the goal of the study as new information is uncovered and assessed. This fundamental iterative nature of the LCA components cannot be overemphasized and needs to be consciously revisited throughout the conduct of an LCA.
CHAPTER 2.0

A CONCEPTUAL FRAMEWORK FOR IMPACT ASSESSMENT

2.1 INTRODUCTION

Various forms of impact assessment have been integral to many disciplines in the past. Drawing upon these disciplines, this workshop sought to define the components of impact assessment for the particular application to an LCA. In its simplest form, impact assessment is the evaluation of impacts to any system as a result of some action. The application of impact assessment to a life cycle, with the long list of issues that could arise from a full life-cycle study, is a relatively new area of activity.

This chapter outlines the conceptual framework for a life-cycle impact assessment that emerged from the workshop. The chapter begins with a discussion of the linkage between impact assessment and inventory analysis. This concept is further elaborated in the discussion of classification of inventory data into impact categories. Then the fundamental assumptions underlying a life-cycle impact assessment are defined and outlined. Finally, the three major phases of life-cycle impact assessment agreed to by the participants in the workshop are described: classification (identification and assignment of inventory data into impact categories); characterization (analysis of the relative impact magnitude of each stressor within an impact category); and valuation (integration and weighting across impact categories).

2.1.1 Stressors: Link Life-Cycle Inventory and Impact Assessment

One of the significant developments of the Sandestin workshop was the formulation of the concept of stressors as applied to an LCA. Stressors are conditions that may lead to human health or ecological health impairment or to resource depletion. To link the LCI to the life-cycle impact assessment, the environmental loading and resource consumption data that are developed in the LCI analysis are categorized into stressor groups. Some examples of stressor categories are shown in Table 2-1. A single stressor may be associated with multiple impacts. For example, a typical inventory might quantify the amount of SO₂ released to the air. This SO₂ may contribute to the formation of acid rain, which may lead to the acidification of lakes and potentially to fish kills that ultimately decrease the biodiversity of the lake. This example is illustrated in Figure 2-1. Likewise, a single stressor can be assigned to multiple stressor categories. For example, the sulfur dioxide emissions could be assigned to the acidification precursors and the toxicants stressor groups within the ecological health category. Clearly, there can be complex pathways from the simple accounting of loading and consumption in the LCI to the analysis of impacts. Classification of inventory data into stressor categories that are potentially linked to ecological and human health and welfare impacts is the first step in life-cycle impact assessment.
TABLE 2-1. EXAMPLE STRESSOR CATEGORIES WITHIN MAJOR CATEGORIES

<table>
<thead>
<tr>
<th>Resource Consumption*</th>
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<tbody>
<tr>
<td>• Flow Resources</td>
</tr>
<tr>
<td>• Stock Resources</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Ecological Health - Chemical Stressors</th>
</tr>
</thead>
<tbody>
<tr>
<td>• Climate Change Gases</td>
</tr>
<tr>
<td>• Ozone Depletion Gases</td>
</tr>
<tr>
<td>• Toxicants</td>
</tr>
<tr>
<td>• Acidification Precursors</td>
</tr>
<tr>
<td>• Photochemical Oxidants</td>
</tr>
<tr>
<td>• Nutrients</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Ecological Health - Nonchemical Stressors</th>
</tr>
</thead>
<tbody>
<tr>
<td>• Habitat Alteration</td>
</tr>
<tr>
<td>• Siltation</td>
</tr>
<tr>
<td>• Waterborne Litter</td>
</tr>
<tr>
<td>• Compaction</td>
</tr>
<tr>
<td>• Ionizing Radiation</td>
</tr>
<tr>
<td>• Noise</td>
</tr>
</tbody>
</table>

*See Chapter 5 for clarification of terms.

2.1.2 What is an Impact?

An impact may be broadly defined as any effect to human health and welfare or the environment. In the context of an LCA, an impact may be defined as the reasonable anticipation of an effect. In other words, it is unlikely that the data and methods associated with an LCA are sufficient to unequivocally link the occurrence of a specific impact to the consumption or loadings associated with any specific product or process life cycle. It is far more likely that life-cycle impact assessment will assign the consumption and loading data from the inventory stage to various impact categories (classification) and then, through a series of characterization methods, attempt to quantify the magnitude of the contribution that loading or consumption could have in producing the associated impact. Life-cycle impact assessment does not seek to determine actual impacts, but rather to link the data gathered from the LCI to impact categories and quantify the relative magnitude of contribution to the impact category.
2.1.3 Cause and Effect Not Necessary for Impact Assessment

As mentioned previously, one of the generic findings of the workshop was that the linkage between the occurrence of an actual impact and a category of stressors is not necessarily an outcome of a life-cycle impact assessment. Consider the previous example of SO$_2$ emissions (Subsection 2.1.1). It is unlikely that life-cycle impact assessment will prove that the product or process under study is responsible for a certain amount of loss of biodiversity. Such cause-and-effect relationships are usually difficult, if not impossible, to prove. Moreover, it was recognized that quantification of the actual specific impacts associated with any product or process is extremely difficult in the context of LCA. As a result, the workshop participants focused their efforts on surrogate methods for assessing impacts (e.g., stressor concept). These methods tend to link inventory data (e.g., emissions of SO$_2$ to air) with impacts (e.g., a loss of biodiversity) and to assess the potential magnitude of the impact by the quantity of release, the potency of the release, the expected environmental concentrations, and the probable exposure regime.
2.1.4 Nonthreshold Assumption

As suggested in Subsections 2.1.2 and 2.1.3, the term *impact* in the context of an LCA must not be overinterpreted. The results of LCI studies are often expressed in units of product or process equivalents. This means that life-cycle impact assessment results are also quantified on a product or process equivalent basis. Participants in the workshop quickly realized that it is difficult or improbable to associate the consumption or loading with any single product or process with an impact. For example, life-cycle impact assessment is not able to predict the precise loss of biodiversity associated with the production of a single product. In addition, the participants realized that if the impacts associated with each product or process equivalent had to be quantified, the result would often be indistinguishable from zero.

A nonthreshold assumption, which is imbedded in life-cycle impact assessment, is that although specific impacts often cannot be directly ascribed to a given product or process, the loading and consumption associated with the product or process life cycle contributes to impacts and, therefore, must be considered. For example, although it is unlikely that any given single product or process is responsible for an appreciable rise in average global temperature, many products and processes emit gases that contribute to the ultimate impact of global climate change. Simply because the individual product or process is not solely responsible for the ultimate impact does not mean that its relative contribution to the impact should not be considered. It is also true for many impacts, especially those of a more localized nature, that the initial loading may be within the assimilative capacity or no observable adverse effects levels of the ecosystem. For the majority of life-cycle impact assessments, where site-specific effects are not considered, such thresholds will not be identified and impact potential will be associated with each unit of loading. Using this assumption, life-cycle impact assessment can distinguish the relative contribution of the various loading and consumption data from the LCI to impact categories, rather than the often-used alternative of assuming that all loadings and consumption are of equal importance.

It is imperative that this assumption be made clear in the presentation of results for every life-cycle impact assessment. It would be very easy for a reader to misinterpret life-cycle impact assessment results and decide that a single product item is responsible for a loss of biodiversity when, in fact, the actual magnitude of the effect of the single product item's life cycle to biodiversity is unknown. In some cases, just the mention of the impact category will lead to the impression that the product or process is responsible for the impact. Presentation tools must be developed that will minimize these misconceptions but also allow the life-cycle impact data to be conveyed in a meaningful way.

2.2 IMPACT CATEGORIES

The workshop participants identified four key categories of impacts relevant to an LCA: ecological health, human health, resource depletion, and social welfare. Table 2-2 presents examples of the types of impacts that fall within each of the four categories.
TABLE 2-2. CATEGORIES OF IMPACTS RELEVANT TO LCAs

<table>
<thead>
<tr>
<th>Ecological Health</th>
</tr>
</thead>
<tbody>
<tr>
<td>Structure</td>
</tr>
<tr>
<td>• Population, community, and ecosystem</td>
</tr>
<tr>
<td>• Trophic levels</td>
</tr>
<tr>
<td>• Habitat</td>
</tr>
<tr>
<td>Function</td>
</tr>
<tr>
<td>• Productivity</td>
</tr>
<tr>
<td>• Processes (e.g., carbon, nitrogen, and sulfur cycles)</td>
</tr>
<tr>
<td>Biodiversity</td>
</tr>
<tr>
<td>• Habitat loss</td>
</tr>
<tr>
<td>• Rare and endangered species</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Human Health</th>
</tr>
</thead>
<tbody>
<tr>
<td>Acute effects</td>
</tr>
<tr>
<td>• Safety issues (e.g., accidents, explosions, and fires)</td>
</tr>
<tr>
<td>Chronic effects</td>
</tr>
<tr>
<td>• Disease issues (e.g., cancer)</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Resource Depletion</th>
</tr>
</thead>
<tbody>
<tr>
<td>Stock resources (e.g., energy and materials)</td>
</tr>
<tr>
<td>Flow (Renewable) resources (e.g., energy and materials)</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Ecological, Human Health, or Resource Depletion Associated with Social Welfare Activities</th>
</tr>
</thead>
<tbody>
<tr>
<td>Air, water, and land quality/quantity (e.g., use impairment)</td>
</tr>
<tr>
<td>Agricultural productivity (e.g., food and fiber production)</td>
</tr>
<tr>
<td>Natural resource productivity (e.g., fish and timber)</td>
</tr>
<tr>
<td>Recreation</td>
</tr>
<tr>
<td>Materials damage (e.g., buildings and cultural resources)</td>
</tr>
<tr>
<td>Aesthetics (e.g., visibility, noise, and odor issues)</td>
</tr>
</tbody>
</table>
Subsequent chapters in this report discuss in detail the state-of-the-art practice of impact assessment within each of the first three categories. The nature of the social welfare category and its relationship to the other three categories are briefly discussed in Text Box 2.

### 2.3 STEPS OF IMPACT ASSESSMENT

Based on discussions at the SETAC LCA workshop and at the SETAC-Europe workshop held in Leiden, Netherlands, December 1991, participants formulated a three-step conceptual framework for impact assessment:

- **Classification**: The process of assignment and initial aggregation of data from inventory studies to relatively homogeneous stressor categories (e.g., greenhouse gases or ozone depletion compounds) within the larger impact categories (i.e., human health, ecological health, and resource depletion). This step may also entail some initial aggregation of outputs from the inventory based on similarities in their potential impacts.

- **Characterization**: The analysis and estimation of the magnitudes of potential impacts on the ecological health, human health, or resource depletion for each of the stressor categories, derived through application of specific impact assessment tools.

- **Valuation**: The assignment of relative values or weights to different impacts and their integration across impact categories to allow decision makers to assimilate and consider the full range of relevant impacts across impact categories. Formal valuation methods should be used to make this process explicit and collective, rather than one based on implicit, individual value judgments.

Figure 2-2 presents a conceptual schematic of these three steps and how they interface with the other primary LCA components. This schematic is elaborated in Figure 2-3 and explained in detail in this chapter. Figure 2-3 begins with the four major impact categories identified as relevant for consideration in impact assessment. These categories both derive from and inform the content of the LCI: the data and other information collected through the inventory should be of sufficient breadth to allow for an analysis of impacts in all four categories.¹

The remainder of this chapter discusses each of the three major phases of life-cycle impact assessment with reference to Figure 2-3.

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¹Two additional points should be emphasized here. First, as explained in Text Box 1, the inclusion of the social impact category should not be read to mean that all social impacts are necessarily included in an LCA. Rather, the primary emphasis should be on environmental impacts that arise directly or indirectly from other social impacts; a direct examination of social impacts may be essential to identify the full range of environmental impacts that need to be considered.
TEXT BOX 2

Relationship of Social Welfare Impact Category to Environmental Categories

Four basic categories of impact have been identified by workshop participants: human health, ecological health (chemical and nonchemical stressor mediated), resource depletion, and social welfare. Inclusion of the last of these acknowledges:

1. The important indirect environmental impacts that can arise from direct impacts in the social welfare category; for example, the environmental consequences of an increase in the size of the labor force required for a given process or product.
2. The indirect effects that direct environmental impacts can exert on social welfare.
3. The environmental character of certain social welfare impacts (e.g., effects on aesthetics and noise or odor pollution).
4. The close interplay between environmental and other social values engendered by environmental issues (e.g., property value concerns associated with facility siting).

As a result, drawing a straight line between the traditional set of environmental impacts and those affecting social welfare is at best tenuous. This is certainly the case in the LCA context, where it must be acknowledged that socioeconomic and aesthetic values may frequently be important in assessing the environmental burdens or advantages of products and processes. However, given the primary emphasis on environmental factors in the present context and the difficulty in assessing too large a set of impacts under the single umbrella of LCA, the following conceptual approaches to identifying and considering relevant social welfare impacts are suggested:

1. During the initial goal definition and scoping stage of an LCA, the potential social impacts (Table 2-3) associated with the products or processes being assessed or compared should be explicitly considered.
2. Following the inventory and upon entering the impact assessment component of the LCA, the same full range of categories of social welfare impacts should serve as a checklist to identify categories in which there are significant impacts or differences in impacts to social welfare between the products or processes being assessed or compared; these can then be scrutinized for associated (indirect) environmental impacts. For example, if one is comparing production of detergent surfactants from Malaysian palm oil and U.S. petrochemical sources, one should identify labor requirements as a major differential impact. Environmental consequences of the different labor requirements (e.g., habitat destruction resulting from the construction of laborer housing) should then be explored and added to the impact assessment.
3. Impacts identified as arising in the various traditional environmental categories should be further analyzed to examine their potential or actual indirect social welfare impacts; the latter impacts may, in turn, engender additional environmental impacts relevant to consideration in the LCA. For example, loss or reduction in the productivity of a fishery because of pollution discharges from a manufacturing facility (i.e., socioeconomic impact) may cause a shift of the fishing-dependent community to other areas; this may in turn cause environmental impacts in the new areas subjected to fishing.

These approaches are consistent with the primary focus of LCA on environmental issues, while recognizing that these impacts may both produce and be produced by non-environmental social welfare impacts. This interaction is depicted in Figure 2-3 by the arrow linking the social welfare box with the other three environmental boxes. A direct parallel analysis of social welfare impacts is, of course, possible, which would allow the consideration of a broader array of social and economic impacts that could then be integrated along with the results of life-cycle impact assessment for the ultimate decisionmaking.
2.3.1 Classification

Classification entails the categorization of inventory information based on consideration of the downstream impacts to which individual data release categories are expected to be linked. As discussed earlier in this chapter, the stressor concept is key in linking inventory analysis to impact assessment.

Stressors. When environmental releases or other disturbances occur as a result of activities within the life cycle of a product, a number of steps must occur before that release is associated with an impact to the environment. These steps are mediated by what are defined here as stressors, i.e., sets of conditions that may lead to human health, ecological, and resource depletion impacts.

An output from an inventory may be a stressor on one or more environmental compartments, eliciting a linked chain of impacts (Figure 2-1). The environmental compartments where impacts may occur can be simplistically identified as:

- Humans.
- Abiota (physical environment).
- Biota (biological environment).
- Welfare (socioeconomic environment).
Figure 2-3. Elaboration of the Three Phases of Life-Cycle Impact Assessment
### TABLE 2-3. SOCIAL WELFARE IMPACT CATEGORIES

<table>
<thead>
<tr>
<th>Demographic Impacts</th>
<th>Community Impacts</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fertility and Mortality</td>
<td>Community Conflict</td>
</tr>
<tr>
<td>Morbidity</td>
<td>Community Cohesion</td>
</tr>
<tr>
<td>Migration</td>
<td>Land Use</td>
</tr>
<tr>
<td></td>
<td>Services and Facilities</td>
</tr>
<tr>
<td></td>
<td>Health</td>
</tr>
<tr>
<td></td>
<td>Welfare</td>
</tr>
<tr>
<td></td>
<td>Education</td>
</tr>
<tr>
<td></td>
<td>Public Safety</td>
</tr>
<tr>
<td>Economic Impacts</td>
<td>Community Infrastructure</td>
</tr>
<tr>
<td>Opportunity and Transaction Costs</td>
<td>Housing</td>
</tr>
<tr>
<td>Real Property Value</td>
<td>Transportation</td>
</tr>
<tr>
<td>Inflation</td>
<td>Physical Appearance</td>
</tr>
<tr>
<td>Growth</td>
<td>Community Identification</td>
</tr>
<tr>
<td>Sectoral (Fishing, Recreation,</td>
<td>Community Satisfaction</td>
</tr>
<tr>
<td>Tourism, etc.)</td>
<td>Community Institutions</td>
</tr>
<tr>
<td></td>
<td>Fiscal Impacts</td>
</tr>
<tr>
<td>Public Services and Facilities</td>
<td>Supply/Demand</td>
</tr>
<tr>
<td></td>
<td>Costs/Revenues</td>
</tr>
<tr>
<td>Fiscal Impacts</td>
<td>Sociopolitical Impacts</td>
</tr>
<tr>
<td></td>
<td>Legal</td>
</tr>
<tr>
<td></td>
<td>Governmental</td>
</tr>
<tr>
<td></td>
<td>Credibility</td>
</tr>
<tr>
<td></td>
<td>Centralization</td>
</tr>
<tr>
<td></td>
<td>Intergovernmental Relations</td>
</tr>
<tr>
<td></td>
<td>Regulatory</td>
</tr>
<tr>
<td>Sociopolitical Impacts</td>
<td>Family Impacts</td>
</tr>
<tr>
<td></td>
<td>Labor Force Availability and Participation</td>
</tr>
<tr>
<td></td>
<td>Family Structure</td>
</tr>
<tr>
<td></td>
<td>Employment</td>
</tr>
<tr>
<td></td>
<td>Family Stability</td>
</tr>
<tr>
<td>Social Impacts</td>
<td>Sociocultural Impacts</td>
</tr>
<tr>
<td>Social Integration and</td>
<td>Way of Life</td>
</tr>
<tr>
<td>Cohesion</td>
<td>Cultural Survival</td>
</tr>
<tr>
<td></td>
<td>Cultural Heritage</td>
</tr>
<tr>
<td></td>
<td>World View</td>
</tr>
<tr>
<td></td>
<td>Values</td>
</tr>
<tr>
<td></td>
<td>Higher Principles</td>
</tr>
<tr>
<td></td>
<td>Social Justice</td>
</tr>
<tr>
<td></td>
<td>Aesthetics</td>
</tr>
<tr>
<td></td>
<td>Environmental Values</td>
</tr>
<tr>
<td>Social Networks and Social Support</td>
<td>Psychosocial Impacts</td>
</tr>
<tr>
<td>Social Mobilization and</td>
<td>Self-esteem</td>
</tr>
<tr>
<td>Participation</td>
<td>Autonomy and Dependency</td>
</tr>
<tr>
<td>Social Conflict and Tension</td>
<td>Apathy and Alienation</td>
</tr>
<tr>
<td>Social Disorganization and</td>
<td>Uncertainty</td>
</tr>
<tr>
<td>Deviance</td>
<td>Anxiety</td>
</tr>
<tr>
<td></td>
<td>Stress</td>
</tr>
<tr>
<td></td>
<td>Stigma</td>
</tr>
<tr>
<td></td>
<td>Psychopathology</td>
</tr>
<tr>
<td></td>
<td>Quality of Life</td>
</tr>
</tbody>
</table>
Stressor-Impact Chains. A stressor may be linked to an initial impact, which might in turn become a new stressor leading to a secondary impact in the same or some other environmental compartment, and so on through a sequential and possibly branched process. Any of these impacts may be defined as an endpoint of concern, depending on the scope and purpose of a given LCA study. In order to make a reasoned judgment about the potential impact of a stressor, one must consider the inherent properties or characteristics of the stressor, various possible transformations of the stressor in any of the environmental compartments, and potential routes of exposure. Examples of stressors and some of their associated impacts are provided in Table 2-4.

The process of formulating stressor-impact chains associated with each inventory item parallels the process of the impact assessment literature search. Each inventory item (stressor) and initial impact is explored in the literature to identify further stressors/impacts that fall within the scope of the assessment.

Using the previous example of SO₂ emissions to the air as a stressor, it might first be recognized that SO₂ released to the atmosphere can form an acid aerosol, which leads to a number of impacts, including acid rain and decreased visibility. Higher-order impacts resulting from these impacts (and new stressors) can then be explored. For example, secondary impacts associated with acid rain could include the corrosion of structures, the leaching of metals from soils, the acidification of unbuffered water bodies, and the destruction of high-altitude red spruce. Further consideration of the destruction of red spruce will lead to other identifiable impacts, such as loss of biodiversity.

### TABLE 2-4. EXAMPLES OF STRESSORS AND THEIR ASSOCIATED IMPACTS

<table>
<thead>
<tr>
<th>Inventory Item/Stressor</th>
<th>Initial Impact</th>
<th>Some Secondary Impacts</th>
</tr>
</thead>
<tbody>
<tr>
<td>Acid emission</td>
<td>Acid rain</td>
<td>Acidified lakes</td>
</tr>
<tr>
<td>Photochemical oxidants</td>
<td>Smog</td>
<td>Health impairment</td>
</tr>
<tr>
<td>Nutrients</td>
<td>Eutrophication</td>
<td>Bogs</td>
</tr>
<tr>
<td>Greenhouse gases</td>
<td>Global warming</td>
<td>Sea level rise</td>
</tr>
<tr>
<td>Ozone depletors</td>
<td>Ozone depletion</td>
<td>Skin cancer</td>
</tr>
<tr>
<td>Malodorous chemicals</td>
<td>Aesthetics</td>
<td></td>
</tr>
<tr>
<td>Toxic chemicals</td>
<td>Toxic effect</td>
<td>Health impairment</td>
</tr>
<tr>
<td>Solid waste</td>
<td>Land consumption</td>
<td>Habitat destruction</td>
</tr>
<tr>
<td>Chemicals released into groundwater</td>
<td>Groundwater impacts</td>
<td>Health impairment</td>
</tr>
<tr>
<td>Fossil fuel use</td>
<td>Resource depletion</td>
<td></td>
</tr>
<tr>
<td>Noise</td>
<td>Human/biological disturbances</td>
<td></td>
</tr>
<tr>
<td>Construction</td>
<td>Habitat destruction</td>
<td>Loss of biodiversity</td>
</tr>
</tbody>
</table>
As the chain increases in length, a decision is required at some point whether or not to continue. Factors such as the relative importance of impacts and the availability of data or other information to characterize them enter into this decision. For example, data on the effects on biota of metals leached from soils may not be available. Where data prove too limiting, the analyst may have to resort to describing a lower-order impact, such as contribution to acid rain formation, as a surrogate for the higher-order impact.

The preceding discussion illustrates that multiple chains may originate from a single inventory item and lead to different impacts. Similarly, several different chains may lead to the same intermediate or higher-order impact (e.g., methane and carbon dioxide emissions as contributors to global warming and subsequent impacts). Clearly, the identification and selection of specific impacts to be examined within each of the four major categories entails some valuation, which must be made explicit in the presentation of results. Some of the methods described in Chapter 6 may be employed at this stage.

2.3.2 Characterization

Once the intermediate and higher-order impacts within the various categories deemed relevant to the analysis are selected, stressors linked to them are first individually analyzed. Where possible, quantitative measures specific to each type of impact may be assigned; alternatively or in addition, qualitative measures of impact may be appropriate.

A variety of impact assessment tools were identified at the workshop, based on experience in related areas such as human and ecological risk assessments, impact assessments, and environmental assessments/environmental impact statements (EAs/EISs) under the U.S. National Environmental Policy Act (NEPA). These tools are discussed where applicable in the chapters relating to each of the major impact categories.

The remainder of this chapter describes the use of characterization methods to derive impact descriptors from inventory data, and then presents a tiered approach to impact assessment that may allow for the breadth and depth of the analysis to be varied, based on the purpose, scope, and intended use of the LCA study.

Impact Assessment Tiers. Within the context of LCA, impact assessment may serve two purposes:

- It serves to make inventory data more relevant by increasing knowledge about potential environmental impacts.
- It can facilitate the aggregation and interpretation of inventory data into forms that are more manageable and meaningful to the decisionmaker.

Currently there are a number of different possible approaches to impact assessment that differ in their breadth and depth and range from generic to site specific. These can be arranged in a hierarchy as follows:
• **Level 1 — Loading Assessment** in which specific data from the inventory are simply listed and perhaps grouped according to their potential effects; this method may also include summing up the inventory data for stressors that can be assigned to a particular impact category.

• **Level 2 — Equivalency Assessment** in which the inventory data are aggregated on the basis of equivalency factors (e.g., Critical Volume).

• **Level 3 — Toxicity, Persistence, and Bioaccumulation Assessment** in which the inventory data are aggregated based on consideration of inherent chemical properties.

• **Level 4 — Generic Exposure/Effects Assessment** in which impacts are determined based upon generic information about environmental processes rather than site-specific information.

• **Level 5 — Site-Specific Exposure/Effects Assessment** in which impacts are determined based on site-specific information about the relevant impact area.

Levels 2 and 3 provide additional information on inherent physical and chemical properties while Levels 4 and 5 provide more comprehensive impact assessment methods because exposure and effects are considered. The inventory data needed for all five approaches will vary greatly in magnitude and detail. The latter two methods would render the results of the LCA more robust, but may limit feasibility because of substantially increased data demands.

The Loading Assessment does not establish any quantitative linkages between loadings and effects. A pure list of inventory data, even if grouped according to potential effect, is difficult to translate into a set of measures of environmental impacts, although it may be useful as a baseline against which changes in inputs/outputs can be measured.

For illustrative purposes, major issues associated with the use of one of the intermediate methods, Equivalence Assessment, are discussed in more detail in Text Box 3.

**Characterization Methods and Impact Descriptors.** The application of various impact assessment tools to inventory data provides preliminary measures of impact. In some cases, these measures can be aggregated, based on common mechanisms of action or other common features. For example, multiple measures may be combined or converted to common units using normalization factors or other means. For example:

• Mass quantities of air releases of different greenhouse gases may be converted into CO₂ equivalents.
TEXT BOX 3

Issues Associated with the Use of the Equivalence Assessment Method

The major question for equivalence assessment is how to define the different equivalency factors. In one way or the other, the factors need to relate the inventory data to specific endpoints or receptors in the stressor-impact chain. A first task, then, will be to select relevant endpoints or receptors. These will vary with the purposes of the study and need not be the highest-order impacts in a given impact chain. For instance, changes in climate may be chosen as an endpoint, even though climatic effects will only be an intermediate impact engendering further impacts along the chain.

The method essentially consists of multiplying the inventory data by the appropriate equivalency factors, thus expressing the inventory data in equivalency units. In this form, they can then be aggregated within each impact category. The equivalency factors should be based upon impact mechanisms that directly relate the inventory data to the chosen endpoints or receptors.

The equivalence assessment method is perhaps the simplest way to relate inventory data to valued endpoints and at the same time serve as a basis for technically based aggregation of the inventory data. In some cases, however, the delineation of appropriate equivalency factors is a simple procedure. Equivalency factors are being developed for impact categories such as climatic change, ozone depletion, acidification, photochemical ozone creation, nitrification, and biochemical/chemical oxygen demand (BOD/COD) loadings. These should be further examined to ensure their scientific validity. For most resource depletion categories, such factors are not yet available, although the principles for determining the factors are relatively straightforward. For disturbance categories, equivalency factors and the principles that should be used to determine them are much less clear.

The development of toxicity equivalency factors, while straightforward in principle, is complicated by the fact that multiple mechanisms typically will be involved, and there is frequently a lack of data about dose-effect mechanisms. These factors limit our ability to define a scientifically sound basis for data aggregation.

- Mass quantities of air releases of different carcinogenic chemicals may be converted into a single measure of relative cancer risk.

- Mass quantities of air releases of different acid rain precursors may be normalized according to their relative contribution to the regional air shed.

These characterization methods (see Figure 2-3) most often can be feasibly applied only within each major impact category (e.g., human health) rather than across categories, and only to a single medium of release (e.g., air). Characterization methods typically draw upon functional or structural similarities among subgroups of individual environmental stressors or intermediate impacts within the category. These characterization methods must be researched further to ensure scientifically valid applications to LCA.

Some examples of characterization methods that have been used in practice or developed conceptually are provided in Appendix B. Because this area is still emerging and
requires significant development, the discussion should not be construed as a technical endorsement of any individual method or class of methodology; rather, it is intended to be illustrative of approaches that currently exist. Considerable care should be exercised in applying such models to ensure that they are firmly based on empirical or other technically defensible grounds, and that the clustering or aggregation that results retains as much useful information content as possible while presenting it in a form that maximizes comprehension.

The initial outputs of characterization methods can include a wide variety of data points or other information characterizing specific impacts within the various categories. These outputs will also vary with respect to their analytical completeness based on data availability, knowledge of impact mechanisms, boundary definitions, and the extent of characterization of the impacted systems. Thus, the application of methods to further aggregate or cluster the information will be important or even essential.

Clustering or normalization of these impacts into subclasses based on common characteristics or mechanisms of action can be used to simplify the array. For example, human health impacts could be aggregated by endpoint, that is reproductive toxicity, mutagenicity, and respiratory impacts. Toxicity equivalency factors based on potency and exposure can be used in some cases to generate normalized weights so that each subclass can be treated as a mixture.2

Similarly, approaches that compare loadings to estimates of the assimilative capacity of a given environmental medium could group substances into subclasses based on the commonality of their chemical or physiological properties. Subclasses could relate different chemical members to one another by, for example, relative rates of in situ biodegradation by bacteria (for degradable organic chemicals), rates of photodegradation mediated through hydroxyl radical reactions in the troposphere (for volatile organic chemicals), rate and efficiency of uptake by indicator plant species (for nutrients), and so on. Functional or process equivalents can be calculated to normalize across members of a subclass, again allowing the subclasses to be treated as mixtures.

The application of characterization methods achieves some clustering or aggregation of the outputs of the inventory and the initial stressor/impact characterization stages of an

\[ \text{Chemical} \quad \text{Concentration} \quad \text{Relative Toxicity} \quad \text{Toxic Equivalents} \]

<table>
<thead>
<tr>
<th>Chemical</th>
<th>Concentration</th>
<th>Relative Toxicity</th>
<th>Toxic Equivalents</th>
</tr>
</thead>
<tbody>
<tr>
<td>Isomer A</td>
<td>10 parts per million (ppm)</td>
<td>1</td>
<td>10 ppm</td>
</tr>
<tr>
<td>Isomer B</td>
<td>100 ppm</td>
<td>0.3</td>
<td>30 ppm</td>
</tr>
<tr>
<td>Isomer C</td>
<td>50 ppm</td>
<td>5</td>
<td>250 ppm</td>
</tr>
<tr>
<td>Isomer D</td>
<td>1 ppm</td>
<td>60</td>
<td>60 ppm</td>
</tr>
</tbody>
</table>

Total toxic equivalents 350 ppm

---

2 Toxicity equivalency factors allow quantities or concentrations of related but distinct chemicals present in a material or in the environment to be converted to common units, based on their relative toxicities, and then added together to provide a single value in the units of toxic equivalents of the reference chemical. For example, four different chemical isomers of a family of compounds detected in an analysis of a sample at different concentrations could be combined as follows:
LCA, resulting in a simpler set of *impact descriptors* (e.g., extent of increase in lake acidification, magnitude of decline in species diversity, magnitude of increase in specific pollutant emission, etc.) within each category (Figure 2-3).

### 2.3.3 Valuation

Once a set of impact descriptors has been developed that as concisely as technically possible characterizes the important environmental attributes or impacts of the products or processes being assessed, the explicit application of *valuation methods* or *models* is appropriate. It is at this stage that explicit relative values are assigned to the various impacts and impact categories, as a prelude to ranking of alternatives or reaching other decisions.

Traditionally, this valuation step has taken place in a variety of forms outside the realm of technical analysis; indeed, the two have been viewed as entirely distinct, one falling entirely within the bounds of science and the other entirely outside. Moreover, more often than not, the valuation process is conducted implicitly and on an individual rather than a collective basis.

Important objectives in developing a conceptual framework for impact assessment are to define an explicit role for valuation and to identify some methodological approaches to collectivizing the process. The range of interested individuals, groups, and publics whose values are solicited and considered in decisionmaking and the relative weights assigned to each of their values will necessarily vary and will be largely dictated by the application or use of a particular study. Nevertheless, developing explicit, systematic techniques for both elucidating and integrating the multiple sets of values relevant to LCA-based decisionmaking should be viewed as a central component of the LCA concept and methodology.

Chapter 6 discusses several conceptual approaches to valuation, as well as specific methods used in other fields, that may be applied or adapted for use in life-cycle impact assessment. These fall into two major categories: expert judgment approaches and participatory (or social judgment) approaches.
CHAPTER 3.0

ASSESSING THE ECOLOGICAL IMPACTS OF CHEMICAL AND NONCHEMICAL STRESSORS

3.1 INTRODUCTION

This chapter assesses the ecosystem impacts from chemical and nonchemical stressors. The term stressors means the set of conditions that may lead to impacts. The term impact mechanisms is used for the environmental processes set into motion by these stressors that result in impacts on valued endpoints like human health, ecological health, and resources.

The scope of this chapter includes major chemical and nonchemical pollution and disturbances to the environment. It does not include resource depletion per se, but includes any nonchemical impacts associated with extracting resources, such as habitat loss in surface mining. Resource depletion as an impact is discussed in Chapter 5. Major chemical stress categories addressed in this chapter include climate change gases, ozone depletion gases, toxicants, acidification precursors, photochemical oxidants, and nutrients. Under the heading of nonchemical stressors, physical and nonphysical stressors have been included that can be introduced into the environment throughout the product life cycle, such as noise, heat, and radiation. Environmental disturbances include activities that impact the ecosystem without withdrawing a resource or adding a pollutant, including habitat loss, species diversity loss, hydrological changes, soil compaction, and local or regional climate change due to local activities.

Because the ecological impacts of chemical and nonchemical stressors are difficult to link to traditional LCI results, it is important that when an impact assessment is part of the LCA, a scoping activity be conducted to assess the significance of these impacts so that appropriate information can be collected during the inventory. There also may be a scoping step after the inventory step of an LCA but prior to the impact assessment to determine the significant ecological impacts of nonchemical stressors. Chapter 2 addresses the role of scoping in life-cycle impact assessment.

Subsection 3.2, chemical stressors, focuses on methods or approaches that can be used to assess ecological impacts. Five methods, ranging from simple analyses of loadings to more complex approaches that use fate and effects models, are presented.

In Subsection 3.3, we have discussed for nonchemical stressors the different levels of impact assessment relevant for the particular impact category, depending on data availability and/or the assumed importance of the impact category. These levels of impact assessment start with the most generic and proceed to the most site-specific exposure/effects assessment.
A product life cycle may be associated with several different potential impacts. LCAs aim to be a useful selection tool to nonexperts in the environmental sciences. This requires the environmental expert to provide some perspective on the environmental importance of different potential impacts. Chapter 6 makes an initial attempt at providing some guidance on decisionmaking.

### 3.2 METHODS FOR ASSESSING THE ECOLOGICAL IMPACTS OF CHEMICAL STRESSORS IN AN LCA

#### 3.2.1 Chemical Stressors

As defined above, stressors are conditions that lead to human health or ecological health impacts. One of the first steps in an LCA is to identify the chemicals associated with an LCI and to place the chemicals into the appropriate stressor categories. These categories then provide a linkage between the inventory component of an LCA and the impact assessment component. An analysis of the stressors associated with a product's LCA provides the basis for predicting ecological impacts. Examples of chemical stressor categories useful in analyzing ecological impacts include:

- Climate Change Gases
- Ozone Depletion Gases
- Toxicants
- Acidification Precursors
- Photochemical Oxidants
- Nutrients

The environmental loadings of "greenhouse" gases such as carbon dioxide and methane should be carefully noted in the inventory phase and assessed in the impact phase since they have the potential to contribute to the emerging problem of global climate change. Similarly, a product's contribution of ozone-depleting chemicals such as CFCs, carbon monoxide, and methyl furan are essential to document in the inventory phase. Air emissions of particulate and gaseous SO$_x$, HCl, and NO$_x$ contribute to the acidification of rain and subsequent impacts on the terrestrial and aquatic ecosystems. Inorganic chemical loadings and organic chemical releases should be inventoried because they may cause toxicity to exposed organisms. Likewise, emissions that contribute to air pollution and photochemical smog formation, such as volatile organic compounds, should be included in the inventory. Nutrients such as carbon, nitrogen, and phosphorus discharged to aquatic receiving systems may cause enrichment and contribute to cultural eutrophication.

There are other stressors associated with a product's life cycle that can have potential ecological impacts. Land use changes and habitat destruction associated with mining, materials extraction, and cutting of forest and land consumption associated with landfills all have the potential to cause adverse impacts.
3.2.2 Methods

There are several alternative methods that can be used to assess the potential ecological impacts of chemical releases associated with the life cycle of a product. These methods range from simple approaches that examine loadings (total weight released per unit time, weights/unit product, etc.) to more complex approaches that estimate environmental exposure and link that exposure to effects on populations, communities, and ecosystems. What is obvious is that not all methods will be used in each LCA. The choice of method or methods is dependent on the specific goal of the LCA and on the results of the scoping activity described in Chapter 1.

As outlined in Chapter 2, five methods or approaches that can be used to assess the potential ecological impacts of chemical stressors are Loading Assessment; Impact Equivalency Assessment; Toxicity, Persistence, and Bioaccumulation Profile; Generic Exposure/Effect Assessment; and Site-Specific Exposure/Effects Assessment. In the sections which follow, these methods are described and some of their strengths and weaknesses are identified. They each have different input data needs that must be included in the inventory before they can be applied. For example, some of the methods require ecotoxicity data for chemicals listed in the inventory in addition to loading data; therefore, it is essential that persons responsible for conducting the Ecological Impact Assessment in the context of an LCA communicate information needs to those responsible for developing the inventory component of an LCA. Ideally, inventory development and impact assessment should be conducted in an integrated manner, not sequentially. This will avoid missing items in either one or the other. Inventory release data often are not provided at a chemical-specific level. However, for an ecological impact assessment such information is essential.

Level 1 — Loading Assessment. Historically, the inventory component of an LCA has included data on the loadings of contaminants to the environment at various stages in the life cycle of a product. Data are included on the weight or volume of contaminant released to the environment either by life-cycle stage (manufacturing, transportation, use/reuse, and/or disposal) or by total for the product throughout its entire life cycle. Frequently, data may be reported as loadings per unit of the product produced. This information can be used to identify stages in the life cycle where loadings can be decreased. Another use of loading data is to compare the overall loading of chemical stressors between products. When applying this method, it is assumed that less loading of contaminants to the environment will result in some gain in environmental quality, although the ecological consequences of the decreased loadings cannot be directly assessed. The advantages of using environmental loading data directly are: (1) convenience and ease, (2) areas for reduction in environmental loading can be identified, and (3) chemical loadings for different products can be compared. However, there are also several distinct disadvantages in not performing an impact assessment, such as: (1) there may be little justifiable nor scientifically defensible ecological improvement, which is the intent of the effort; (2) needed ecological improvements may not be attained because real ecological impacts were not assessed; (3) resources would have been wasted on improvement actions that were not part
of the real environmental issue; and (4) opportunities for improvements may have been missed.

The strength of this method is its simplicity. However, care should be taken in ascribing any direct environmental improvements to the decreased loadings since the approach does not establish any linkages between loadings and effects on ecological resources. This lack of linkage and absence of any quantification of the consequences of the loadings is a major drawback.

**Level 2 — Impact Equivalency Assessment.** Another approach that can be used to estimate the potential ecological impacts of chemicals released to the environment during the life cycle of a product is to assign chemicals found in the inventory to a chemical stressor group. For some of these groups (global climate change gases, ozone depletion gases, acidification potential chemicals, and photochemical oxidant precursors), it is feasible to develop impact equivalency units that can be summed up and used to assess the collective contribution of emissions to environmental problems (Guinee, 1991). Potential ozone depletion factors and global warming factors have been developed by WHO/UNEP (Anonymous, 1989b) and Rotmans et al., 1990. Another example of emission-dose factors are H⁺ equivalents used to aggregate acidifying emissions (Mekel et al., 1990; Lindeijer et al., 1991; and Heijungs et al., 1991). Other factors that relate to various environmental problems are still in the process of development. For example, methods to estimate the photochemical ozone creation potential of gases have been suggested by Anonymous, 1990. Other categories of emissions, such as nutrients and oxygen consuming chemicals (COD and BOD), do not yet lend themselves to such an approach. However, research is being conducted to develop methods to address these categories.

Another modification to this category exists. It is possible to take the total environmental loading data for an individual chemical from the inventory phase and express that quantity in a volume of air, water, or soil that would be needed to dilute it to the concentration equivalent to some generally known toxicity endpoint for that chemical. For instance, if the acute rainbow trout lethal dose (LC50\(^{1}\)) for chemical Z was 10 mg/L, then the water volume equivalency for a total inventory loading of 10,000 mg would be 1,000 L. Or, in the case of air, the emission load could be diluted to the known threshold limit value (TLV). The primary disadvantages of this method are the need to create a new understanding of the equivalency unit approach, the lack of exposure information, and the absence of direct relevancy to ecological improvement. The advantage is it can make different inventory phase output data comparable within the framework of an exposure/toxicity concept for which there is at least some familiarity. In addition, it allows for a method for normalizing a variety of chemicals by expressing their loadings as volumes. These volumes can then be summed if the same endpoints were used, which allows chemical loading data to be aggregated.

\(^{1}\)Concentration causing mortality to 50% of test population.
Level 3 — Toxicity, Persistence, and Bioaccumulation Profile Approach. The Toxicity, Persistence, and Bioaccumulation Profile approach (TPBP) considers physical and chemical properties of substances listed in the inventory to assess their fate and potential environmental effects. Data generated are based on accepted testing methodologies. These methodologies include acute toxicity testing (LC50, EC50), chronic toxicity testing [No Observed Effect Level (NOEL)], biodegradation (half life and CO2 evolution), bioaccumulation (solubility, octanol/water coefficient, and bioaccumulation factor). For many compounds, these data exist in the literature or can be predicted using computer models. Literature sources include Environmental Fate of Organic Chemicals, Fate Books (Verschueren, 1983; Howard, 1990; Meylan and Howard, 1990). Available computer models include AQUIRE (water) and APO (air).

These properties may be used in decision processes and for developing priority lists of inventory data for further analysis. When actual data are not available, it is possible to use predictive structure activity relationships from computerized databases. The disadvantages of this method are that it applies to only a few stressors and, most importantly, does not consider environmental exposure.

Level 4 — Generic Exposure - Effect Assessment. The approach discussed in this section is the simplest method available for quantifying the fate and ecological effects of chemical stressors. The approach involves using simplified models and readily available data to link environmental loadings to individual, population, and ecosystem consequences.

One especially simple and widely used approach to quantifying the environmental fate of chemicals is the unit world approach (Mackay, 1979). The unit world is a hypothetical 1 km³ box containing air, soil, water, sediment, and aquatic biota. The unit world is represented mathematically as a set of thermodynamic equations that describe the partitioning and transformation of a chemical introduced into the box. A relatively small number of chemical-specific parameters are sufficient to predict the partitioning of the chemical between the compartments of the unit world. If rates of transformation due to photolysis, oxidation, biodegradation, or other processes can be estimated, then the unit-world approach can be used to predict steady-state concentrations, residence times, and removal rates. Concentrations in air and water can, in turn, be compared to concentrations found in laboratory tests to be toxic to aquatic and terrestrial biota. Data on chemical toxicities to aquatic biota are available in readily accessible on-line databases (e.g., the Environmental Protection Agency (EPA) AQUIRE database); federal or state standards for the protection of aquatic life have been issued for many chemicals. Data on toxicities to terrestrial plants and animals are more scarce, and we are aware of no applications of the unit-world approach to the assessment of risks of chemicals to terrestrial biota.

Somewhat more complex and realistic models of chemical fate and effects in multiple environmental media have been developed using reference sites or "canonical environments." This approach is similar in concept to the unit world, but instead of a uniform 1 km³ box, the simulated environment is a generalized stream, lake, pond, or other ecosystem type. The reference environments usually do not represent any specific real ecosystem, but are
Assessing the Ecological Impacts of Chemical and Nonchemical Stressors

generally representative of a class of ecosystems within a region. Unlike the unit world, a variety of environmental parameters (e.g., streamflow, percentage of organic matter in sediment) are required. These are often supplied by the developers of the models and need not be generated by the user. Models of canonical environments are routinely used by EPA in pesticide risk assessments and have also been used in assessments of ecological risks of energy technologies (Barnthouse et al., 1985a and 1985b; Suter et al., 1984 and 1985).

The models described above are principally models of aquatic ecosystems. Analogous approaches for quantifying the fate of contaminants in terrestrial systems have been developed (Barnthouse et al., 1985a, 1986; Suter et al., 1984 and 1985). These models simulate (1) atmospheric dispersion and deposition of pollutants on soil and (2) subsequent uptake by plants and animals. A few studies have used this approach to assess risks of atmospheric pollutants to terrestrial biota (Barnthouse et al., 1985a). The principal use of this modeling approach has been the assessment of radionuclide movement in terrestrial food chains.

A variety of approaches have been used to link environmental exposures predicted by the models described above to effects on biota. The principal tools available are toxicity databases and mathematical models. Standardized toxicity test results can provide initial comparisons between exposure concentrations and concentrations known to affect biota. This approach is widely used in water quality assessment, toxic chemicals regulation, and pesticide registration. On-line toxicity databases (e.g., the EPA AQUIRE database) containing toxicity information on thousands of chemicals are available. If fish are the organisms of interest, empirical regression models (Suter et al., 1983, 1987) can be used to extrapolate test results from any tested species (e.g., fathead minnow or sheepshead minnow) to species of interest for an assessment (e.g., salmon or largemouth bass). Population-level effects (changes in abundance or yield to fisheries; risk of extinction) can be predicted using models developed and used for decades by fisheries managers (Ginzburg et al., 1982; Barnthouse et al., 1990). Models that extrapolate from environmental fate and toxicity data to ecosystem-level effects (changes in trophic structure or nutrient cycling) also exist, but the range of ecosystem types represented is limited to lakes and ponds.

Impacts of major air pollutants (sulfur and nitrogen oxides; ozone) on crops can be estimated from dose-response data developed through the EPA National Acid Precipitation Assessment Program (NCLAN) program. Models of forest succession (Shugart, 1984) that could, in principle, be used to quantify air pollution impacts on forests exist, but information on phytotoxicity to forest trees is currently insufficient for impact quantification (NAPAP, 1991).

The advantage of the approach described above, as compared to the simpler approaches described in earlier sections, is that impacts on organism, population, and ecosystem characteristics can be more readily valued than can simple environmental loadings. Uncertainties in loadings, environmental scenarios, or parameters can be quantified using Monte Carlo techniques (Gardner et al., 1981; Bartell et al., 1983 and 1988). Site-specific environmental characteristics can be used if they are known, but the
generic values provided by model developers are probably adequate for many or most LCAs. The utility of this approach is limited, however, to local environmental contamination and to particular classes of contaminants. Only a few attempts to model effects of stressors on watersheds or landscapes have been performed (Hunsaker et al., 1990; NAPAP, 1991), and these have involved specific regions that cannot be generalized. The aquatic fate models described above have been experimentally tested and provide credible results for radionuclides, hydrophobic organic chemicals, and nutrients.

It is probably not feasible to model regional or global effects of chemical stressor releases as part of an LCA. However, many studies of this kind have already been performed in both North America and Europe and can be incorporated in LCAs. Examples include (1) regional sulfur transport and deposition modeling studies performed for the U.S. National Acid Deposition Assessment Program, (2) nitrogen and sulfur "critical loadings" studies performed in Europe (Nillson, 1986; Hordjik, 1991), and (3) general circulation modeling studies performed to assess impacts of greenhouse gases on the global climate. Results of this work can be used to put a perspective on loadings estimates from an LCA.

**Level 5 — Site-Specific Exposure/Effect Assessment.** Actual site-specific field studies designed to directly investigate ecological health and well-being provide the most ecologically relevant understanding on the existence of or lack of chemical impacts on ecosystems. Studies should include chemical exposure measurements in all the environmental compartments of air, water, soil, sediments, and biota. Terrestrial and aquatic populations, communities, and ecosystems should be assessed for structure, function, and diversity. Environmental transport models developed for predicting contaminant transport and fate at nuclear facilities and hazardous waste sites can be calibrated to any specific pollutant source and local environment. Atmospheric transport models that account for local meteorological conditions and terrain also exist. Measurements of the abundance and distribution of local biota can be performed, and site-specific population and ecosystem models can be developed. The value of such studies for LCAs is unclear, given the expense and time required. All published studies of these kinds relate to specific sites with widespread environmental contamination from past disposal practices or the potential for future accidents. If such studies were already being performed in connection with safety analyses or Superfund/Resource Conservation and Recovery Act (RCRA) actions, the information could be used for an LCA.

### 3.3 NONCHEMICAL STRESSORS

#### 3.3.1 Ecological Effects of Ionizing Radiation

Ionizing radiation may be released during product life cycles due to energy requirements based on nuclear power, the incorporation of radioisotopes in the product itself (e.g., smoke detectors), and mining activities where radioactive elements are associated with the mineral or ore being extracted (e.g., phosphate mining). Ionizing radiation impacts plants and animals on a cellular level, causing cell death and genetic mutations. Thus, there
are acute and chronic impacts for individual plants and animals and potential long-term impacts on genetic material that affect whole populations and ecosystems.

Although it is easy to measure the amount of ionizing radiation and specific radionuclides released into the environment from an activity, it is extremely difficult to associate a particular level of release with ecological impacts without information on the particular types of radiation and radionuclides being released and without site-specific information on the flora and fauna and their exposure routes and sensitivities. This is further complicated by the fact that background levels of ionizing radiation exist everywhere to which plants and animals have adapted through the millennia.

On the most generic level, which can be described as a loadings assessment, all of the releases of ionizing radiation can be summed from an LCI inventory and presented in an impact assessment as having the potential for acute and chronic damage to plants and animals and long-term impacts on the genetic integrity of local populations. If this summing were done without some type of scaling factor, it would assume that different types of radiation and radionuclides have the same intensity of effects and similar bioavailability once released into the environment and would also assume that the sensitivities of different plant and animal species were the same. As a first approximation of potential impacts, this at least allows more interpretation of inventory data.

At the next level of specificity of impact assessment, the impact equivalency unit assessment, a scaling would be done for the different types of radiation and radionuclides as a factor to normalize the release quantities when they were aggregated. This scaling would depend on the degree of the effects of the different types of radiation and radionuclides and some measure of their bioavailability once released. The determination of such a scaling factor is an important research need if the potential impacts of radiation releases are to be aggregated in a more scientifically valid manner.

At the next level of specificity, a generic exposure/effect assessment, it becomes much more difficult to assess impacts without detailed information about the particular radionuclides released and the plants and animals exposed. The generic exposure/effect assessment can proceed by assuming the introduction of the sum of the particular forms of radiation and radionuclides from the inventory into a generic environment (e.g., Mackay's unit world). This may allow for a comparison with another product, but it would not permit a risk assessment in any absolute sense. A generic risk assessment could also proceed by assuming a reasonable exposure level and applying that exposure level to an average plant and animal environment. The impacts on the plant and animal environment would then be summed for the total releases of radiation and radionuclides. Of course, the assumption of exposure levels would be critical and would drive the whole analysis.

Finally, given site-specific information about radiation and radionuclide releases and the organisms exposed throughout the product life cycle, an actual risk assessment could be performed. This would proceed by exposure and effects modeling and could provide a quantification of impact on any particular species exposed. The data needs, uncertainties,
and research needs for the site-specific risk assessment approach are large. Although ecological risk assessment techniques exist, the determination of ecological risk for even one site in the product life cycle would be a monumental task. There may or may not be recognized measures of the impacts of radiation on plants and animals that can be calculated through such a risk assessment approach, and the dose-response relationships may not have been determined.

3.3.2 Heat

Along with noise, heat can seriously affect humans, animals, birds, fish, and other living species. Heated water and/or gases are generated by virtually all industrial processes and the resulting impacts are mainly site specific. Also, depending upon the LCA methodology employed, the amount and intensity of the heat emitted at specific locations may not be apparent from the data. Heat can be measured in megajoules as the sum of all production output stages, but this figure has little meaning unless the analyst gives some indication of the change in ambient temperature in the area of release.

Setting aside questions concerning effects upon humans (covered elsewhere), the LCA practitioner, in order to determine if heat pollution is a problem, will have to understand the processes involved in all of the LCA stages and the heat sensitivity of all corresponding locations. This is a tall, and perhaps unnecessary, order. A scoping trigger suggesting that a more detailed examination is required would be high process energy consumption in metallurgical systems, power generation plants, paper mills, and the like sited close to rivers, forests, or seashores.

While the amount of heat emitted from industrial processes is small relative to that from global sources (sun, ocean, and atmospheric sinks), emission sources may produce adverse effects. For example, a warm gas plume descending at some distance from a power plant could affect animal and/or bird habitats. Beneficial effects sometimes have been claimed — larger and greater numbers of certain fish in sea water warmed by cooling water from nuclear power plants, for example. It may be noted that this benefit may have resulted from change in the relative abundance of the different fish originally present. Changed metabolic rates are complicated by several factors, such as changed oxygen concentration.

3.3.3 Noise

Noise is subject to many of the general comments made about heat. Except for transportation noise (where noise is spread over the transportation path), noise pollution is site specific and at high decibel levels can adversely affect humans and other species. Likewise, noise data are not usually included in LCA exercises. In the past, noise problems, if they exist, have usually been managed after the fact by company safety practices and local community processes. It is difficult to conceive of a method providing a meaningful aggregated noise value representing all processes included in an LCA study.
A few processes — blasting for minerals, for example — require attention, and certain products — for example, gasoline-powered lawn mowers, leaf blowers, edging tools, airplanes, and other transportation systems — inherently tend to generate high noise levels. These impacts and ways to mitigate them should be included in an LCA if feasible.

### 3.3.4 Environmental Disturbance

Each of the following impacts and effects should be considered in LCAs where appropriate. The current level of detail in inventory information lends itself more to generic than to site-specific environmental impacts. In some situations, the generic assessment will be sufficient for the purposes of the LCA. In other situations, site-specific data will be necessary. Physical measures for site-specific impacts, covered in Subsections 3.3.2, 3.3.3, and 3.3.4, are well developed.

In each of the following categories, we will note what extra information would be needed to determine site-specific impacts. To be meaningful, the analyst would need to measure the baseline of impacts not caused by the product/process in question.

#### 3.3.4.1 Habitat Alteration

Habitat alteration from nonchemical stressors is defined as a physical disturbance that may result in a structural or functional change in aquatic or terrestrial communities. Such a disturbance can result in a gain or loss in habitat area and in changes to biogeochemical cycling, ecosystem integrity, economic value, and biodiversity, including loss of endangered species. Such habitat alteration is often a key area of concern in an LCA (e.g., Alaska oil pipeline).

Habitat alterations in an LCA are primarily triggered by the following considerations: (1) resource extraction of raw materials, (2) fuel sourcing and production, and (3) transportation activities. These processes are more closely related to product system inputs (e.g., mining, agriculture, water use) than system outputs. However, some operational outputs, initial plant siting, and accidents clearly result in habitat alteration.

Key questions related to habitat alteration are: Does the product or process have the potential to require or result in the use, development, destruction, or alteration of natural undisturbed areas? If so, what are the characteristics of the habitat and the extent of potential impacts? For example, threatened tropical forest or marine ecosystems such as seagrass beds, mangroves, and coral reefs are particularly fragile systems.

There are no "cookbook" methods for assessing all site-specific impacts and LCAs rarely provide this level of detail. However, life-cycle inventories can provide generic information for identifying the potential for habitat alteration. Quantitative measures include resource requirements per product unit, resources extracted per unit area habitat, and total habitat area altered. Qualitative measures can include habitat type, geographic location, and method of extraction or habitat use. For example, an LCA on a paper product could estimate numbers of acres and forest type required to produce raw material inputs.
This information, along with data from existing habitat classification systems, is useful in identifying species and habitat vulnerabilities.

Site-specific information will be required in order to assess more detailed impacts of habitat alteration. If significant habitat alteration is anticipated in initial scoping, then it is recommended that site-specific habitat information be included in the life-cycle inventory. Key quantitative measures for site-specific habitat evaluation include specific location of habitat, biodiversity, and numbers of rare and endangered species. Qualitative information includes degree of habitat disturbance before and after extraction, importance of habitat for migratory species, uniqueness of habitat type, habitat fragility and/or resiliency, and importance for ecosystem function.

Certain types of information can be determined directly with a wide range of ecological and biological measurements, while other data are available from published sources. Biodiversity can be measured with a variety of techniques (e.g., Shannon Weaver index). The International Union for Conservation of Nature (IUCN) produces lists of endangered species and habitats, as do the U.S. federal and state governments. Many federal and state agencies classify and determine management priorities for habitat types and river systems in the United States (e.g., wetlands and undisturbed wilderness areas).

Most determinations of habitat value and priority will rely on expert judgment. The U.S. Fish and Wildlife Service has developed habitat evaluation procedures that support recommendations for selecting project alternatives or for designing mitigation and compensation measures. Habitat management and conservation priorities have been prescribed through a variety of state and federal programs and regulations; these can be used as a guide for assessing the ecological value of the habitat in question.

3.3.4.2 Physical Change to Water

Siltation and Flow. Typical activities that can result in changes to siltation and flow include mineral extraction, timber harvesting, construction, agriculture, and raw material processing. For a generic assessment, any of these should serve as a signal to pursue a further level of detail, depending on the expected magnitude of impact. Although generic measures are not commonly available, it may be possible to extrapolate these effects from, for example, total overburden removed in a mining operation, minus percentage of silt commonly removed in settling ponds. If generic measures can be developed for all relevant levels of production, it should be possible to add those effects to obtain a total siltation load.

Site-specific assessment of siltation would require data on solids loadings and stream flows caused by sources other than the product or process under consideration. Flow will depend on season and vary from year to year. One measurable effect of siltation would be decreases in submerged aquatic vegetation, benthic species, and habitat. For a more complete discussion of habitat effects, see Subsection 3.3.4.1.
♦ Typical environmental disturbances to address during scoping include mineral extraction, hydro construction, raw material processing, timber harvesting, and agriculture.

Total Suspended Solids (TSS), Including Turbidity. TSS is a commonly measured effect for solids discharged to receiving waters and, often, may be measured by turbidity. TSS and turbidity have no real meaning apart from a particular site, so we do not recommend a generic approach for these measures. Impact would be increased stress on certain aquatic species. TSS would have synergistic effect with other factors.

♦ Typical environmental disturbances to address during scoping include mining, raw material processing, processing, timber harvesting, paper manufacture, and agriculture.

Interrupted Surface or Subsurface Drainage. This will always be a result of certain extraction activities, including underground mining, so these activities could serve as an indicator of potential drainage impacts. The specific impacts will be unique to the hydrogeology of the site; assessing these would require a full characterization of the hydrogeology before and after the extraction activity. Effects include decrease in water output from nearby wells, which can be measured. Some effects are not reversible within human time spans.

♦ Typical environmental disturbances to address during scoping include mineral extraction.

Waterborne Litter. Sources include consumer littering and manufacturing losses during raw material/scrap transfer. These activities are connected mostly to plastic and plastic-containing items, such as nets, plastic films, and some packaging. The impact is ingestion by or entanglement of marine life. This is probably more suited to a generic measure than site-specific measure because it is very difficult to link specific releases with an actual wildlife incident. Possible measurement index: pellets measured upstream and downstream from a facility during stormwater discharges.

♦ Typical environmental disturbances to address during scoping include plastic forming processes especially near watercourses, and consumer use and discard of some plastic products.

Drop in Aquifer Level. Aside from depletion of the resource, a drop in level can result in increased salinity of groundwater, such as saltwater intrusion following pumpout of aquifer adjacent to the ocean. This is probably not suited for a generic measure because aquifer recharge and other demands are distinctly site specific. Other effects: ground subsidence, stress or death of vegetation with erosion following.

♦ Typical environmental disturbances to address during scoping include agriculture and manufacturing relying on underground water supplies.
3.3.4.3 Physical Change to Soil

Compaction, Including Paving and Other Impermeable Soil Coverage. Measurable on a site-specific basis as significantly increased storm runoff, potentially resulting in flooding. Also measurable as reduced soil moisture, reduced aquifer recharge and siltation. Measurable effects are net vegetation loss or reduced productivity, through interference with water infiltration and root health. As long as baseline data are available, and the total paved area is known, this mechanism is easily measurable.

♦ Typical environmental disturbances to address during scoping include buildings, highways, and heavy construction. Compaction due to heavy equipment activity can be remedied by freeze-thaw cycle and soil biota where healthy; paving will have a much longer-term effect.

Erosion and Soil Loss. On a generic level, erosion can be expected as an effect of certain activities, such as those listed below under scoping triggers. Erosion is measurable on a site-specific basis as an increase in water siltation, loss of topsoil, and low particulate count. If available, topsoil loss/acre for the studied area before and after the current LCA activity process should be studied. Measurable effects of erosion are reduced carrying capacity and reduced species diversity. Some effects are self-limiting when the disturbing activity ends; others such as ravines may persist for long periods.

♦ Typical environmental disturbances to be addressed during scoping include soil displacement, mineral extraction, timber harvesting on slopes, and agriculture.

Desiccation. Measurable on a site-specific basis as loss of soil moisture or significant water table drop. An extreme case is loss of wetland habitat.

♦ Typical environmental disturbances to be addressed during scoping include construction and/or agriculture in water-short areas.

Subsidence. This site-specific effect (e.g., change to surface topography, pond formation, building damage) is chiefly economic damage rather than environmental. Subsidence is measurable by expert judgment, reduced property values, and aerial observation.

♦ Typical environmental disturbances to be addressed during scoping include underground extraction of water or minerals.

3.3.5 Regional Climate Change

Measurable as local- or region-specific, long- or short-term changes in relative humidity, cloud cover, temperature, insolation, and precipitation. Background measurements on a regional scale (but perhaps not local scale) should be readily available because of the long history of weather observations. Cloud cover, ground temperature, and other factors are measurable by satellite. Other measures of changes: soil moisture, rainfall, or pollen
counts. Key question: Is a measurable change related to a particular product/process, as opposed to reflecting a natural change or some other human activity? Effects are changes to species diversity, vegetation productivity, energy use, and human respiratory effects.

♦ Typical scoping triggers: wetland changes, hydroelectric construction, other large-scale changes in land use.

3.3.6 Species Change

Species changes may include both alterations in composition or in total diversity. Conservation of genetic diversity is important for maintaining adequate gene pools to ensure biological adaptation to global and local perturbations. In addition, biologically diverse areas hold the promise of species with future economic and medical importance. Changes in biodiversity can result from all of the above factors (e.g., loss of habitat, changes in soil, climate, and hydrological cycle).

An important consideration is whether or not the process inputs are of a magnitude that can cause changes in biological diversity. The measures outlined in Chapter 4 are important for assessing whether or not habitat alteration can lead to loss of biodiversity.

Important nonchemical triggers for assessing potential impacts to biodiversity include loss of natural habitat, major changes in soil or water functions, and removal of a single species or natural product that could change natural food chains and affect species balances. Additional triggers are introduction of exotic (nonnative) species, which can also cause major changes in natural biodiversity by changing natural competition mechanisms and food chains and the introduction or release of bioengineered species.

The evaluation of changes in species and biodiversity is difficult. A relative value index for different species based on their scarcity, vulnerability, and replaceability has been proposed.

3.4 DECISION ANALYSIS

There are many techniques available to combine the results of the impact assessment component in order to make decisions about the product. Offered below are two techniques for the Ecological Effects — Chemical Stressors area; neither is the "best" way, each has clearly positive and clearly negative aspects. The first can be referred to as a "Decision Matrix," the second as the "Uniqueness, Area, Reversibility, Magnitude" approach. Chapter 6 presents a more thorough discussion of decision analyses and evaluation techniques.

3.4.1 Decision Matrix

The results of the Inventory Component identify the chemical releases during the life cycle of a product. The Impact Assessment will tell us whether or not these chemical
releases or the product itself has an influence on the important ecological stressors described above. A matrix, similar to Table 3-1, can then be constructed.

Initially this matrix can be used by simply placing checkmarks within each cell as appropriate. For example, if it has been determined that somewhere in a product's life cycle, Chemical A is used, and Chemical A has been predicted to, or found to be adversely impacting a valued ecological component (through one of the assessment methods), a checkmark would be placed in the appropriate cell (Table 3-1a). The process of comparing each chemical to every ecological stressor would be completed for all chemicals used or created during the product's life cycle that could be exposed to an ecological system. Each product must be considered in a separate matrix.

The next step would be to enter an indicator of magnitude (e.g., loading) within each cell that has been given a checkmark. Perhaps the first round would simply require an indicator such as "Small," "Medium," and "Large." However, a preferable approach would be to place actual levels and their associated "units" within the matrix. For example, if the loading assessment was the highest level of analysis utilized for Chemical A, simple mass/year loadings could be entered into the cells. If, on the other hand, a site-specific exposure/effect study had been completed on Chemical A, a more sophisticated indicator could be entered into the cell (e.g., percent reduction in abundance). The other assessment methods described in Section 3.2 would provide other types of units which could be placed within the matrix. Ideally, some type of indicator would also be entered into the cell which indicated which of the assessment approaches was utilized to provide data. Table 3-1b illustrates this concept for Product "X."

A decision can be made at this point to stop or to continue with the analyses and utilize the information for either an internal or external LCA. Several decisions can be made at this point. First, a decisionmaker may determine that the product and chemicals used in manufacturing have a small enough ecological impact to proceed with manufacturing in the manner analyzed. Second, a decisionmaker may look at the data and decide that certain impact categories are too great and some processing changes must be made to reduce impacts. Third, the decisionmaker may determine that enough impact categories are too great and the product must be dropped from production. Finally, the decisionmaker may decide that further analysis is required because of the number of simplistic methods that were used with their inherent lack of ecological relevancy. The decision may be to undertake the analyses again, striving to use the more ecologically relevant approaches.

Multiple products can also be compared, as can multiple processes used for the same product, but a formalized decisionmaking technique must be employed. In all but the most extreme cases, no two products will have the same set of "checkmarks" in the matrix to allow a simple summing of the loads and selection of the "less is best" product. Almost always, several cells within the matrix will be checkmarked by one product, but not the other, and vice versa. Thus, a weighting approach will be required to compare the relative importance
### TABLE 3-1. RELATIONSHIP OF INVENTORY RELEASES TO CHEMICAL STRESSOR CATEGORIES — MATRIX

<table>
<thead>
<tr>
<th>Inventory</th>
<th>Stressor Categories*</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>CCG</td>
</tr>
<tr>
<td>Chem. A</td>
<td></td>
</tr>
<tr>
<td>Chem. B</td>
<td></td>
</tr>
<tr>
<td>Chem. X</td>
<td></td>
</tr>
<tr>
<td>Actual Product</td>
<td>X</td>
</tr>
</tbody>
</table>

*CCG = Climate Change Gases  ACID = Acidification Precursors  
OD = Ozone Depletors  PO = Photochemical Oxidants  
TOX = Toxicants  NUT = Nutrients

### TABLE 3-1a. INITIAL USE OF MATRIX

<table>
<thead>
<tr>
<th>Inventory</th>
<th>Stressor Categories*</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>CCG</td>
</tr>
<tr>
<td>Chem. A</td>
<td>X</td>
</tr>
<tr>
<td>Chem. B</td>
<td>X</td>
</tr>
<tr>
<td>Chem. X</td>
<td>X</td>
</tr>
<tr>
<td>Actual Product</td>
<td>X</td>
</tr>
</tbody>
</table>

### TABLE 3-1b. USE OF MATRIX FOR PRODUCT "X"

<table>
<thead>
<tr>
<th>Inventory</th>
<th>Stressor Categories*</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>CCG</td>
</tr>
<tr>
<td>Chem. A</td>
<td>500a</td>
</tr>
<tr>
<td>Chem. B</td>
<td>200a</td>
</tr>
<tr>
<td>Chem. X</td>
<td>28a</td>
</tr>
<tr>
<td>Actual Product</td>
<td>2.3d</td>
</tr>
</tbody>
</table>

*aTons/year, from Loading Assessment  
*bMillion liters/year, from Impact Equivalency Unit Approach  
*cMillion cubic meters/year, from Exposure Toxicity Threshold Equivalency  
*dReduction of mg biomass/liter, from Generic Exposure/Effect Assessment  
*ePercent reduction in abundance of species A, from Site Specific Exposure/Effect Assessment  
*fChronic No Observable Effect Concentration (NOEC)/Water Exposure, from Margin of Safety Analysis
of three categories of data: (1) the relative importance of each of the stressors, (2) the relative importance of each of the assessment methods, and (3) the relative importance of the magnitude of the impact. A formalized approach such as the Analytic Hierarchy Process (AHP) (Saaty, 1980) will then allow a decisionmaker to determine the preferred product or process.

3.4.2 Uniqueness, Area, Reversibility, Magnitude

The second approach the workgroup considered for decisionmaking involved developing four broad, but far-reaching decision criteria that are related to ecological health. These decision criteria force LCA analysts to consider how the impacts relate to: (1) uniqueness, (2) area, (3) reversibility, and (4) magnitude.

Uniqueness, a unitless concept, relates to an idea of importance — does the impact affect ample resources, moderately available resources, or scarce resources? For example, in the midwest United States, pasture is an ample resource, upland forest is a moderate resource, and wetlands are a scarce resource. In another area, the uniqueness of these resources may change. Resources other than land uses must also be considered, e.g., water quantity and airsheds.

Area, with units of length, refers to the spatial extent of the impacts. Preferably, actual measurements or estimates should be made. Since this is not always feasible, however, a relative scale can be applied. The terms local, regional, continental, and global are offered, but can be modified if necessary. Our concept is that local refers to site specificity (perhaps you could drive a car around it in a couple of hours); regional refers to naturally definable geographic regions (e.g., watershed, state, country); continental refers to the major continents (North America, South America, Europe, Asia, etc.); and global, of course, refers to impacts that extend across continental boundaries and are worldwide.

Reversibility, with units of time, is associated with the temporal scale over which the impacts occur. Again, actual measurements or estimates of time should be made, but some categories are suggested if actual timeframes cannot be measured. The categories of short, moderate, and long should provide enough of a decisionmaking sensitivity. Short-term refers to impacts that are expressed for days, moderate-term refers to impacts expressed for months, and long-term refers to impacts expressed for years.

Magnitude refers to the size of the impact, as determined by the impact assessment phase. For example, the loading assessment may indicate that 200 tons per year of SO\textsubscript{x} are released during the product's life cycle. Each of the assessment methods can provide these data, and the preferred approach is to use the more ecologically relevant approaches when possible (i.e., site-specific exposure/effect approach).

The four decision criteria offer a surrogate method of expressing ecological concepts of structure, function, and biodiversity, as well as the ecological levels of individuals, populations, communities, and ecosystems. While LC50s and NOELs are the types of
endpoints that scientists need to understand the ecological impacts of products, decision-makers need broader concepts that do not require a scientific background. These decision criteria should satisfy both the scientist and the decisionmaker, who in fact may be the same person.

3.5 RESEARCH NEEDS AND SUGGESTIONS

1. A critical need exists for methods and approaches for factoring uncertainty into the impact assessment process. Users need to communicate how much uncertainty exists in all steps of the analysis.

2. The inventory frequently contains information on chemical loadings to the environment. Analysts attempting to perform a life-cycle impact assessment need an organized reference (source) so that these loadings can be placed in perspective with loadings from other natural and manmade sources.

3. A critical need exists for case studies that demonstrate the application and utility of the various methods being developed for ecological impact assessment in relation to an LCA.

4. Consideration should be given to develop a technical support document for life-cycle impact assessment. Currently, guidance on the various methods exists as a diffuse body of knowledge. This guidance needs to be pulled together into a technical support document.

5. Workshops need to be held to train individuals on how to conduct life-cycle impact assessments.

6. Because of the complexity of an LCA and the impact assessment component, expert systems need to be developed to guide analysts.

7. The use of decision analysis methods needs to be further explored as applied to an LCA.

8. Chemicals in the inventory component need to be specifically identified instead of grouped in such categories as total organic carbons (TOCs), volatile organic carbons (VOCs), and suspended solids. Regulatory chemical listings under Superfund Amendments and Reauthorization Act (SARA) Title III, National Pollutant Discharge Elimination System (NPDES), and RCRA should be made available. Analytical research should be conducted on methods to achieve a complete material balance on TOC.

9. Methods to conduct toxicity tests on aquatic and terrestrial organisms on unknown mixtures of chemicals in air, water, sediments, and soil are needed.
Concurrently an understanding of how to interpret these data in the context of a life-cycle impact assessment is needed.

10. A need exists to continue to field validate our many short-cut predictive models of fate and toxicity tests. Always relying on the generally very expensive field studies cannot be done.

11. There is a need for information on the influence of habitat disturbance on maintenance of habitat fragments.

12. A clarification should be made that life-cycle impact assessment methods deal with potential impacts rather than attempt to estimate actual impacts.

13. At present, it is recommended that impact assessment concentrate on methods that deal with potential impacts, and in particular, on the impact equivalency method.

14. High priority should be given to the development of equivalency factors for a number of impact categories, such as resource depletion, human and ecological toxicity, and environmental disturbances.

15. There is a need for techniques for quickly estimating biodiversity changes and alterations in habitats.

16. The use of generic approaches has the potential to reduce data needs to a significant extent, but the generic analytical tools are not as well developed as site-specific measures.

17. For areas of significant impacts, there is a strong need for data capable of linking particular releases to particular impacts, in a quantitative cause-and-effect relationship, within an acceptable level of uncertainty.
CHAPTER 4.0

HUMAN HEALTH IMPACT ASSESSMENT

4.1 DEFINITIONS

A human health impact is defined as an adverse effect to human health. In the context of life-cycle assessment, a human health impact can be defined as the reasonable anticipation of an adverse effect to human health.

4.1.1 Human Health Effects Of Concern

An analysis of impacts on human health should be included in an LCA. The depth of the analysis of health impacts that can be performed with data gathered from an LCA is a function of both how readily one can make inventory data serve as either a surrogate for human exposure or a basis for further exposure analysis. The available data set for characterizing health impacts affects the loadings or resource depletion reported in the inventory.

An analysis of health impacts, mainly injuries, associated with the processes themselves (as opposed to releases and consumption) can be done without exposure analysis. For example, vehicular accident rates per mile are available as actuarial data and can be used to compare processes that have different vehicular transportation needs. Similarly, occupational health statistics for commercial activities are often collected by governments and insurance providers and can be used to compare health impacts of activities.

4.1.2 Stressors

Stressors are conditions that may lead to human health impacts (e.g., loading of a toxic chemical to the ecosystem); they provide a linkage between the LCI data and human health impacts. Health impacts are difficult to measure and often must be predicted. Stressors can be evaluated from inventory data and provide the basis for the prediction of impacts. For the toxicity stressor, the simplest form may be the mass loading output from an LCA. In the most developed form, the toxicity stressor would describe the route, duration, frequency, and level of exposure of any particular stressor to human individuals or populations. The stressor concept is explained in Chapter 2. Chapter 3 discusses its application to ecological impacts.

4.1.2.1 Types of Stressors. The following list is provided to further define the concept of a stressor. This list may not be all inclusive but serves to illustrate the concept.

1. Chemical toxicity.
2. High energy radiation.
3. Conditions creating the potential for accidents.
4. Pathogenic organisms.
5. Food or water deprivation.

The potential for chemical toxicity arises from exposure to chemical products or wastes entering the environment as a result of activities during the life cycle of a product. Substances released to the environment are identified and quantified in the LCI.

High-energy radiation may result directly from the presence of radioactive materials in the product or waste. Alternatively, exposure to ultraviolet (UV) radiation may result from secondary effects (e.g., depletion of stratospheric ozone). Accidents may result from the transportation and storage of raw materials, intermediates, or products; worker injury on the job; or from a variety of other activities evaluated in the context of a specific LCA. Food or water deprivation may result from resource depletion.

4.1.2.2 Categories of Effect and Stressors. Several broad categories of human health effects and stressors can be considered in the study design for an impact assessment:

<table>
<thead>
<tr>
<th>Categories of Effect</th>
<th>Categories of Stressors</th>
</tr>
</thead>
<tbody>
<tr>
<td>Disease</td>
<td>Chemical Toxicants</td>
</tr>
<tr>
<td>Injury</td>
<td>Radiation</td>
</tr>
<tr>
<td>Discomfort</td>
<td>Physical Trauma</td>
</tr>
<tr>
<td>Deprivation</td>
<td>Biological Organisms</td>
</tr>
<tr>
<td></td>
<td>Depletion of Sustenance</td>
</tr>
<tr>
<td></td>
<td>Nuisances</td>
</tr>
</tbody>
</table>

Diseases include infectious diseases as well as illness due to physiological dysfunctions (e.g., cancer, heart disease). Injuries include the physical effects of trauma. Discomfort is a broad category including effects on quality of life of imposed nuisances such as odor, dust, and noise. (Noise may also cause injury as well as anxiety about potential health impacts.) Deprivation is a broad category of potential effects of lack of food or water.

4.1.3 Targets to Consider

<table>
<thead>
<tr>
<th>Development</th>
<th>Hematopoietic</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cell Growth (Cancer)</td>
<td>Respiratory</td>
</tr>
<tr>
<td>Heritable Genetic Change</td>
<td>Digestive</td>
</tr>
<tr>
<td>Physiological Systems</td>
<td>Excretory</td>
</tr>
<tr>
<td>Reproductive</td>
<td>Immune</td>
</tr>
<tr>
<td>Nervous (Central and Peripheral)</td>
<td>Skin</td>
</tr>
<tr>
<td>Skeletomuscular</td>
<td>Endocrine</td>
</tr>
<tr>
<td>Cardiovascular</td>
<td></td>
</tr>
</tbody>
</table>
4.2 STUDY DESIGN ISSUES

4.2.1 Matching the Scope of the Impact Assessment With the Intended Use

The expectations for the workshop as expressed by the participants included the following questions and needs that related to study scope and design:

- Identify practical impact assessment methods; what can be applied or done now?
- Can options for impact assessment be identified?
- Can screening criteria (technical or objective) be developed?
- The level of detail of LCAs should be kept within the anticipated use.
- How are missing data dealt with?

The varied purposes of LCAs potentially encompass labeling; certification of products; comparison of products, processes, and sites; product development; pollution prevention; and use as a management tool. They may be developed for private internal use or for external public purposes and may range in scope from site specific to global in nature. As a result, no single method for human health impact assessment can or should be recommended.

In some cases, as in screening LCAs, only readily available data are used. In other more extensive and elaborate LCAs, the development of a human health database will be part of the overall LCA process. In all cases, it will be necessary to decide which data are needed and how extensive the database needs to be. Resource capacities (i.e., time, money, laboratories, technically qualified people) frequently limit the development of every data element for every exposure potential for each stressor. As a result, decisions will have to be made and priorities set as they are in other risk assessments for determining what data are needed to arrive at a valid and reasonable conclusion. Stated simply, the scope of the impact assessment must be bounded.

For each stressor identified (as discussed below), exposures (e.g., occupational, environmental, product use, disposal) and exposure routes (inhalation, ingestion, contact) need to be considered. For each medium (i.e., air, water, soil) and for each exposure route a variety of health effects are possible and may need to be considered. In addition, the NOEL or other standard or criterion of safety should be compared to the concentration of the stressor in each of these media. As a result, decisions need to be made on the type and extent of testing for health impacts.

When making these decisions, it is important that the testing/evaluation program proposed should retain or achieve the confidence level needed for its intended application.
In the context of this discussion, the term confidence level refers to the overall level of confidence one has that the life-cycle impact assessment will provide an answer to the question at hand rather than the specific confidence intervals or boundaries associated with any specific test result. Factors that are meaningful in determining the confidence level required for the human health impact assessment include the degree of stressor hazard (which is a function of its toxicity and exposure potential) and the importance or social value placed on the health impact associated with the agent or stressor. For example, a mild transient sensory irritation effect is of less concern (i.e., lower social value) than an effect on reproduction or cancer. Likewise, the same surface water concentration of a chemical in a drinking water reservoir is likely to have a greater social value than the same concentration of that chemical in a nonpotable water body. Specific criteria for low, medium, and high confidence level testing programs would need to be developed, depending on the nature of the stressor and purpose of the LCA. For example, a product-related impact assessment might have acute and other short-term studies in an impact assessment program for which a low confidence level is acceptable, but in most cases requires chronic exposure studies and mechanistic studies to achieve a high confidence level. If this same product had an intermediate exposure potential and social value, an impact assessment providing a medium level of confidence might be all that is needed to satisfy the purpose of the LCA.

Since health assessments are by their nature iterative, it is possible to approach the assessment as an iterative process with feedback loops. The analysis may begin with the most readily available data or clearly critical studies and is continued until the desired confidence level of the human health impact assessment is achieved. As a result, the program scope may vary widely yet achieve the same confidence level. For example, an LCA for a global issue is very likely to require far more data than an LCA performed for the purpose of making an internal decision or for labeling a low production volume chemical. This strategy, which is amenable to achieving consensus, would provide the necessary confidence at a minimum level of detail, thus conserving limited resources. It is fully consistent with the concept expressed in Chapter 6: "The optimal amount of analysis is the minimum that will distinguish between policy alternatives." (C. Wolf)

Initially, there should be no predetermined limits on the extent of the impact assessment. If a scoping process is undertaken to limit the number or extent of stressors evaluated, or to exclude a type of impact or otherwise limit the scope of analysis, it should be recognized as being at least somewhat subjective. The importance of transparency in assumptions, data categories, and methods is imperative. A number of existing LCA programs (Impact Assessment Matrix Approach, Danish Program, Environmental Choice Program, German Eco-Labeling Program) perform this activity, recognizing that professional judgment, sometimes in the form of an expert panel or peer review, is necessary to optimize the result. Finally, it should be recognized that all of the iterative tiers or stages of the impact assessment provide critical feedback loops to the inventory analysis.
4.3 METHODS AND MODELS

4.3.1 Introduction

The purpose of this section is to describe the range of potential approaches to the valuation of human health impacts in the context of LCA. Although the range of possible approaches is provided for the reader, a single approach is developed that is based on the level of information typically available from LCI studies. The philosophy is to provide the user of this document with a sense of the range of possible approaches while still maintaining a linkage to the LCI data. This philosophy provides sufficient flexibility to make this document useful for the practice of LCA today and in the future.

4.3.2 Evaluation of Stressors

Stressor evaluation is important for the prediction of human health effects. The data necessary to begin the evaluation of stressors comes from the LCI together with the chemical, physical, and biological properties of each substance.

Presented below are potential methods that could be used to evaluate stressors in the context of LCA. This list is not intended to be mutually exclusive. There are a myriad of possible combinations of these techniques. This presentation is intended to give the reader a range of the possible approaches presented in order of increasing complexity. The reader is encouraged to review Chapter 1 on scoping for a discussion of the appropriate level of complexity for different study applications. Chapter 2 presents and discusses the possible approaches to impact assessments. Their application to ecological health is discussed in Chapter 3 and to human health, below.

To outline the approaches to stressor evaluation for human health impacts, an example of the step-wise evaluation of a toxic stressor, focusing primarily on atmospheric pollutants, is described below. Similar approaches could be used for other media appropriate to a specific LCA.

Level 1 — Loading Assessment. Mass loading is obtained directly from the inventory and may be expressed as a mass of substance A per use equivalent of a product I. Aggregation for multiple chemical releases is accomplished by summing all chemical releases. The stressor is expressed as total mass of all pollutants released per unit equivalent. In using this approach for comparisons with other LCA studies, one might make the assumption the lower total loading would indicate a superior product or process from a human health standpoint. This approach has been called the "less is best" approach to impact assessment. It should be noted that combining the assessment of various health effects is not advocated by this method. This method does not explicitly value the human health implications of various stressors but simply the aggregates all of the stressors. Section 4.4 provides a more detailed discussion of aggregation techniques.
Level 2 — Impact Equivalency Assessment. The mass loading may be modified by the use of some toxicity standard, e.g., a reference dose. For chemicals that partition to the atmosphere, a critical volume (i.e., volume of air contaminated to the toxic standard) can be calculated by dividing the mass loading by the toxicity standard expressed in the appropriate units. Aggregation can be accomplished by summing the critical volumes across chemicals dependent on the health effect of concern (see Section 4.4). The stressor is expressed as a volume of air contaminated per use equivalent of product.

This approach might be most useful for evaluating the toxicity stressor in air in the vicinity of the loading source. If used in the workplace or immediate vicinity of product use, loading factors should be modified to reflect release within the confines of the work or use area.

Level 3 — Loading Modified by Toxicity and Persistence and Severity of Effect. For evaluation of stressors due to ambient atmospheric exposures and global buildup, mass loading should be modified by both toxicity and persistence. It is rational to assume that persistence in the environment is positively correlated to potential for exposure. The lifetimes of many organic chemicals are readily available from the literature.

Additionally, the severity of the health endpoint for which the toxicity indicator has been developed should be considered. The stressor should be modified by an additional factor to account for the severity of the effect of concern. The stressor can then be expressed as a product of the critical volume divided by the health benchmark, the atmospheric lifetime, and the severity of the effect of concern.

Level 4 — Generic Exposure/Effect Assessment. This approach utilizes the exposure/assessment method described in Chapter 3. It defines a Mackay unit world model and takes into account mass loading, a distribution between media, persistence, bioaccumulation, and soil adsorption to estimate an ambient concentration in each medium in the unit world. Comparison of the ambient unit world concentration with toxicity and severity of effect (as described above) provides a measure of the stressor. For example, the calculation will show the ambient concentration is some multiple (or fraction) of the toxicity standard. Aggregation occurs by summing the individual stressors within the bounds described in Section 4.4.

Given the level of information typically available in an LCI, it is likely that this approach will be the most feasible to value the potential for human health impacts in the context of an LCA. This general approach is presented in greater detail in Subsection 4.3.3.

Level 5 — Site-Specific Risk Assessment. This approach utilizes classical risk assessment methods. It requires site-specific data on both chemical contamination or resource depletion levels and the exposed population. The stressor is expressed in terms of human exposure route, duration, magnitude, and frequency. Actual risk is estimated in terms of individual probabilities of occurrence or population incidence.
4.3.3 Detailed Methodological Description - Example: Level 4

In the previous subsection, the range of possible approaches to stressor evaluation were presented in brief. In this section, one of the approaches, level four, which seems to be most adaptable to the current state of LCI data, is presented in more detail.

In human health risk assessment it has been a common practice to establish thresholds of effect. A typical risk assessment based on threshold levels would compare exposure data to the threshold for adverse effects. The concept underlying this technique is that exposures below the thresholds would not adversely affect the population of concern. These thresholds can be quite useful for risk characterization and certain risk management decisions; however, there is some controversy regarding the ability of the available analytical tools to accurately predict adverse effect thresholds for human populations with a wide range of intraspecies sensitivity.

Given the controversy that exists regarding the use of adverse effects thresholds in risk assessments that are relatively data rich, it is difficult to envision an effective use of this technique in the context of an LCA wherein the exposure data are likely to be very poor or nonexistent. Although there are many sources of information on effects thresholds (e.g., occupational exposure levels, NOELs, reference doses/concentrations, etc.), the exposure information required for the comparison with these levels would not be available in most LCA studies.

It may be infeasible to estimate threshold exceedances in a life-cycle impact assessment that provides little or no information on human exposure. In this case, the consideration of human health impacts can only be assessed on a surrogate basis. In other words, certain assumptions will have to be made to replace the missing exposure data. For example, if one were to assess the impacts associated with a particular chemical loading where the only exposure information available were limited to the magnitude of release to a given media, the operative assumption might be that exposure in that media is directly proportional to the magnitude of the loading. It would not be feasible to accurately extrapolate an ambient concentration or an absorbed dose (either for an individual or population) from the magnitude of the loading without extensive additional information.

In this scenario, it would not be feasible to attempt to predict threshold exceedances. One method to consider an adverse effects threshold in this case would be to simply weight the magnitude of the loading by the inverse of the threshold level relative to other loadings. For example, if the agent in this case were an extremely potent reproductive toxin and had been ascribed a very low threshold level in the available literature, the magnitude of the loading could be multiplied by the inverse of the effects threshold (resulting in a very large weighting factor).

Other assumptions can be added to this technique if additional data are available. If, for example, the loadings to the environment occurred in proximity to a major population center, an additional weighting factor could be added to the evaluation.
A common assumption in human health risk assessment is that there is no threshold for genotoxic carcinogenesis. For carcinogens, a potency or risk factor is often calculated from the available data. This factor should be directly proportional to the slope of the dose response curve for the cancer endpoint and thus could be applied directly to the inventory data as a weighting factor.

Using this method, all of the loadings reported in the LCI for chemicals with available threshold levels or risk factors could be weighted proportional to their perceived toxicity. This method assumes that (1) the stressor is proportional to environmental loading, appropriately modified by persistence and environmental distribution; (2) health effects benchmarks or standards are available or can be derived for all chemical loadings, and these can be used in the estimation of the magnitude of the stressor; and (3) stressors are an appropriate surrogate for health impacts. Although this method does not result in specific estimates of threshold exceedances, it can provide a relative indicator of the potential human health impact resulting from environmental loadings.

Exposure data are often lacking in LCI data. Consideration of the intrinsic variability of the exposed population (sensitive subpopulations), different exposure pathways, worst case loadings, and the averaging time of the effect of concern inexorably correspond to the information provided in the inventory. Given that these data are often not collected in the inventory, the analyst can (1) endeavor to collect such data; (2) attempt to compensate for missing data with assumptions regarding the available data; or (3) ignore exposure mediated impact issues. The choices between these options will depend on the purpose and intended use of the study and the resources available for the analysis.

The pathway of exposure often determines the ultimate impact to human health. Often the only information from the inventory regarding the exposure pathway is the media receiving the loading. In these cases, the analyst must make assumptions about the exposure pathway. One tool for making such assumptions is the Mackay unit world model (see Chapter 3). This model takes into account mass loading, distribution between media, persistence, bioaccumulation, and soil adsorption to estimate an ambient concentration in each medium per unit loadings. For example, for a loading to the air medium, the Mackay model would utilize physical properties and degradation rates of the chemical loading to estimate the equilibrium concentrations in each medium (i.e., air, water, soil, and biota). The results of this model would be expressed as ambient relative concentrations in each medium.

The resultant estimates of ambient concentrations are subsequently compared to the appropriate health benchmark level. Using the approach discussed above (i.e., estimated exposure divided by the health benchmark) in instances where the health effects benchmark is exceeded, the model would predict a score of greater than 1. Conversely, ambient concentrations less than the benchmark level are characterized as fractions of 1. Within the bounds established under Subsection 4.4, these scores would be combined into an overall score representing the human health stressor for the product of process of concern. These scores are not indications of absolute risk but rather relative scores that represent the
human health effects stressor. These scores are valuable only in comparisons with other scores generated by the same method and do not represent absolute indicators of human health effects.

4.4 HEALTH RISK CHARACTERIZATION

4.4.1 Issues of Valuation

Even in cases where the data are rich enough to support a quantitative estimate of absolute excess risk (see below), the risk estimate obtained should only be converted to a dollar value (or to another common metric such as utility) with caution. The techniques used to estimate the value of a statistical life (VOL) are controversial; even if this hurdle is overcome, the range of empirical estimates is too broad to allow for any single point estimate to be carried through the analysis as a "black box" output (e.g., values ranging from approximately $100,000 to $7,000,000). Techniques to estimate the cost of a nonfatal disease or injury are perhaps less controversial (e.g., values obtainable from workmen's compensation guidelines), but there may be little utility in monetizing the morbidity impacts if the mortality impacts in an LCA dominate and yet cannot be monetized.

In addition, considerable controversy exists over whether or not "lives saved" or "life-years saved" is the appropriate metric for valuing mortality impacts; the former measure treats saving the life of an 80-year-old and that of a 5-year-old equivalently. Both measures should be computed and carried through the analysis whenever the risk data are available; even if specific information is unavailable, any plausible differential impact (of two products or processes) on the age distribution of the population-at-risk should be highlighted in the impact assessment. Effects on other special populations unrelated to the age distribution should also be highlighted; examples include pregnant women and minority, low-income, or other groups already at a disadvantaged position.

If health risk estimates are converted to a monetary or other scale to be combined with other effects, deaths predicted to occur in the medium-term future (i.e., 10 to 40 years hence) should not be discounted, even if elsewhere in the analysis future costs are discounted to account for the time value of money.

4.4.2 Issues of Aggregation

Because of the different techniques used to estimate consequences, and the subjective and highly variable weights different individuals place on avoiding various diseases, the reader is cautioned against combining cancer risk estimates and noncancer risk estimates into a single health effects score for a product or process. This may not be a significant drawback. In some cases, the products or processes being compared may only differ according to one type of health effect or the other (e.g., lead-acid versus nickel-cadmium batteries, or ethylene dibromide versus ethylene dichloride for fumigation). In many other cases, cancer risks will tend to dominate other risks, especially when the processes involved are already under relatively tight control. In still other cases, technological or other controls...
to bring cancer risks to acceptable levels will also eliminate or virtually eliminate concomitant health effects that have thresholds for exposure.

However, in cases where both types of effects must be considered, there is one possible approach that would avoid the apples and oranges problem inherent in combining presumed threshold and nonthreshold effects. This approach would be to seek consensus on two "bright lines" — one a numerical cancer risk level and the other a function of the relevant RfD for a noncarcinogen — that are believed to represent roughly equivalent breakpoints between de minimis and significant risks. Then, the total number of persons facing exposures above either bright line could be used as a common metric for comparing the health consequences of different products or processes. In cases where the available LCI data do not support the calculation of the exposed population, the method described in Section 4.3 (i.e., the health benchmark divided by the exposure level) could also be used as the common metric.

If sufficient data are available, the characterization of cancer risks should be expressed whenever possible using two different measures: the estimated number of excess cases in the population (D) and the excess probability of cancer for a maximally exposed individual (RMEI), as defined by the EPA or other guidelines relevant to the medium of interest. If a single measure of cancer effect is desired, the two disparate measures of impact can be combined into a single (though only a relative) measure by computing their product (either weighted or unweighted). Thus, for example, a diffuse but widespread exposure that was predicted to cause 100 deaths nationwide but expose no individual to a risk greater than $10^{-6}$ would be deemed equivalent to a geographically concentrated risk that was predicted to confer a risk of $10^{-2}$ on a community of 10,000 people ($10^{-2}$ deaths times a $10^{-2}$ RMEI also equals $10^{-4}$). (If non-equal weights were applied to the two components of this product, the two situations would not be equivalent, but they could still be compared.) As stated above, when concentration data are not available, a relative cancer risk score can be obtained by multiplying the amount of pollutant emitted by the cancer potency factor. For noncancer effects, a relative score can be obtained by dividing the amount emitted by the applicable RfD.

### 4.5 USES AND PRESENTATION OF LIFE-CYCLE HEALTH IMPACT RESULTS

Perhaps other impacts included in a health life-cycle impact assessment are susceptible to considerable misinterpretation and misuse. Not only is the public easily capable of misinterpreting health risk assessment data, but, as outlined above, the data and their interpretation are subject to considerable uncertainty and controversy. Therefore, extreme care must be exercised in presenting and communicating the results of a life-cycle human health impact assessment. Two fundamental principles should be kept in mind:

1. The level of detail required in an LCA health assessment will vary depending on the intended use of the LCA.
2. The data resulting from a health impact assessment must be weighed against other important social factors.

4.5.1 Level of Detail Required in Health Assessment

As noted above, the extent and certainty of the data required in a health life-cycle impact assessment are the minimum required to discriminate among the technical or policy options under consideration.

Given the current impossibility of assessing all of the relevant health risks of a product, special care should be exercised in LCAs intended for public or external (noncompany) use. In our opinion, the current state of life-cycle human health impact assessment methodologies will not support, standing alone, public policy decisions (including legislation, regulation, or labeling) relating to products. Although life-cycle human health assessment can provide important input into these public decisions, those decisions will have to continue to be based on more traditional decisionmaking methods. By contrast, LCAs intended solely for internal company use often will be able to rely on less detailed data and can tolerate greater levels of uncertainty in that data.

Table 4-1 presents several cautions on the use of LCAs, in particular applications given the substantial limitations now existing in the available models and data.

4.5.2 Consideration of Nontechnical Risk Issues

The estimation of the quantitative health risks from a product are not alone sufficient to support risk-based product decisions. Numerous other nontechnical factors are relevant to the acceptability of human health risk. For example, one must consider such issues as the reversibility of the health impact (whether it is an incurable cancer or a treatable kidney ailment) and the equity of the risk distribution (whether the poor or minorities bear a disproportionate share of the impact). Table 4-2 contains a more complete list of the nontechnical factors relevant to health risk assessment.

4.5.3 Treatment of Uncertainty

1. "A decision made without taking uncertainty into account is barely worth calling a decision." (Wilson, Crouch, and Zeise, 1985) In this spirit, we strongly recommend that practitioners quantify and display the uncertainty [as both a probability density function (PDF) and a cumulative distribution function (CDF)] for whatever absolute measures are computed. Schematic guidelines for undertaking such quantification and presentation can be found in Finkel (1990), among other sources. In particular, we caution practitioners not to use lack of data as an excuse for not considering uncertainty — although the quantification may have to involve expert judgment rather than statistical techniques, it is precisely these cases where ignoring uncertainty can lead to the most frequent and most severe errors (e.g., the 1986 Challenger accident).
TABLE 4-1. CAUTIONS ON USE OF LIFE-CYCLE HEALTH ASSESSMENT DATA IN PARTICULAR APPLICATIONS GIVEN THE EXISTING STATE OF THE ART

<table>
<thead>
<tr>
<th>LCA Application</th>
<th>Cautions on Use of LCA Health Assessment Data</th>
</tr>
</thead>
<tbody>
<tr>
<td>Corporate Policy Decisions</td>
<td>Data can be very useful in making broad policy decisions, such as whether to implement broad recycling programs or energy conservation efforts.</td>
</tr>
<tr>
<td>Corporate Product Design or Modification</td>
<td>Data are most useful when comparing two closely related products (e.g., paper bags made from different processes). Data are much less useful when comparing different products (e.g., paper versus plastic bags). In either case, limitation on health assessment methods must be factored into the decision and other forms of data sought.</td>
</tr>
<tr>
<td>Forecasting the Impacts of New Technology</td>
<td>May have limited value. Will be most useful when comparing a new technology that can be analogized to a related existing technology.</td>
</tr>
<tr>
<td>Facility Siting</td>
<td>Of limited value. Site-specific health data are not easily obtained and are subject to considerable uncertainty. In any event, the EIS is most frequently used to make siting decisions.</td>
</tr>
<tr>
<td>Public Education</td>
<td>Studies can be used to educate the public about the need to assess the cradle-to-grave impacts of products. However, they should not be used alone to recommend consumer action relating to products or product types.</td>
</tr>
<tr>
<td>Public communications, advertising, product claims</td>
<td>Of very limited value. Should only be used as one source of data in justifying product claims.</td>
</tr>
<tr>
<td>Public policy, legislation, regulation, labeling</td>
<td>Of very limited value. Should only be used as one source of data.</td>
</tr>
</tbody>
</table>
TABLE 4-2. NONTECHNICAL ISSUES RELEVANT TO HEALTH RISK ASSESSMENT

<table>
<thead>
<tr>
<th>Issue</th>
</tr>
</thead>
<tbody>
<tr>
<td>Reversibility of effect</td>
</tr>
<tr>
<td>Chronic versus acute effect</td>
</tr>
<tr>
<td>Impact on future generations</td>
</tr>
<tr>
<td>Impact on children or other special subpopulations</td>
</tr>
<tr>
<td>Voluntary versus involuntary exposure</td>
</tr>
<tr>
<td>Familiar versus unfamiliar</td>
</tr>
<tr>
<td>Benefit of assuming risk</td>
</tr>
<tr>
<td>Beneficiary of risk (e.g., corporation versus individual)</td>
</tr>
<tr>
<td>Equity of risk distribution</td>
</tr>
<tr>
<td>Personal nature of risk</td>
</tr>
<tr>
<td>Catastrophic risk (e.g., plant explosion)</td>
</tr>
<tr>
<td>Dread of risk</td>
</tr>
<tr>
<td>Cost of controlling risk</td>
</tr>
<tr>
<td>Human versus natural cause</td>
</tr>
</tbody>
</table>

When risks are quantified, the corresponding uncertainty estimate may take the following form: "This product/process is estimated to expose X persons to risks above de minimis levels, where X is lognormally distributed with a median of 100,000 and a geometric standard deviation of 10." Even when no exposure data are available, one can still calculate the uncertainty in a statistic such as $Q = \sum_{i} E_i / \sum_{i} R_{fD,i}$ (where the $E_i$ are kg of emissions of each chemical and the $R_{fD,i}$ are the appropriate reference doses), by propagating the uncertainty in each emissions estimate.

2. Display the PDF and CDF for the ratio of any two absolute or relative measures that are compared in the product or process life-cycle impact assessment. This is crucial because the uncertainty in the comparison may be even larger (contrary to common misconception) than the uncertainty in either risk computed in isolation (Finkel, 1991). The description of uncertainty in the comparison would take the following general form: "The cancer risk for product A is between 10 and 200 times greater than that for product B, at a 95 percent degree of confidence." Or perhaps "A is between 8 times less risky and 20 times more risky than B; although the mean value of Risk A is twice that of Risk B, the ‘noise’ in the comparison is larger than the ‘signal’ and a confident decision cannot be made about which is truly larger."

3. One issue the work group could not resolve is how to deal with different cancer risk estimates for substances with different weights-of-evidence or classifications. Rather than provide specific guidance, we offer four options:

a. Ignore possible differences between Class A, B, and C carcinogens.
(60)

b. Divide all risk estimates for Class C by an arbitrary factor, perhaps 10.

c. Produce separate scores for the sum of all (A and B) and all (C) carcinogens.

d. Express qualitatively any special cases without folding judgmental probabilities into the risk estimates (e.g., "Cancer risk score is X, conditional on the chemical being a human carcinogen, and zero otherwise: here are the reasons why scientists are divided about whether the chemical is/is not relevant to humans...").

4. The life-cycle impact assessment must express limitations on use of the results of the analysis (e.g., "This analysis says that paper cups cause more (less) cancer risk than plastic cups, if all paper cups are produced using a chlorine process, the regulations currently in force in California, etc."). The impact assessment must also explain what the risk estimates mean (e.g., "This comparison does not imply that all residents near pulp mills face greater (lower) individual risks than residents near plastics factories, only that aggregate risks are greater in the former (latter) case.").

4.5.4 Guidelines

Given the data and other limitations that apply to human health impact assessment, the authors of LCAs are cautioned to observe the following guidelines:

- Authors of LCAs must exercise care in extrapolating beyond the specificity of the data contained in the LCI.

- The assessment of classes of health effects should be reported separately and not combined. Some aggregation can occur as detailed in Section 4.4.

- The limitations of the state-of-the-art in health impact assessment should be made clear.

- Where possible, the uncertainties associated with the health impact assessment should be quantified.

- Risk estimates of health impacts should not be presented without accompanying characterization of the qualitative or quantitative confidence level, including which part of the exposure distribution the estimates relate to (i.e., worst case, high end, average).

- Risks for high-end exposures of one product or process should not be compared to risks for average exposure to another product or process.
4.5.5 Presentation of Life-Cycle Health Impact Assessment Results

Because the techniques of assessing the health impacts of products in the context of an LCA are in their infancy, there are few models for data presentation. It is assumed that the format of data presentation will vary depending on the purpose of the LCA. A few general rules should guide the presentation of the health impact assessment regardless of the LCA’s purpose:

- The results should be clearly separated into quantitative and nonquantitative sections.
- The assessment should be closely linked to the inventory. It should be clear to the reader which data in the inventory are being relied upon.
- The analysis method should be clearly identified and important underlying assumptions stated. As noted above, the LCA should be explicit about known limitations of the analysis methods employed.

4.6 FEEDBACK ISSUES IN THE LCA PROCESS

As discussed in Chapter 1, an LCA is a process to evaluate the environmental burdens associated with a product, process, or activity by identifying and quantifying energy and material usage and environmental releases to assess the impact of those energy and material uses and releases on the environment and to evaluate and implement opportunities to effect environmental improvements. The life-cycle impact assessment is only one of three components of a life-cycle assessment. Although this workshop has focused on this specific life-cycle component, it is necessary to discuss the implications that this stage has on the other life-cycle components.

The first component, life-cycle inventory or LCI, quantifies the energy and material use and environmental releases associated with all life stages of a product, process, or activity. The data generated from an LCI serve as inputs to the life-cycle impact assessment in order to characterize and assess the effects associated with the life stages of a product, process, or activity. The remaining LCA component, life-cycle improvement analysis, evaluates the opportunities to reduce those impacts.

The LCA process is not necessarily a linear process, but rather a dynamic and iterative process. For example, changes in material inputs to a manufacturing process or changes in the production process itself — whether the result of the impact assessment or improvement analysis — may trigger the need for an updated inventory. New information pertaining to toxicity will trigger the need to update the impact and improvement analyses. This section discusses the interaction of the human health impact assessment stage with the other stages of an LCA.
4.6.1 Impact Assessment and Inventory

When performing the impact assessment, it may be determined that the data generated from the inventory may not be sufficient for assessing human health impacts. For example, emissions data averaged over a annual period will mask information on peak emissions and the duration and frequency of those peaks. However, peak emission rate data may be necessary to assess acute exposures or effects; therefore, it may be necessary to revisit the inventory to attempt to determine this information. Oftentimes, however, these data are not available.

Many past inventories have reported releases of categories of pollutants such as hydrocarbon emissions, total organic emissions, etc. Such data may be insufficient for assessing associated health impacts. Here again, the LCI should be revisited to determine the speciation of classes of compounds. This speciation can be conducted in several ways. Preferably, the original source of data should be consulted. However, if the original data source does not provide speciation information, the scientific literature should be consulted. An example may be inventory data that list the pounds of hydrocarbons emitted into air associated with the life cycle of a material. If the original data source for that emission factor did not provide a breakdown as to which hydrocarbons were actually released, other data sources could be consulted. Emission test results performed on the emissions from the production process in question and reported in the literature may list the specific hydrocarbons emitted and the percentage of each found in the emissions. This information could then be used to disaggregate the original emission factor into its components.

4.6.2 Impact and Improvement

The information resulting from an impact assessment has relevance to the component. The results of an impact assessment may suggest scenarios to be explored in the improvement analysis. If the human health impact assessment points to some dominant impacts, for example occupational safety and health impacts, the improvement component could explore methods of reducing those impacts. If a specific stage in the life cycle of a material is found to pose the greatest impact, the improvement analysis could focus on those pertinent stages. However, it should be kept in mind that during the evaluation of opportunities to reduce the impacts under examination, other new impacts might unintentionally result. For example, reducing chemical exposures in the workplace may increase emissions to the ambient environment. Therefore, it is necessary to revisit the inventory and impact assessment for each alternative improvement scenario examined.

4.6.3 Generic Improvements to Health Component of LCA

Both the data and the models likely to be used in the human health impact assessment component of LCAs have considerable research needs. Among the areas needing attention are the refinement of existing models, the filling of data gaps, and the development of entirely new models.
4.6.3.1 Model Improvements. There exist considerable scientific controversies over key issues associated with the assessment of certain health risks. These disputes range from conflicting opinions over fundamental issues (e.g., do threshold doses exist for some carcinogens?) to disputes over the impacts of specific stressors (does tetrachlorobenzene-p-dioxin (TCDD) produce chronic health effects at ambient exposure levels?). Among the improvements needed in our current assessment models are the following:

- Refinement and validation of biologically-based dose-response models, such as cell kinetics models, especially for compounds thought to be non-genotoxic.
- Development of models for the possibly quite substantial natural variations in human susceptibility to cancer.
- Improvement in the ability of models to account for the cumulative, synergistic, and antagonistic interactions of various health stressors.
- Improvement in the ability of models to handle very large numbers of stressors. It is difficult enough to generate a valid risk assessment for a single chemical. Product LCAs may involve an assessment of hundreds, if not thousands, of discrete stressors.
- Development of models to account for variations in human susceptibility to chemical stressors due to compromised health status.
- Improvement of the models available for predicting human exposure to environmental loadings in the context of LCA.

4.6.3.2 Filling of Data Gaps. The many types of data that need to be collected to improve life-cycle health impact assessments include:

- Exposure data on many of the health stressors of concern, including ambient monitoring data and personal exposure data outdoors and indoors.
- Short-term and long-term bioassay data on the majority of substances for which no such data now exist.
- Speciation of effluent data into individual substances. Data such as pounds of hydrocarbons or pounds of kraft mill effluent have little if any value in health impact assessment.
- Data on unintended use of products that might present significant health risks. Typical examples of the unintended use of a product are the use of a plastic container to hold food after the container's original purpose has been fulfilled, or the use of gasoline to clean metal parts in a home workroom or garage.
• Data on human exposure to health stressors from nonmanufacturing, non-point sources. For example, there is relatively little data on human exposure to gases or other chemicals released from products and product packaging.

4.6.3.3 Development of New Models. In some instances, entirely new models will need to be developed to improve health analyses for:

• Estimating the linkages between resource depletion, ecosystem damage, and human health impacts.

• Predicting the physical and other health impacts imposed by the fear of a product or its manufacture or disposal — whether those fears are real or imagined (e.g., sickness induced by the fear of radiation from a nuclear plant or toxics from a nearby incinerator).

• Predicting the indirect health effects of individual changes in welfare resulting from a product decision. An example is increased alcoholism induced by boom and bust cycles in mineral production.

• Evaluating social impacts; for example, estimating the possible indirect health effects of the macroeconomic impact of product decisions.
CHAPTER 5.0

RESOURCE DEPLETION

5.1 INTRODUCTION

Natural resources typically supply inputs to a product, process, or activity life cycle in the form of energy and raw materials. Previous chapters have dealt with impacts to ecosystems and human health associated with the use of natural resources, e.g., the loss of assimilative capacity, habitat destruction, aesthetics, and climate change. This chapter focuses specifically on the depletion of natural resources as characterized by their relative or absolute scarcity. In this context, the term resources refers to any component of the natural environment, including air, water, land, and biomass, used as an input to the system under evaluation. The definition excludes human capital inputs (e.g., labor) and processed natural materials.

The depletion of natural resources may be of concern because of their intrinsic worth or because future generations may not inherit the same quantity or quality of resources enjoyed by the current generation. In the first instance, the intrinsic or "existence" value of the resource is wholly unrelated to any actual or potential use of the resource. One may attach great value to the existence of a landscape feature regardless of whether or not the individual intends to experience that landscape in any way and regardless of the interests of future generations. Methods to estimate existence values are currently the subject of a good deal of discussion in the economic literature. Incorporation of that theory and methods into the LCA context is left to future research. This chapter focuses on resource depletion, primarily in the context of sustainable development.

5.1.1 The Concept of Sustainable Development

Definitions of sustainable development abound, but all involve the notion of bestowing on future generations some continuing wealth of natural resources. A commonly cited definition, which broadly links economic, environmental, and social development, is found in the Brundtland Report (World Commission on the Environment and Development, 1987):

"...development that meets the needs of the present without compromising the ability of future generations to meet their own needs."

A more narrow concept, and the one emphasized in this chapter, focuses on environmental wealth such as found in the two illustrative definitions below:

"...concern with optimal resource and environmental management over time...requires maximizing the net benefits of economic development, subject to maintaining the services and quality of natural resources."
"...that is, future generations must not inherit less environmental capital than the current generation inherited."

The precise definition of sustainable development will not be resolved here. It is the subject of ongoing discussion and economic valuation techniques beyond the scope of this effort. For example, what does sustainable development mean with respect to nonrenewable resources in which any use depletes the stock to be inherited? How is an orderly transition to alternative resources ensured if that is the appropriate approach in the case of such finite resources? How do we account for the changing composition of physical and natural assets over time? To what extent should it be assumed that rationing mechanisms, such as prices and public policy, will sufficiently conserve scarce or uniquely valued resources?

Finally, it should be recognized that two economic tools have considerable bearing on our ability to translate sustainable development concepts into decisionmaking criteria: (1) the common use of discount rates, which serves to devalue resources over time, and (2) recent efforts to incorporate measures of resource depletion into national income accounts.

At a minimum, three facets of sustainable development are pertinent for life-cycle impact methodology as it relates to resource depletion:

- An expanded time horizon, whenever feasible, to reflect concern for intergenerational equity.
- An expanded spatial dimension, wherever appropriate, to reflect concern for intragenerational equity.
- An enhanced appreciation for the linkages among economic, social, and environmental values.

5.1.2 Scope of Resource Depletion Assessment in LCAs

As with any other type of impact assessment, the scope and detail of a resource depletion assessment will hinge on the objectives of the user and the intended audience. The selection of time and spatial boundaries is particularly important for characterizing the nature and rate of depletion. Indices of depletion rates can be wholly misleading if timeframes are set inappropriately. Likewise, a resource may be relatively plentiful and sustainable in the country of manufacture, but may be near depletion in the country supplying the resource. Failure to consider these types of impacts can overlook considerations of both intra- and intergenerational equity.

In general, the time horizon used in analysis should reflect the best estimates available of the rate of renewal for flow resources in addition to any other time constraints imposed by the user. The spatial boundary should include, at a minimum, all regions involved in the various stages of the life cycle, including countries and regions supplying natural resources.
An optimal study will include global scales whenever feasible and will note gaps in data and other limitations.

5.1.3 Linkages with the Inventory and Improvement Component

The relationship between inventory, impact assessment, and improvement component is an iterative one with feedback and linkages throughout. With few if any exceptions, inventories conducted to date have not considered resource depletion as discussed in this chapter. Future inventories should remedy this gap to the maximum amount feasible particularly for scarce resources of special concern, e.g., tropical rainforests and scarce groundwater supply regions. Identifying which resources to include in the inventory and the extent of data collection should be part of the scoping and screening process. It is likely that data available for the optimal impact methodologies will be limited for quite some time.

Analysis of the various depletion indices and calculations below should provide insight into the nature and extent of any resource depletion that is occurring. This information, in turn, should lead to management improvements that will bring the rate of resource use to acceptable levels. Such improvements could affect the rate of use, activities to increase the resource supply, or efforts to restore the quality of the resource sufficient to continue or resume resource use.

As defined in Chapters 1 and 2, the stressor concept serves as the explicit mechanism for linking the inventory, where resource consumption is quantified, with the impact assessment, where the consequences of such consumptions are analyzed. In resource depletion analysis, stressors constitute the forces that either drive or accelerate the consequences of consumption. For many resources, the direct consequences, i.e., the depletion itself, will be associated with indirect effects such as biodiversity changes included in other impact categories.

5.2 DEFINITION OF DEPLETION

5.2.1 General Considerations

The concept of depletion refers to the idea that the reserves\(^1\), stock\(^2\), or flow\(^3\) of a resource are being diminished or impaired by human activity in such a way that the resource can no longer serve as input to the system under consideration. The direct impacts of the depletion of a resource are (1) the reduction in opportunity for future generations to access it, (2) the burden it may place on substitutes, and (3) the inability to pursue the activities upon which it depends.

\(^{1}\)Standing quantity, mass, volume, or flow of nonrenewable and renewable resource.
\(^{2}\)Standing mass or volume of a nonrenewable reserve.
\(^{3}\)Resource for which natural qualities allow replenishment.
There are also indirect depletion impacts associated with the use of a particular resource. For example, the question of depletion may be inconsequential for a sustainable forestry operation; however, the harvesting activity and related infrastructure requirements may have depleted the stock of a particular bird species by altering its habitat. Similarly, exploration activities may have depletion consequences on resources other than those under consideration. Although it may not be possible to measure all indirect impacts, their consideration should be determined based on the objectives and scope of the study. Pollution impacts as well as environmental disturbances related to raw materials acquisition activities are considered in Chapter 3.

The measurement of depletion is based on several temporal and spatial indicators. The depletion of stocks, the rate at which it is occurring, and the relative importance of the system's depletion impact on the current mass of resources are the main indicators. The consequences of using renewable reserves are also important in evaluating depletion impacts. The use of a reserve at a rate that exceeds its replenishment may have significant consequences on its substitutes for which there may be a finite quantity or which may impose on society large economic costs. For this reason, the question of depletion must be examined in light of the different characteristics of the natural resources being considered in this chapter.

### 5.2.2 Depletion and Type of Resources

Although there are definite natural characteristics that have traditionally served to judge whether a resource is replaceable or not, there are other factors that will influence the renewability or nonrenewability of a resource. For example, in some cases a resource is considered renewable if stocks are replenished at a rate equal to or above its rate of consumption. At first glance, wood appears to be a completely renewable resource; however, problems of soil erosion and degradation, regeneration rates, and management techniques can jeopardize the continued availability of the resource as an input to a specific production system. This uncertainty could also extend to water and fisheries resources where management practices and extraction rates influence the future availability and accessibility of the resource to society.

Similarly, the valuation of resources that are considered nonrenewable are also influenced by factors outside the natural quality of the resource. For example, the estimation of reserves of fossil fuel is intricately linked to economic mechanisms: market forces, costs of exploration, and resource rent. Although it is recognized that these resources are available in a finite quantity, the relative importance of the consumption rate depends on the estimation of reserves. In this case, the depletion impact of extraction would depend upon whether one is considering reserves under exploitation, other known reserves, accessible reserves, or speculated reserves.

Given these considerations, it is difficult to readily evaluate the consequences of resource depletion based solely on natural renewability characteristics. Factors influencing the availability of specific resources for current production systems, as well as for future
generations, must be considered in the evaluation. Therefore, in the context of an LCA, five evaluative criteria must be considered:

1. Reserve or stock of resource (local/global).
2. Rate of current consumption.
3. Intrinsic rate of natural replenishment.
5. Degree of substitutability.

These criteria will form the basis of the theoretical model for the evaluation of the relative impact of systems on stock and flow resources. Specific measurement considerations will be discussed in the next subsection as they relate to these criteria so that an assessment of the consequences of depletion can be made.

5.2.3 Categories of Resources

As explained, resource categories generally consist of resources that are intrinsically renewable (flow) or nonrenewable (stock). The actual renewability of a resource, however, hinges on such factors as rate of use and on economic factors that may affect both consumption rates and resource reserves.

Flow Resources:
- Air (including wind)
- Water
- Solar radiation
- Ocean currents
- Biotic resources (e.g., agricultural, forest, living species)

Stock Resources:
- Land (including the space dimension of land)
- Primary energy sources (e.g., natural gas, petroleum, coal)
- Minerals

5.3 RESOURCE DEPLETION CONCEPTUAL DEVELOPMENT

Consideration of resource depletion in the context of an LCA necessitates the establishment of a conceptual paradigm. As noted in Section 5.2, sustainability forms the basis for the ideal framework. This section describes how this concept could be embodied in the LCA process. First, issues related to the development of the concept are presented. Inasmuch as resource consumption and supply invariability exist within an economic system, some attention is devoted to linkages the analyst must be aware of when dealing with this area. Feedstock cost, manufacturing technology, and performance affect supply and demand and, hence, the potential depletion of resources (Lemons and Amey, 1991).
Second, a conceptual model for resource depletion analysis is presented. This model is intended to provide a benchmark for characterizing resource depletion implications in LCA relative to actual depletion measurements. These actual measurements range from those that currently could be utilized to those that contain a significant research component to be realizable in practice.

Although the terminology for dividing resources into two categories of stock and flow has been previously defined, it is essential to provide a more analytical specification. This analytical approach allows clear understanding of the similarity and differences between the categories, as well as making it possible to specify measurement units. In general, a resource depletion concept for stock resources reflects its exhaustion rate. Dimensional analysis suggests that the appropriate depletion measure should have the units of time:

\[
\frac{(M)}{(M/t)} = t
\]

where \( M \) and \( t \) refer to mass (reserves) and time, respectively. Thus, the variable \( t \) on the right side of the equation is a measure of years of supply left. In theory, the numerator reflecting the resource pool has units of time, but for all practical purposes the production rate of a stock resource is zero.

A resource depletion concept for flow resources reflects two attributes: (1) a pool consisting of the standing crop or reserves at time \( t \), and (2) a replenishment component reflecting the net of natural and managed replacement and consumption. Dimensional analysis here indicates a depletion measurement of the forms:

\[
\frac{(M)}{\left(\frac{M}{t}\right)} = t
\]

Both the signed magnitudes of \( t \) are important. The sign is indicative of accumulation (+) or depletion (-). The magnitude reflects the time to exhaustion for \( t<0 \) and the reserve doubling time for \( t>0 \). A flow resource whose short-term excess consumption is not overcome by management decisions will have a standing crop of zero in a finite period of time. Overfishing of certain species of food fish is an example of this type of depletion.

The generalized matrix of this to any type of resource is shown in Figure 5-1. Stock resources are indefinitely sustainable only if their consumption rates are reduced below the production rate. For all intents, this means all uses of the resource would have to cease. However, this brings up an interesting point as to whether the pool size should include only the "raw" stock, i.e. not yet acquired of a resource, or all stock, including the portion already in use. For example, should the resource pool for copper include only ore deposits or also copper plumbing pipes?
Practically speaking, stock resources should be evaluated from a perspective of either intergenerational equity or human welfare. Decisions made from the former impact perspective, through taxes, for example, extending the pool lifetime into the hundreds of years, would follow from a preservation ethic. Shorter-term exhaustion periods would be more pertinent to deal with from a quality-of-life perspective.

These general ideas create the need to know several things about a resource prior to evaluating it from an LCA perspective. The following subsection describes some of these factors.

### 5.3.1 Measurement Issues

Resource depletion issues are difficult to separate from one another. While the discussion tries to present these as individual items, it is important to remember that they are interactive in the way they affect the resource outcomes.

#### 5.3.1.1 Stock Resources

As defined above, stock resources are those whose properties preclude significant replacement on timescales relevant to society. Issues associated with

---

**Figure 5-1. Resource Depletion Matrix**

<table>
<thead>
<tr>
<th></th>
<th>STOCK</th>
<th>FLOW</th>
</tr>
</thead>
<tbody>
<tr>
<td>LOCAL</td>
<td>$S_{LS}/C_{LS} = U_{LS}$</td>
<td>$C_{LF}/R_{LF} = D_{LF}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td>If $D_{LF} &lt; 1$, then $C_{LF} - R_{LF} = E_{LF}$ and $Q_{LF}/E_{LF} = U_{LF}$</td>
</tr>
<tr>
<td>GLOBAL</td>
<td>$S_{GS}/C_{GS} = U_{GS}$</td>
<td>$C_{GF}/R_{GF} = D_{GF}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td>If $D_{GF} &lt; 1$, then $C_{GF} - R_{GF} = E_{GF}$ and $Q_{GF}/E_{GF} = U_{GF}$</td>
</tr>
</tbody>
</table>

- $S$ = Stock (amount)
- $C$ = Consumption rate (amount/time)
- $U$ = Use-Years (Time)
- $R$ = Replenishment rate (amount/time)
- $D$ = Depletion Index (dimensionless)
- $E$ = Excess consumption rate (amount/time)
- $Q$ = Standing quantity (amount)
- $L$ = Local
- $G$ = Global
- $F$ = Flow
stock resources include base consumption rates, economic factors, substitutability, and induced consumption.

**Base Consumption Rates.** To formulate any picture of the influences of a product system on a resource, it is necessary to have an accurate value for the current consumption rate. While this may seem obvious, it raises a number of critical issues. First, is there a clear definition of the resource of interest? In an LCI, the resource being analyzed may be ambiguous. For example, a resource impact assessment of glass bottles should not reference the global reserves of beach sand in the resource consumption quantity.

Second, what spatial and temporal periods should be used to determine the current consumption rates? Unlike the inventory where consumption is defined only for the product system and the fluctuations in demand associated with the product are considered, impact assessment must include an appropriate time horizon for overall consumption as well. Short-term fluctuations in consumption rates due to natural disasters, manmade events, economic conditions, and similar factors should not be factors in resource consumption calculations for an LCA.

For resources, the spatial scale of relevance for an LCA will be global. Although there may be practical or economic reasons why the actual supply of the resource to the product system is at a lesser scale, it is the societal impacts that typically are of interest. There may be LCA applications that are exceptions to this situation, however. A country may be specifically interested in the impact on depletion of its own reserves of materials. Care must be taken in drawing boundary conditions to properly include all of the necessary components. One could envision taking a resource depletion analysis to a local level where the issue could pertain to an individual mine or group of supply sources in a small area. The LCA impact assessment must clearly define the scale of the resource issue being evaluated so that it is transparent to the reader what the consumption rate and resource pool mean.

**Economic Factors.** Despite the desire that economic factors not enter into resource depletion impact assessment in environmental life-cycle studies, there are many intersections that must be recognized. On the reserves side, it is important to understand that not only do economic factors influence the fraction of the known reserves recoverable at any given price, they also affect the size of the known reserves themselves. This is because increasing prices allow suppliers to obtain additional knowledge about the state of the resource. Appropriate updating of reserve information is essential to attempt to track this aspect. Some statistical analysis of the tendency for reserve estimates to increase with the increasing price of a product might also be helpful.

On the consumption side, it is clear that in many situations material selection is based principally on the lowest installed system cost. Where stock resources are a significant input to the system, some fluctuations in consumption would affect general economic conditions. The appropriate averaging of trends data should be used to ensure that remaining lifetime estimates are accurate.
Substitutability. Related in some ways to the overall economic factors is the issue of substitutability. In the categories stock resources, the underlying premise is that only consumption rate changes can affect the remaining reserve horizon. Although many factors contribute to resource demand, one major issue influencing the accuracy and utility of resource depletion analysis is the potential for greatly reducing overall demand for some reserves through substitutions. A classic case is the exhaustion horizon for copper changing dramatically with the advent of fibre optic signal transmission. Legislation in areas removed from the product system can also affect remaining reserve consumption estimates.

Again, it will be important for the LCA analyst to understand shifting material uses in the economy and how they might influence the particular resource under investigation. Various entities such as government statistical groups or resource bureaus track these types of trends closely and should be consulted for guidance and, in some instances, data.

Induced Consumption. The purpose of life-cycle resource impact assessment is to assess the consequences of a product system on environmental resources. However, this necessitates an understanding that the state of depletion (sustainability) must be evaluated both before and after the recommendations included in the LCA are implemented.

In a comparative LCA, whether internal to a company or externally through policymaking, the sustainability status of the resources could change depending on the extent to which the "better" choice is implemented. Thus, in addition to the base consumption assessment issue discussed above, the additional induced consumption should be compared to the resources. For an internal study directed at a global analysis the incremental consumption may not be significant. However, the impacts on a regional or national resource pool from a national decision may be highly significant. In addition to the overall magnitude of the induced consumption component, its timing should be investigated. While this is not as great an issue for stock resources as for those of the flowing category, longer implementation timeframes may be less impacting. Since this situation has some correlation to economic demand elasticity analysis, some of the same tools may be useful in creating and analyzing various scenarios.

5.3.1.2 Flow Resources. Most of the measurement issues discussed above for stock resources are applicable to flow resources as well. In particular, mention is made about the time factor associated with induced consumption. Since flow resources have a replenishment time component, it is reasonable that the more slowly a change is induced, the more quickly the production system will be able to accommodate it.

For example, if an LCA were to be conducted comparing glass beverage containers to paper-foil laminate cartons, the impact assessment could show that the paper-foil laminate carton was preferred. This may or may not be true for reasons that have little to do with potential resource depletion impacts. In either case, it is incumbent upon the analyst to assess the depletion impact after the implementation of the recommendation. It would be counterproductive to shift demand patterns in a manner that could change a sustainable resource into a depleting situation. Clearly, implementation rate assumptions
will affect the degree of impact. For example, slower implementation through regular turnover and market introductions would be better accommodated by the resource system than a sudden tax or a ban.

One area related to flow resources alone is the issue of intrinsic renewability rates. As noted above, flow resources have the capability to be managed in ways to increase the rate of production as well as decreasing rates of consumption. Increasing production can come about through either increasing the area devoted to production or, for biotic resources, by increasing the growth per unit area. Any biotic resource should be characterized as to its growth rate and compared to maximum rates limited by the organism. Several sustainable yield models are available in forestry and fishery management to facilitate this analysis.

5.3.2 Resource Depletion Analysis Matrix

The measurement of resource depletion impacts can be conceptually portrayed in the form of a matrix. The dimensions of the axes of the matrix are resource type (stock or flow) and impact assessment spatial scale (global or local). The matrix shown in Figure 5-1 shows the depletion measures for the stock resource category. Both of these measures are defined as:

\[
\begin{align*}
\text{quantity of reserves (M)} \\
\text{demand rate (M/t)}
\end{align*}
\]

An impact assessment would involve a comparison between the remaining use-years with and without the product system. For the flow resources, the situation is somewhat more complicated in that replenishment must be considered. Figure 5-1 includes the depletion measures for this resource category. This category consists of a two-step procedure. First, an overall depletion index is computed as:

\[
\begin{align*}
\text{rate of consumption (M/t)} \\
\text{rate of replenishment (M/t)}
\end{align*}
\]

If the value of the index is computed to be less than 1, then the resource is currently sustainable. Recall that this calculation must be done for consumption both with and without the product system. If the index is greater than 1, then the resource is not or may not be sustainable. Therefore, a second calculation, similar in concept to the remaining use-years, must be performed. This involves subtracting the rate of consumption from the rate of replenishment to yield a rate of excess consumption (M/t). This value can be divided into the current reserves to obtain a remaining life analogous to that for stock resources.

5.4 DEPLETION MEASUREMENTS

This section considers the applicability of the general framework to different categories of resources and the corresponding implications regarding data requirements.
The concern here is the measurement of depletion and not that of the secondary consequences of depletion. The latter, whether they be economical, social, or ecological, may be assessed in a variety of ways, either directly from the inventory data or on the basis of an estimation of the depletion phenomena of the resources that are used in a product system. It is believed that the latter approach, in which an objective estimation of the contribution of a product system to the depletion of various resources serves as a basis for subsequent analysis or decision processes, is preferable. Besides, a depletion measurement is already usable for interpretation and decisionmaking.

Stock resources (intrinsically nonrenewable resources) and flow resources are examined separately. A measurement of resource use that would unify stock and flow resources is not to be excluded, but was not developed. When the general framework is not directly applicable for practical reasons, alternative measurements are discussed.

By no means does this section pretend to be exhaustive or even sufficient. General issues concerning measurement, some of which are examined, could fall under the following headings:

- Spatial and temporal scope.
- Dynamics of the reserves of resources; consumption rates; updating and distribution of this information; basic to impact assessment.
- Uncertainty and data gaps.
- Existing data, tools, and techniques.
- Aggregation of measurement for resources belonging to same category (intra-category aggregation) and for resources belonging to different categories (inter-category aggregation).
- Continuous measures versus discrete scales.
- Economic (or other) subsequent valuation.

LCA studies have paid little and not very consistent attention to resource consumption until recently, both at the inventory level and at the (tentative) impact assessment or interpretation level. Most often, resources linked to material usage are not, or incompletely, listed in the inventories; fossil energy carriers are sometimes detailed in mass units or are already aggregated and translated in primary energy units.

Note that in the first instance, energy consumption is not a stressor (or impact factor) per se and therefore is not an adequate basis for impact assessment. Energy consumption corresponds to the use of different material resources, the scarcity of which may differ, and translates into emissions and waste production. Inputs and outputs linked to energy
production being the ultimate stressors, an impact assessment should be based preferably on these items. Energy use is still important information in an inventory when no information on raw materials is available. Also, in every instance, energy use is an indicator of the overall energy needs in the product system useful for the identification of improvement opportunities.

5.4.1 Depletion Measurements for Stock Resources

The general framework for measuring the depletion of stock resources applies to intrinsically nonrenewable resources, such as minerals and land. It could also apply to some intrinsically renewable resources, in which case an irreversible depletion process is involved.

5.4.1.1 Minerals. Within the general framework, there is no reason why resources should be differentiated according to their use. In particular, the depletion measurement for minerals should be the same, whether the resource is used as a material (e.g., bauxite, iron, oil, nonrenewable water underground supplies), or as an energy carrier (e.g., coal, natural gas, oil, uranium).

Measurement Approaches. The ratio of global annual consumption over global reserves is an indicator of the depletion of a given mineral (number of years left before complete depletion). Global annual consumption estimates may be relatively precise and easily collectable. Values could be averaged over several years to reduce short-term fluctuations associated with transient events.

Global reserves are much more difficult to estimate. First, one could either use total reserve estimates or extractable reserve estimates (judged as such on technical or economical grounds). Resulting depletion measurement could vary considerably, according to the option chosen. Furthermore, estimates of total reserves vary with discoveries of new finds; estimates of extractable resources vary with technical innovations; and both types of estimates vary according to geostategical and economical matters (e.g., the price of the resource itself). A frequent update procedure would then be necessary. For the sake of coherence, the information should be centralized and distributed to all practitioners. Second, it may be difficult to obtain estimates in the first place because of the economic and strategic implications of such information, both at a corporate and at a national level.

Depletion rates should be calculated at a global level and not at a continental, regional, or national level. This could be achieved for most mineral resources. Nonetheless, it may make sense to use national, regional, or continental data and to extrapolate on this basis to a global scale whenever global data are missing or when their reliability is too poor. In some instances, depending on the scope of the study, it is conceivable that regional depletion rates could be considered.

Although depletion tends to be considered a continuous measure, discrete scales could be devised, if necessary. Minerals could be classified in several categories (for
instance, rare corresponding to supplies for less than 100 years; abundant corresponding to supplies for more than 1,000 years).

Finally, in instances where the sustainability of use of flow resources is not measured and where no depletion rates are estimated for stock resources, it would make sense, in order to obtain a crude interpretation of the results, to have detailed inventory data for the stock resources used.

**Measurement of Product System Contributions to Resource Depletion.** In Section 5.3 it was stated that the depletion measurement (years of remaining supply) could be used as a basis to assess the contribution of a unit of product to the depletion of a given mineral resource and to build an overall aggregated index for depletion of all stock resources used in a product system. An exploratory discussion of one approach to developing such an index is proposed below. In the following paragraphs:

- $C_i$ stands for annual global consumption for the resource $i$.
- $S_j$ stands for global stock of resource $i$ (reserves estimate).
- $U_j$ stands for remaining use-years for resource $i$, based on current annual consumption and current reserves estimate. Note that $U_j = S_j/C_i$.
- $n_{ij}$ stands for the amount of resource $i$ used in the product system under study.
- $I_i$ stands for the index measuring depletion for resource $i$.
- $I$ stands for the aggregated index for measuring depletion of all stock resources in the product system.

When considering the problem of aggregating depletion measurements for several different stock resources, one can draw the conclusion that the index should be a function decreasing with the remaining use-years of a given resource (the longer the reserve will last, the less important is the depletion problem linked to the consumption of the resource in the product system). One of the simplest functions for $I_i$ then could be:

$$I_i = m_i/U_i$$

This kind of function is not satisfactory. This can be seen through dimensional analysis, and is illustrated with the following example. The index is the same for resources A and B for which use-years and absolute consumption are the same in the product system, whereas the relative contribution of the product system to the depletion of the resource is higher for resource A.
It appears that it is necessary that the index be a function of a second variable, \( \frac{m_i}{S_i} \) (the larger the contribution of the product system to the depletion of the reserve, the larger the depletion index). One simple function of that sole variable would be:

\[
I_i = \frac{m_i}{S_i}
\]

Again, a function of this single variable would not be satisfactory, as illustrated by the following example. The index is the same for resources A and B (relative contributions of the product system to the depletion of resources A and B being the same), whereas the use-years left are not the same.

The following type of function of two variables, increasing with \( \frac{m_i}{S_i} \), and decreasing with \( U_i \), seems acceptable:

\[
I_i = I \left( \frac{m_i}{S_i}, U_i \right)
\]

The index is a function of both the global depletion problem (the larger the problem, the larger the index), and the relative contribution of the product system to the depletion (the larger the relative contribution, the larger the index).

One such simple function is:

\[
I_i = \frac{m_i}{U_i S_i} = \frac{m_i}{C_i S_i^2}
\]
The values of this index for the previous two examples are shown below.

<table>
<thead>
<tr>
<th>Resource</th>
<th>( m_i )</th>
<th>( C_i )</th>
<th>( S_i )</th>
<th>( m_i/S_i )</th>
<th>( U_i )</th>
<th>( I_i )</th>
</tr>
</thead>
<tbody>
<tr>
<td>A</td>
<td>100</td>
<td>100</td>
<td>1,000</td>
<td>10%</td>
<td>10</td>
<td>10</td>
</tr>
<tr>
<td>B</td>
<td>100</td>
<td>1,000</td>
<td>10,000</td>
<td>1%</td>
<td>10</td>
<td>1</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Resource</th>
<th>( m_i )</th>
<th>( C_i )</th>
<th>( S_i )</th>
<th>( m_i/S_i )</th>
<th>( U_i )</th>
<th>( I_i )</th>
</tr>
</thead>
<tbody>
<tr>
<td>A</td>
<td>100</td>
<td>100</td>
<td>1,000</td>
<td>10%</td>
<td>10</td>
<td>10</td>
</tr>
<tr>
<td>B</td>
<td>1,000</td>
<td>10,000</td>
<td>10,000</td>
<td>10%</td>
<td>1</td>
<td>100</td>
</tr>
</tbody>
</table>

Note that the choice of the function is not neutral and implies some political judgment. For instance, the following function:

\[ I_i = m_i \cdot C_i^2 / S_i^3 \]

with respect to the previous function would penalize resources with low reserves relative to resources with high reserves.

It should be noted that these functions suppose that the annual consumption is a constant. Different hypotheses regarding the evolution of consumption with the size of the remaining stock would yield different functions.

5.4.1.2 Land. It may be difficult to value different alternative land usages one against each other from a global point of view and independently from site-specific consideration. In addition, it is unlikely that the overall land use for a product system could be quantified. Nonetheless, in some specific instances in which a clear depletion problem exists for a specific usage, land use estimates are relevant. This is clearly the case with landfill space in every industrialized country. A measurement of depletion landfill space therefore makes sense, and a volume unit would seem suitable.

The resource depletion conceptual framework seems difficult to apply in this instance. Reserve estimates do not depend as much on physical grounds (geological properties of potential sites) as on social acceptance of new sites. Either such socially accepted reserves should be estimated or the estimate should be based on current available space in existing landfill sites. Eventually, however, such socially accepted reserves should be estimated.
When no further aggregation with other resources is considered, an estimation of depletion is not necessary, and space consumption alone could be sufficient to assess the contribution of a product system to landfill depletion. The analyst should consider the fraction of the total solid waste generated by the product system, whether process waste or post-consumer waste enters landfills versus other solid waste management systems. In the case of incineration, estimation of ash residue is needed if this residue eventually will be sent to a landfill.

The consumption of landfill space could be evaluated through the direct use of mass units of waste. Volume units would be more appropriate, but require an estimation of the density of various materials in landfill conditions. Another dimension that could be introduced is the time it takes for material to reach its final degraded or compacted state. Since the acuity of the depletion of storage capacities varies with the type of waste, it would be useful to distinguish among the main categories of waste (inert, degradable (garbage), infectious, toxic, low-level radioactive, and high-level radioactive).

5.4.2 Depletion Measurements for Flow Resources

5.4.2.1 Biotic Materials. This category covers a diverse group, including biomass resources such as trees used to manufacture products; wood used as fuel; agricultural crops of all kinds (including aquaculture); plants and animals harvested from open water (oceans, rivers, streams, lakes, etc.); solid waste from industrial or other human activity; or any other resource drawn from living organisms that is renewable within a social timeframe.

The depletion of this type of resource depends on its consumption rate as compared to the rate of replenishment. For an agricultural crop, this is conceptually easy to assess. For an annual crop the rates are customarily in balance because only the approximate amount required by consumers is grown each year, which leads to a depletion ratio of 1 (rate of replenishment divided by rate of consumption). This is true locally as well as globally.

For longer-term crops or for primeval (undeveloped) resources, this is not true. For example, trees may take 40 to 100 years to grow and the production cannot be rapidly adjusted to accommodate demand. Depletion of this resource occurs when the rate of consumption exceeds the growth (depletion ratio is less than 1). Appropriate measures might be mass of wood or volume of wood. Both replenishment (rate of growth) and consumption must be in the same units, but the selection of units is a matter of convenience. Annual growth and consumption is commonly documented and is an appropriate time scale.

The spatial consideration raises important issues. Wood resources are desired to be sustainable globally, so a global calculation is relevant. However, global data are highly suspect and quite inaccurate. National data are more accurate in more developed nations, but may be less reliable in countries with poorly developed information systems. In some cases, local data may be of interest, such as for a single state or provinces, or even for a single forest. If global data are not used, the selection of the spatial scale (boundaries)
should be clearly described, along with a justification of the selection. Depending on the application, it may be necessary to subdivide the consumption and growth by use. For example, if an analysis of building lumber was undertaken, the consumption and growth of trees for pulp is not relevant.

The assessment of data quality becomes quite important. The assessment of replenishment should be based on actual measurements of tree growth if possible. Secondary estimates are less reliable. The most accurate measures may be for smallest spatial scales. For many countries, politics may play a strong role in statistics. In other places, private interests may be factors that contribute to skewed data.

If the depletion ratio is greater than 1, then the desired result exists. If it is less than 1, another calculation is needed (i.e., the standing supply depletion ratio). This is the ratio of the standing stock or potential supply to the excess consumption. For example, if you have 100 trees, are growing 100 trees per year, and are using 101 trees per year, you have an excess consumption of 1 tree per year. The standing supply depletion ratio is 100 trees divided by 1 tree per year, or 100 years are left for the standing stocks at this depletion rate.

This ratio is similar to the depletion ratio calculated for stock resources, such as fossil minerals, but differs because it is potentially correctable. In some cases the replenishment rate for flow resources can be altered by management policies or more reserves may be brought into the system through economic or political action, which is not the case for stock resources.

Another issue is primeval resources. These include undeveloped forests, species of fish for which demand suddenly increases, and so on. In theory, these can be analyzed in the same way as for agricultural crops. The same measures are required and the same ratio can be calculated. However, the aesthetic, economic, environmental, and political issues may be very different. Data quality may be lower because primeval resources may not have a history of economic development that is frequently associated with more careful measurements. Otherwise, the technical issues are the same.

Solid waste viewed as a natural resource brings about a new level of analytical complexity. Solid waste may be used as a fuel in which it substitutes for other fuels, such as coal, oil, and natural gas. It also serves as a potential source of raw materials for products made from glass, aluminum, plastics, and others. Solid waste has both flow and stock resource characteristics. For material, a landfill can be viewed as a mine where many materials can be salvaged even after many years. However, at this time it is not economically viable to process landfills for material recovery. This may change in the near future. When that does occur, the recoverable materials should be considered as part of the global reserves of stock resources.

The combustible materials can be considered a part of our global energy resources. However, only a small fraction of the world's solid waste is economically recoverable as an energy resource. These resources are clearly flow resources, but are comparable more to
the use of fallen trees in a forest. Fallen trees are not cut for economic use, but are used because they are available. In the same sense, combustible products are not discarded to waste so that they can be used as an energy resource. Rather, because they occur in sufficient concentration, they become a resource.

There is no general consensus on how to evaluate solid waste as a resource. One possible way is to simply consider all products manufactured that are potentially combustible and that may be discarded to waste as a solid waste energy resource. Another option is to consider solid waste as an energy resource only to the extent that it is economically recoverable. Other possible methodologies may exist. Whatever methodology is selected by a consensus of the scientific community, the general outline of the analysis of natural resource depletion should be followed.

5.4.2.1 Water. Water can serve as either a flow or a stock resource, depending upon its use. An underground water reservoir that is not recharged within a human lifetime is a stock resource. If water is used as a source of hydroelectric power, or if it is used for drinking water or industrial processes and is withdrawn from rechargeable resources, it is a flow resource.

As with other flow resources, water depletion can theoretically be characterized by the depletion ratio. If the depletion ratio is less than 1, a standing supply depletion ratio needs to be calculated to quantify the years of supply left at current rates of depletion. The relevant temporal unit is the annual volume of water used (gallons or cubic meters) and total volume available. In the case of hydropower, this is difficult to measure. While the volume of water flowing through hydroelectric generators in the world can be measured in theory, in practice these numbers will be accurate only for developed countries. The existence of thousands of small hydroelectric turbines makes even these total numbers suspect. Many of these turbines are not included in existing statistics. In addition, just as with annual agricultural crops, hydroelectric installations generally do not exist unless the replenishment rate equals or exceeds the consumption. In fact, more hydroelectric installations share water with other uses, such as flood control, recreation, and controlled ecosystems. Water may be partitioned between uses based on politics and economics. The replenishment rate can be considered adequate for most existing hydroelectric sites.

An important question concerns the total supply of water potentially available for hydropower. Although it is clear that many more hydropower sites exist that are currently exploited, the volume of water associated with this potential for expansion is not known to any reasonable degree of certainty. As with stock resources, this quantity depends very strongly on economics and has limited usefulness in assessing this resource.

Power and drinking water are important renewable resources for which the measurement issues are more clear. The volume of water consumed is the amount of water removed from natural water cycles. An example is water pumped from wells that is used for drinking or industrial purposes and is then discharged to the surface. The underground water reservoir is disrupted by this action and the depletion ratio needs to be calculated.
Another example would be the diversion of water from a river for crop irrigation or municipal purposes, decreasing the flow for users downstream. This ratio is difficult to calculate. Data for water consumption by source are poorly documented and frequently absent at any spatial scale. Local data are more likely available, but global data on this subject are likely quite inaccurate.

Great care must be taken to obtain the appropriate volume. Water that is simply taken from surface water and returned to surface water does not represent any alteration of the resource as it does not interrupt natural cycles. Water taken from underground sources and discharged to the surface may become a new supply for others downstream, complicating the issue. However, water that is actually consumed chemically by a process, evaporated, or included in products should be measured.

The primary problem with water data is that customarily only water intake volumes are known, but sources and points of discharge are not. For resource analysis purposes, it is critical to know the water origins so that, for example, once-through cooling water is excluded.

Another important issue is that water generally is not readily transportable worldwide. For economic reasons, water is used quite near its source. This results in a spacial confinement of analysis. Water issues are generally local or regional. For example, in the United States wet southern coastal regions have very high aquifer recharging rates so that water as a resource is generally not an issue. On the other hand, western regions have little underground water and recharging of surface water impoundments varies greatly from year to year. Water may be the most important resource issue for those areas. Global analysis of water as a resource has limited value and application. When collecting data for LCI studies, it is important to specify precisely the water volumes required.

5.4.2.3 Geothermal. These resources occur as water is heated by volcanic intrusions close to the earth’s surface. In some cases, naturally occurring heated water is used directly. In other cases, water is pumped into hot underground rocks, and the heated water recovered. In some cases there has been a degradation of temperature after years of use. However, a geothermal resource is generally considered renewable when external water is heated. When natural hot water is removed, depletion occurs.

A quantitative assessment of this resource is quite difficult. This is not a well-developed resource, and statistics are erratic and frequently not comparable. Estimates of production are available, but quantitative descriptions of reserves are highly variable and strongly dependent on assumed economics. The concepts of replenishment and consumption are not relevant. An economic methodology can be used to gain some perspective of this resource by considering the global supply as the total economically recoverable heat from known geothermal sites. At this point in time perhaps, geothermal sources should be considered as having depletion ratios of 1; that is, they are not being depleted. However, it should be fully realized that as with other flow resources, geothermal resources have a limited capability. Any significant shift to these sources is not possible, and in fact the
Resource Depletion

economics of these sources at the present time is generally marginal. The low thermal conductivity of hot rocks makes the economically viable recovery of heat quite difficult. For purposes of interfacing with LCA data, geothermal power is similar to hydropower and should be considered in a comparable manner.

5.4.2.4 Solar and Wind Energy. These two energy sources are, for practical purposes, considered to be renewable. The constraints relate to suitable sites and economics, not with replenishing the basic energy source, as is the case with biotic materials. The depletion ratios are always 1 or greater, and depletion is not an issue.

5.4.3 Information Needs and Acquisition

5.4.3.1 Inventory Data. To analyze the impacts linked to resource consumption, the inventory should include the following items:

- Number of units of the product made during the typical time period for the resource depletion, e.g., annually.
- A detailed list of primary energy sources and their individual contributions to the product system.
- A characterization of the origin and use of biomass resources, such as trees used to manufacture products, wood used as fuel, agricultural crops of all kinds, and plants and animals harvested from open water.
- A specification of solid waste outputs according to destination type (category of landfill or of incineration facility).
- If local depletion of resources is to be assessed, an analysis of the consequences of the extraction activity on other local resources.

5.4.3.2 Impact Assessment Data

- Depletion of minerals/energy sources: years of supply versus consumption and reserves.
- Land: estimates of volume occupied by various materials/products, including estimates of rates of degradation or compaction.
- Water: years of supply versus consumption and reserves for water in underground aquifers being mined, i.e., withdrawal exceeds recharge.
5.5 PRESENTATION OF RESULTS

5.5.1 Interpretation

The technique of impact assessment within the context of an LCA is still in its infancy. The ideas and measurement tools discussed in the preceding sections often combine both quantitative and qualitative analyses. Merely using proven scientific tools would prove very restrictive and would limit what could be studied. This chapter attempts to define the ideal and to develop measures to achieve it. By demonstrating the utility of doing this type of analysis, despite the drawbacks of some of the measurements, it is hoped that with time both tools and data sources will evolve. However, as a result of these concerns, there are limits on the utility of this analysis. It can provide some indication or characterization of the impact but will not provide a mathematically exact answer. Conclusions will be drawn from a combination of data — each of which has a differing degree of uncertainty. Every effort should be made to identify the information used and the source and degree of accuracy, as well as any assumptions made. It is also important to be aware that different groups will look to the study to provide different information. Although it is important to consider all potential users in presenting the study, the aim must be to provide a broad picture that includes all considerations. To consider one aspect of the work in isolation, taking it out of context of the whole impact assessment, would devalue the work and render the conclusions meaningless.

When interpreting the results of such a study, it is important to consider the effect of the recommendations being made on the balance of resources. For example, a recommendation to manufacture a product from a flow resource-material may have a dramatic effect upon the stock of that product, particularly if the change has to be effected quickly and will create a depletion effect. Data may be available to measure the impact of such a recommendation on a local scale, but reliable data are unlikely to be available on a global scale and, if available, will be difficult to obtain and of questionable reliability.

5.5.2 Renewable Versus Nonrenewable

Current LCA techniques often address a distinction between renewable versus nonrenewable resources; however, it should be emphasized that this is merely one element of the decision. For sustainable development, the goal would be to maximize the use of those resources defined by the above analysis as renewable. Resource depletion is only one of the impacts considered in an LCA. The production of a product or system might generate a large amount of chemical emissions with detrimental human health implications or impose a large burden on the ecological system. The relative weights of these different impacts must be considered in making a recommendation and is dealt with in another part of this report. In addition, there are other influences that need to be taken into account outside the boundaries of an LCA. A recommendation to shift the manufacture of a product from a nonrenewable to a renewable resource may well affect the demand side of the equation, which in turn could create a set of new problems.
5.6 RESEARCH NEEDS

1. Develop a methodology for analytically deriving replenishment rates for flow resources that is consistent with the complexity of social, ecological, and economic concerns.

2. Develop a methodology for categorizing the relative seriousness of depletion rates for both stock and flow resources.

3. Develop a methodology for determining the extent of stock reserves, including the issue of current products in use and the role of materials in landfills.

4. Develop a methodology for evaluating and incorporating the initial impacts of a flow and stock resource during periods of initial development and closing down developments.

5. Define a way to evaluate the resource implications of hydropower.

6. Develop a methodology for evaluating the energy and material resource potential of solid waste.

7. Develop and refine economic theory related to sustainable development that can be translated into useful decisionmaking criteria.
CHAPTER 6.0

VALUATION

6.1 INTRODUCTION

LCA is an analytic tool to systematically identify and elucidate the multiple attributes that should be considered in assessing or comparing products and processes on environmental grounds. To realize this potential, however, an LCA must wrestle with and resolve a fundamental analytic dilemma: As the number of distinct attributes (i.e., impact categories) increases — a desirable outcome from the perspective of accurately describing the environmental features of the system(s) being assessed — the analytic complexity increases and, above some level, the utility of the information to the decisionmaker decreases.

Whatever the complexity, decisions still need to be and will be made. Once technical data have been assigned to impact categories (the classification phase) and impact assessment tools have been applied to the data to technically describe and assign measures to such impacts (the characteristic phase), a process of **valuation** takes place. In this phase, whether implicitly or explicitly, individually or collectively, values or relative weights are assigned by the decisionmaker to the various categories of impact and their associated descriptors. These values are then compared or aggregated in some manner to reach a decision or ultimate perspective on the multi-attribute problem at hand.

The challenge in valuation is this: to adequately capture and express in sufficient detail the full range of impacts that need to be considered by the decisionmaker without overwhelming that individual or individuals with either the sheer amount of information, the diversity of information, or the lack of comparability of information. In short, while there is likely no alternative to having to compare apples, oranges, kiwi fruit, and watermelons, can we find a recipe for a palatable fruit salad?

It should be recognized at the outset that it will be impossible (and inappropriate) to develop an entirely objective method for impact assessment valuation. While quantitative methods may be available for assessing impacts within a given impact category and even for specifying a process for integrating across categories, the assignment of relative values to the categories is inherently subjective and value laden, and not a task for scientists alone. In this chapter, several conceptual and methodological approaches to valuation are discussed that utilize and apply different methods for valuation. The focus is on identifying approaches that have been used, or might be adapted for use, to explicitly and collectively valuate the information gathered and assessed in an LCA.
6.2 VALUATION DEFINED AND SITUATED IN A LIFE-CYCLE IMPACT ASSESSMENT

The process of valuation has been delineated as the third discrete element or phase in the conceptual framework for a life-cycle impact assessment (see Figure 2-3). It starts with the results of the characteristic phase as applied to each of the impact categories identified through the first phase of classification. Assessed impacts are judged beneficial, neutral, or adverse, or as a net balance of all three. Where those values come from and how weights are assigned to them are the subjects of this chapter.

In addition to its role as a discrete element in life-cycle impact assessment, however, it is important to recognize that to varying degrees valuation occurs at multiple points throughout the entire LCA process. It is desirable to acknowledge and make explicit the role of valuation at each of these points:

- **Goal Definition and Scoping:** Valuation takes place at the outset of a study, when goals and objectives of the assessment are established.

- **Problem Identification and Establishment of System Boundaries:** Valuation is central to the manner in which the problem is viewed and the extent of the system(s) to be included in the analysis.

- **Formulation and Analysis of Alternatives:** Valuation underlies the formulation and evaluation of alternatives to be considered, alternatives that invariably embody different sets of weighting values.

- **Ranking or Weighting of Impact Categories:** This is the formal step of valuation in the conceptual framework for life-cycle impact assessment.

- **Ranking or Weighting of Alternatives:** In comparative LCAs, the process of deciding among the options included in the analysis is clearly value laden.

At each of these steps, valuation is also involved in the process of deciding whose values are to be included or considered. In any situation to which LCA might be applied, there is almost certain to be a differential distribution of impacts; that is, the incidence of impacts (beneficial or adverse) falls unevenly and unequally across various segments and sectors of society and the environment. The question of whose values are utilized or given greater or lesser weight is obviously a critical one to be addressed in valuation.

Both experts and various publics can and should participate in valuation. Expert values are typically applied as "professional judgments" and largely reflect the value system of science. Social values are identified through the identification of interested and potentially affected publics. Several types of publics can be identified; for example, one public may be considered as those persons who are most attentive to a given problem. Another public may be those persons who are inattentive, apathetic, or alienated. Publics
can also include "inarticulate" groups, for example children or future generations, indicating a temporal scale that should be considered in the identification of "relevant" values.

Publics can span a spatial hierarchy from global through national (or societal) and regional to local. In many cases, a useful distinction may be made between public and community interests and groups, with the locality of the latter being a defining characteristic. Even within a highly localized setting, different interests and groups may be discerned. For example, interests within a single ecosystem such as a watershed may well be divided between upstream and downstream residents.

Typically, multiple publics are involved in any given valuation process. For that reason, unitary or unidimensional definitions of "the public interest" may be insufficient. Various social judgment techniques have been developed to take account of the diversity of potentially affected publics; a few illustrative techniques will be briefly discussed later in this chapter.

6.3 APPROACHES TO VALUATION

How do we convince ourselves that the decisions we make are the best ones? To make a successful decision we need to consider not just all the essential information and hard data, but also the goals and criteria that we believe should have a bearing on the decision. It may well be that certain kinds of data that appear most urgent scientifically would not impact our goals as much as other less precisely quantifiable information. The best decisions often do not depend on great precision of measurement because the measurements must eventually be interpreted in terms of our not very precisely understood goals. Thus, how we structure and apply our judgment to make a decision is as essential, if not more so, than having a great deal of data about the problem but no effective way to trade off the different kinds of information. There may be a few alternatives to choose from, none of which is so attractive as to make it a best choice, even if we gathered additional information.

A variety of techniques have been developed to assist in the making of complex decisions (those with multiple objectives and attributes). Methods include the Decision Analysis Using Multi-Attribute Utility Theory (MAUT), Analytic Hierarchy Process (AHP), and Impact Analysis Matrix (IAM) Approach, as well as other methods. Each of these methods varies somewhat in approach and complexity. Although detailed descriptions of each approach are beyond the scope of this chapter, the following discussions on the MAUT, AHP, and IAM methods are illustrative of this class of analysis. These discussions are limited to valuation of environmental attributes, although obviously those same methods could be used to assign values to other factors (e.g., costs, perception).
6.3.1 Decision Analysis Using Multi-Attribute Utility Theory (MAUT)

Decision theory, or decision analysis, provides a logical structure for the analysis of complex decisions such as those addressed in a product LCA. For LCA problems with multiple issues or objectives, the problem is first decomposed into single objectives and attributes. The attributes are then used to measure the degree to which an objective is achieved by a management option; attributes should be meaningful to the issue, measurable, predictable, comprehensive, and nonoverlapping. The identification of objectives and attributes leads to consensus concerning the nature of the LCA. Subsequent analysis should focus on estimating the effects of various management actions on the levels of the attributes.

A decision analysis conducted for the evaluation of packaging options, for example, should have the following components for the evaluation of environmental impacts. First, the system components and boundary should be defined. Second, an inventory and impact assessment is needed that describes the system from sources to effects. In specific terms, this means that the impact assessment should relate human and ecosystem effects of residuals (environmental contaminants) emitted throughout the product life cycle to specific actions from raw material acquisition to waste management. Effects are essential, since it is the human and ecosystem effects that impart utility (or disutility) to various packaging options. In addition, uncertainty analysis should accompany the solution of the model so that the effects are expressed probabilistically. This will result in one component of the decision analysis, the probability model predicting the states of nature (which reflect ecosystem effects and human health effects for this problem).

The LCI and impact assessment for all important attributes yields essential information on the effects of various management actions; however, that alone is insufficient to make a decision. A second essential component of a decision analysis is a measure of value, or utility, associated with each outcome. This may involve estimation of net benefits (costs minus benefits), particularly if market data are available to quantify terms. However, if externalities are involved and value functions are nonlinear, utility analysis may be appropriate.

The overall utility function is invariably multi-attribute, involving issues of consequence in more than one sector. For example, an environmental decision can involve energy use, water pollution, solid waste, and air pollution (Figure 6-1). To understand and properly address these complex issues, all meaningful attributes must first be identified. It makes little sense (and may result in unpopular and inefficient decisions) to conduct a multi-attribute analysis based strictly on its impact on solid waste volume or on air quality regulations (unless all other attributes are assigned a zero weight, which reduces the issue to a single attribute problem).

Once the attributes are identified, the analyst can estimate the multi-attribute utility function. Under certain, often reasonable, conditions the multi-attribute utility function may be decomposed for ease of analysis. Keeney and Raiffa (1976) identify these conditions as preference independence and utility independence; they involve tests of the sensitivity of the
**Figure 6-1** An Objectives Hierarchy for a Generic Life-Cycle Assessment

**Objective:** Minimize Overall Impact

- Minimize Energy Use
  - Overall Energy Consumed in:
    - Resource extraction
    - Manufacture
    - Transportation/distribution
    - Recycle
    - Ultimate disposal

- Minimize Solid Waste
  - Volume of solid waste
  - Toxicity of waste

- Minimize Air Pollutant Effects
  - Toxicity of waste
  - Effect of emitted air pollutants

- Minimize Water Pollutant Effects
  - Effect of emitted water pollutants
decisionmaker's preferences for attributes to changes in other attributes. These conditions
generally hold in most decision problems, permitting decomposition of the utility function.
Therefore, it is often a reasonable approximation to break the multi-attribute function into
single attribute utility functions. After the individual utility functions are estimated in the
manner briefly described in Appendix C, they are combined in a multiplicative or additive
manner according to the values of estimated scaling coefficients. The entire procedure is
presented in detail in Keeney and Raiffa (1976).

There are a number of features of multi-attribute utility analysis that are difficult to
implement. This has led to simplifications and to other methods, such as that briefly
presented in Subsection 6.3.2. As a simplification, Edwards and Newman (1982) work with
additive utility functions only. To combine the functions, they recommend that all attributes
should first be ranked and then scaled (in importance) in relation to the least important
attribute. The scaling constants are then used to weight each single attribute utility function
in the additive multi-attribute utility function. Objectives of greatest importance receive the
largest weights; those of least importance receive the smallest weights. For example, weights
might be assigned on a 1-10 scale. If "Minimize Energy Use" is the most important item-
specific objective, then it could receive a weight of 10. If "Minimize Solid Waste Volume"
is least important, it could receive a weight of 1. Obviously, the importance of objectives
is determined by the decisionmaker. Thus, the weights should be defined by the
decisionmaker, probably assisted by a decision analyst who explains the weight selection
process.

In multi-attribute utility analysis, the predicted impacts could all be scaled on a 0-100
utility scale, multiplied by the importance weights, summed, and then compared to identify
the maximum utility management strategy. However, analytic difficulties and controversies
concerning the utility scale suggest that this may not be done. Instead, the separate
consideration of the importance weights and the impacts evaluation may be of most practical
use to decisionmakers.

6.3.2 The Analytic Hierarchy Process (AHP)

The AHP is a theory for measuring impact priorities in a hierarchic structure or,
more generally, in a feedback network when the diverse concerns of a problem — goal,
criteria, subcriteria, groups affected, and risky outcomes — cannot be put in levels as in a
hierarchy, where only the elements in a lower level depend on those in a higher level but
not the other way around (Saaty, 1980). There are real-life problems for which the
priorities of the criteria governing an outcome are set independently of what the outcomes
may be. For example, religions are based on commandments used to judge acts that occur
afterwards. For other problems, these priorities are modified later by what is observed in
practice as one uses precedents in court cases that are then adjusted to the current problem.
Finally, there are problems for which the criteria priorities can be derived only by examining
the present alternatives. This is particularly true in new situations in which there is no prior
experience to back them up. In feedback problems, the alternatives themselves may be
interdependent on each other.
A sound decision process must use known procedures that capture the best rank from judgments, through weighting and synthesizing them into a hierarchy in a manner that is compatible with how we synthesize a network with various dependencies. Also, we need to meet expectations involving prior knowledge or commitments by being able to modify the ranking that results from the mathematics without considering the commitments.

It may be useful at this point to compare AHP to MAUT. In MAUT, alternatives are ranked one at a time subject to strong axioms about lottery comparisons, transitivity of preferences, and rationality as defined by experts. AHP, too, can compare alternatives, one at a time, in the context of priorities, but it can also use relative comparisons, so essential when one does not have previous experience to create scales to judge alternatives one at a time. Relative, or paired, comparisons are convenient for scaling intangible factors side by side with tangible ones and for dealing with all sorts of dependence in a coherent way. They are also useful in explaining paradoxes on rank preservation and reversal encountered by MAUT.

The scientific literature of hierarchies has driven home to physical, behavioral, and systems scientists, and particularly to people interested in organization theory, the lesson that a hierarchy is a powerful mental construct for studying complex systems. Whether one is simply interested in understanding the actual structure and flow of a system or whether one is concerned with the functional interactions and impacts of its components, a hierarchic model of that system should inevitably be examined. Hierarchic organization is crucial to the synthesis and survival of large systems such as those encountered in LCA. Hierarchic systems have common properties that are independent of their specific content. The purpose of a hierarchy is to project impacts downward from the global to the local, from the general to the particular and special. It is a framework for reaching the most likely outcome in the face of risk and uncertainty. A hierarchy is also useful in determining the best policies (used as criteria or attributes or more general clusters) to follow to control the likely outcome. Both projection and control are what we do iteratively when we plan.

An example of the environmental impact of the level of water maintained in a dam (see Text Box 4 and Figure 6-2) makes the development of the hierarchy and the assignment of judgments clear. A matrix is set up by listing the criterion of comparison somewhere above and listing the elements to be compared on the left and on top. Then one begins with an element on the left and asks, how much more important is it than an element listed on top? When compared with itself, the ratio is 1. When compared with another element, if it is more important than that element, then an integer value from the scale is used or its reciprocal in the opposite case. In either case, the reciprocal ratio is entered in the transpose position of the matrix. Thus, we are always dealing with positive reciprocal matrices and need only elicit n(n-1)/2 judgments. We do not assume that people are consistent and, except for reciprocals in the transpose position, do not force judgments for consistency.
TEXT BOX 4

AHP: An Example

The following is an illustration of impact assessment using AHP. The scale for comparisons among pairs of elements consists of verbal judgments ranging from equal to extreme. Corresponding to the verbal judgments are the absolute (not ordinal) numerical judgments (1, 3, 5, 7, 9) and compromises (2, 4, 6, 8) between these values. The derived priorities have been proven to be very stable when small changes in the numerical judgments are made. Small can be as large as a whole unit or two, in either direction. A software package that implements AHP, Expert Choice, is available and allows one to conduct sensitivity analyses to test the effect of the uncertainty in the criteria on the choice of a best alternative. Steps to follow in using the AHP are presented in Appendix D.

What one does is to compare the importance, priority, impact, or contribution of the factors in a level with respect to their parent factor in the level above. Proceeding systematically from the top down (or from the bottom up), we arrange each group being compared in a matrix. We then ask how strongly the factor on the left dominates the factor on top with respect to the parent using verbal judgments converted to numbers. When the opposite is true, that is the factor on top is dominant, we enter the reciprocal value. For every judgment, its reciprocal is automatically entered in the transpose position, i.e., \( a_{ij} = 1/a_{ji} \). Appendix E provides special matrix information derived in this example. The priority vector is derived from the matrix, and the overall final result is obtained by successively multiplying all the entries of a vector by the priority of the parent criterion and adding values together. The process is repeated from top to bottom to obtain the overall ranking of the alternatives. Note that if a matrix of comparisons is perfectly consistent, the priority vector is obtained by adding each row and normalizing by the total value. If it is inconsistent, we need to raise the normalized matrix to large powers to capture the inconsistencies and obtain a best ranking priority vector by adding and normalizing its rows.

In the absolute measurement approach, each criterion above the level of alternatives is decomposed into intensities that are compared in pairs. Each decision alternative is then rated one at a time by assigning it an intensity. The intensities are weighted by the importance of their criteria as before, and an overall weight for each alternative is obtained. The absolute mode is used when it is desired to maintain the rank of alternatives when other alternatives are added.

By taking the priority of the intensity of each of the two alternatives, \( \frac{1}{2} \) full (\( \frac{1}{2} \)) and full (F), shown next to each intensity, weighting it by the importance of its criterion, and then adding up the criteria, we obtain the following overall ratio scale weights:

\[
\begin{align*}
\frac{1}{2} \text{ full} & = .417 \text{ normalized to .594} \\
\text{Full} & = .285 \text{ normalized to .406}
\end{align*}
\]

In this case, absolute ranking of the alternatives gives the same outcome as relative ranking.

\[\text{Expert Choice Software available through Expert Choice, Inc., 4922 Eusworth Ave., Pittsburgh, PA 15213. Phone (412-682-3844).}\]
DECISION TO KEEP DAM FULL OR HALF FULL

- Ecological
  - Biological Factors
  - Physical Factors
  - Risk of Disaster
  - Mental/Physical
    - Water Species*
    - Animal Species*
    - Vegetable Species*
    - Soil Erosion*
    - Water Quality Pollution*

- Human

- Resources Depletion
  - Conservation of Renewable Resources*
  - Conservation of Nonrenewable Resources*
    - Collapse of Dam*
    - Ability for Flood Control*
    - Recreational Opportunities*
    - Aesthetic Quality*
    - Pollution Reduction*

Half-Full Full

FIGURE 6-2 IMPACT ASSESSMENT USING AHP
The process of comparing elements in each level is continued down the hierarchy whereby the set of elements in each level is compared to the set of elements in the level above which they affect as to relative importance. From these pairwise comparison matrices, we generate a set of local priorities that expresses the impact of a set of elements on the elements in the level immediately above.

The principle of synthesis of priorities is now applied. Priorities are synthesized from the second level down by multiplying local priorities by the priority of their corresponding criterion in the level above and adding them for each element in a level according to the criteria it affects. The second level elements are each multiplied by unity, the weight of the single top level goal. This gives the composite or global priority of that element which is then used to weight the local priorities of elements in the level below compared by its criterion, and so on to the bottom level. In general, the bottom level of the hierarchy contains the resources to be allocated, or the alternatives among which the choice is to be made.

An intrinsic useful byproduct of the theory is an index of consistency, which provides information on how serious are violations of numerical (cardinal) and transitive (ordinal) consistency. The result could be to seek additional information and reexamine the data used in constructing the scale to improve consistency. Lack of consistency may be serious for some problems but not for others. For example, if the objects are two chemicals to be mixed together in exact proportion to make a drug, inconsistency may mean that proportionately more of the one chemical is used than the other, leading possibly to harmful results in using the drug. But perfect consistency in measurement, even with the finest instruments, is difficult to attain in practice, and what we need is a way of evaluating how bad it is for a particular problem.

6.3.3 The Impact Analysis Matrix (IAM) Approach

The impact analysis matrix (IAM) method is an exploratory, qualitative, expert judgment-based approach to impact assessment that directly builds on the results of life-cycle inventory. It was developed as part of a broader assessment of source reduction potential for halogenated solvents, which included an assessment of alternatives to such solvents in specific applications. The IAM allows a direct evaluation of the relative environmental burdens of a particular application of a halogenated solvent and its alternative(s), and makes explicit the tradeoffs among them. Two specific comparisons involving the uses of 1,1,1-trichloroethane (TCA) and alternatives were evaluated in this manner:

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1The approach to impact analysis described here is taken from the final report of the Source Reduction Research Project (SRRP), entitled Potential for Source Reduction and Recycling of Halogenated Solvents. SRRP was a joint multiyear research effort of the Environmental Defense Fund and the Metropolitan Water District of Southern California to seek source reduction and recycling opportunities for the use of halogenated solvents in 12 industrial categories. The impact analysis approach was developed by Jacobs Engineering Group, Inc., Pasadena, California, based on input and direction from SRRP's sponsors and staff.
• Substitution of a caustic aqueous cleaner for TCA vapor cleaning of metal parts.

• Substitution of supercritical CO₂ spraying of paint for TCA-based paint spraying.

These two comparisons were each conducted on two different levels, making explicit an additional area of tradeoffs among options; user- or shop-level impacts and global impacts were separately evaluated, using the process discussed in Text Box 5. User-level impacts were limited to those emanating from a boundary drawn around a particular facility using the halogenated solvent or its alternative. Global impacts incorporated all of the traditional life-cycle stages, including raw material acquisition and manufacturing, as well as use of the solvent and its alternative. In this manner, options that appear favorable from the user’s perspective but unfavorable from a global prospective, or vice versa, could be identified.

6.4 PARTICIPATORY VALUATION METHODS

Besides expert judgment techniques, there are also participatory (or "social judgment") methods for achieving public choice. A few examples are discussed here only to illustrate the nature and capabilities of such methods.

Participatory methods may be broadly classified under the headings of "revealed preference" and "expressed preference." The former is inferred public choice based on observed social behavior (social participation) using nonsurvey techniques. The use of accident statistics to gauge risk acceptance is an example.

In contrast, expressed preference is the direct response of a surveyed population to survey research instruments that elicit preferences. Two of the most common metrics are willingness-to-pay and willingness-to-accept-payment for the provision or loss of public goods. The outcome of such judgments is contingent on the hypothetical existence of a market in which such a good can be purchased.

The most familiar technique in this class is contingent valuation (CV). One CV method entails a property rights approach combined with the use of a referendum (single response) format for periodic assessment of the value assigned to public goods whose provision requires continuous expenditures. This method is presumably sensitive to both increases and decreases from the current levels of provision. Through use of the referendum format, the respondent is asked to come up with a valuation. Other CV methods are more interactive; for example, an interviewer may state values to which the respondent reacts. CV methods have the ability to value both market and nonmarket goods.
TEXT BOX 5

Expert Judgment and the Impact Analysis Matrix

The process of developing the IAM for each of the comparisons involved convening a group of experts to conduct the following sequence of steps; in each case, the outcome represented a consensus among the expert group.

Step 1: For these comparisons, the IAM consisted of five columns of inventory input (resource utilization) and output (emission) parameters, and seven rows of environmental impact categories. The impact categories, selected using expert judgment, included: (1) the major global impact areas specific to the use of halogenated solvents: global warming and ozone depletion potential; (2) other important global considerations: nonrenewable resource utilization; and (3) important regional and local impact areas: air and water quality, land disposal, and transportation effects.

In applying this approach in other settings, these categories could be varied to match the particular characteristics of each comparison and could also be expanded to include other impact areas (e.g., occupational exposures). It will be important to provide sufficient rationale for the selection and exclusion of various categories given the absence of any rigorous methodology or consensus to guide the selection process.

Step 2: The next step was to determine whether some cells represent either double-counting or meaningless comparisons. In the present case, for example, it was judged that aqueous wastes have no significant impact on global warming, so this cell in the matrix was eliminated. In this way, 17 of the 35 cells were deemed meaningful.

Step 3: Next, unweighted “scores” were assigned to each viable cell in the IAM. These scores were assigned relative to a particular option chosen as a base; a “+1” means a discernably larger environmental impact than the base option, a “-1” means a discernably lesser impact, and a “0” means little or no perceived difference in impact. Scores were assigned based on a combination of inventory data and expert knowledge of associated impacts.

Step 4 (optional): Next, the initial unweighted scores can be given weightings to determine whether results will change significantly: a relatively strong environmental impact is assigned a “++” and a relatively large reduction in impact is assigned a “--”.

It should be noted that the comparisons used to arrive at the weightings need not be restricted to a single impact category (i.e., row), depending on the views of the expert judges. However, the basis for assigning weights and the scope of comparison (i.e., within an impact category or across categories) should be explicitly described.

Step 5: The +s and -s, and 0s were then summed to derive an overall score for each row and column; if appropriate, the summation can be done for the entire matrix. Unweighted scores covered a potential range from +18 to -18, weighted scores from +36 to -36.

The user-level and global IAMs for the metal parts cleaning application are shown in Figures 6-3 and 6-4, respectively. As one example of the type of information provided by the IAM approach, one can compare and contrast the scores under the energy inputs column evaluated at the user (shop) versus global level. From the perspective of the user, impacts arising from energy input requirements are a dominating category, and are much higher for the aqueous-based relative to the TCA-based system because of the high pumping and heating requirements of the former; in contrast, viewed from a global perspective, impacts arising from energy requirements were found to be essentially the same for the two systems.
Valuation

FIGURE 6-3  LOCAL LEVEL IMPACT ANALYSIS MATRIX FOR AQUEOUS CLEANING VERSUS VAPOR DEGREASING

FIGURE 6-4  GLOBAL LEVEL IMPACT ANALYSIS MATRIX FOR AQUEOUS CLEANING VERSUS VAPOR DEGREASING

Notes:
b. Shaded squares signify no basis for impact on risk area.
c. Rating on -1 represents decreased impact, 0 represents the same impact, and +1 represents an increased impact.
6.5 INTEGRATED METHODS FOR VALUATION

It is important to note that the mere aggregation of individual preferences for public goods, which results from CV and related methods, does not constitute an adequate conception of the common good. Hardin’s "Tragedy of the Commons" is the classic statement of this discontinuity between personal and social judgments. In this regard, the EPA Science Advisory Board concluded that individual willingness-to-pay or willingness-to-accept-payment "may be inconsistent with fundamental ecological principles. Individuals may enjoy the benefits of ... [ecosystem] services without any knowledge of their existence, thus their preferences may imply values that do not reflect the ecological importance of natural systems and the services they provide to humans."

For this reason, it is necessary to seek integrative valuation techniques that combine ecological and economic principles, as well as expert and social judgments.

In the context of an LCA, it must be anticipated that the metrics or descriptors used to characterize a particular impact category will vary considerably. In some cases, multiple indicators will be needed to provide an adequate characterization. For example, the extent of quantitative description possible in a given impact category may vary from negligible to considerable. Moreover, even in those cases most amenable to quantification, additional means typically will be needed to translate the numeric values into terms that are meaningful and accessible to the decisionmaker. For these reasons, the depth of analysis and completeness of description will likely be uneven across the range of impact categories.

This characteristic of the output from the evaluation phase of LCA impact assessment reinforces the need in most if not all cases to employ combinations rather than single valuation methods. For example, inventory data may provide for quantification of air emissions of a variety of gases that contribute to global climate change. As a first step in the evaluation phase, characterization methods may be applied to combine these quantities into the common metric of CO₂ equivalents. This new quantity may have meaning to an atmospheric chemist, but will require further interpretation and assignment of value if it is to be of use to a decisionmaker. Thus, as a next step, an expert judgment-based valuation system may be employed to take up where the mechanistic/quantitative method left off. The expert assigns a value based on his/her understanding of the intermediate and ultimate impacts associated with an increased emission of a given quantity of CO₂ equivalents. Finally, a participatory valuation method may be employed to reflect the values of a broader public whose input into the decision has been deemed desirable or necessary. This step would seek to assign further value to the expert judgment by, for example, assigning a weight to the impacts predicted by the expert relative to other impacts.

One major research need in the application of valuation methods to LCA impact assessment is the development of approaches that allow for the integration of disparate valuation methodologies applied to the array of impact categories. Criteria for judging the individual methods and their amenability to integration need to be developed. Among other requirements, these methods need to be transparent and replicable.
SUMMARY OF LIFE-CYCLE ASSESSMENT
SETAC OPEN FORUM HELD ON MARCH 17, 1992, IN WASHINGTON, DC

Following the SETAC Workshop on LCA Impact Assessment, a one-day Open Forum was held March 17, 1992, at the Hyatt Regency Hotel in Crystal City, VA. The purpose of the forum was to gather additional public perspective on LCA issues and questions discussed at the Sandestin, FL, workshop. Approximately 130 United States and Canadian representatives from industry (56%), government (17%), public interest groups (6%), consulting firms (16%) and academia (5%) participated.

The forum consisted of a series of presentations summarizing the Sandestin Workshop, followed by group discussions. Discussion groups were asked to address six general questions posed to Sandestin participants (see Foreword). Each group summarized their findings through brief oral presentations and written synopses. This format was designed to allow all participants equal and ample opportunity to provide input.

Major comments and recommendations from the group discussions are summarized below. They do not necessarily represent the consensus of any group, nor do they reflect the views of the SETAC Foundation or forum cosponsors. Comments are arranged according to eight topic areas reflecting the breadth of the discussions.

KEY ASPECTS OF LCA

1. Key aspects of an LCA, in the sense of those parameters that should be measured, are highly case-specific and are not amenable to specific listing. In addition to conventional LCA parameters, other parameters may at times represent significant stressors. For example, the use of LCA in a plant siting decision might require consideration of labor needs (transportation, health requirements, etc.). In other cases (e.g., LCAs involving agricultural products), land use could itself be an important criterion.

Other possible criteria that may play a major role in an LCA study include occupational health, unplanned events/risk of disaster, impacts related to product storage, compatibility of products with conventional waste treatment, and cumulative and/or synergistic effects of chemicals.

GENERAL APPLICATIONS OF LCA

1. The possibility that LCA could be redundant with other environmental assessment or management procedures was recognized. A concern was that the establishment of formal methodology and reporting formats for LCA, like
those developed for environmental impact assessments, could render the LCA too cumbersome and voluminous to be practicable.

2. The inherent value of the LCA inventory analysis alone was recognized in: (1) bringing increased understanding to complex multi-institutional systems; (2) using industry average data to evaluate how an individual plant is performing; and (3) identifying, based on the performance evaluation, the possible areas for improvement. "Less is better" criteria may be adequate for some internal applications.

3. Corporate legal liability considerations relevant to compiling LCA data need to be understood, including antitrust considerations. LCA impact assessment may inadvertently create a regulatory benchmark.

4. The uncertainties associated with an LCA were seen as an obstacle in certain formulations of public policy.

5. Some felt that the unique benefit of an LCA was to ensure long-term sustainability of natural resources.

7. Some felt that the LCA should be restricted to internal applications only.

8. After refining the LCA method and adopting it within a company, it was suggested that it be integrated as part of the corporate ethic, similar to TQM. In this sense LCA was viewed as more than another tool, but as a corporate ethic. The dynamic aspect of LCA suggests that it is a continuing process, in a constant state of refinement.

9. If an LCA is a tool to educate the public and decisionmakers on the breadth of impact/issues of a product, integrating LCA data into a single criterion or scale would defeat this purpose.

ON THE CONCEPTS OF SCOPING AND BOUNDARY SETTING

1. It was pointed out that scoping could make the LCA less comprehensive and more vulnerable to popular opinion (i.e., public myths, and/or "issues of the moment"). On the other hand, it may be appropriate that the scoping process include those considerations that the consumer/layperson views as being important to environmental quality of consumer products.

2. If scoping is to be applied to identify key issues at the start of an LCA, an iterative review process is needed to reinsert issues possibly identified at a later stage. Also, the scoping process should clearly distinguish internal (LCAs conducted by an institution for internal use only) versus external applications.
3. Scoping activities, issues, and topics included and excluded from study during the scoping exercise must be clearly transparent, as should be assumptions and system boundaries. A formalized procedure for the scoping process is recommended.

4. Scoping must involve expert judgment and, to an extent, the use of a value system. This begs the question of who will enumerate the values.

5. The difference between scoping and boundary- or goal-setting needs clarification. Due to present difficulties (e.g., data gaps) in understanding global issues, the possibility of scoping out these issues was raised.

6. The broader the system boundary, the greater the potential for imprecise data. There is a need for scoping at both the overall product level and at the individual subcategory level.

**ON THE CONCEPT OF SOCIAL WELFARE CONSIDERATIONS IN AN LCA**

1. Some participants believed that the social welfare aspects of LCA require further consideration and development as they may involve several different and possibly ambiguous interpretations. Certain social criteria were viewed as being fundamental to public decisionmakers regardless of their inclusion in an LCA. It was recommended that the criteria and methods for evaluating social impacts be better defined. At this time, it is premature to saddle LCA with the entire range of social welfare issues, because it would make it burdensome to use and unwieldy as a decision tool.

2. It was further recommended that consumer product LCAs should recognize and quantify value-related product attributes such as performance, convenience, and function. Some believed that conventional LCAs tend to emphasize negative environmental or resource-related issues without considering product benefits.

**EXISTING PROCEDURES WITH POTENTIAL APPLICATION TO AN LCA**

1. A number of conventional procedures and methods developed for other scientific studies with potential application to LCA were identified, including:

   a. Peer or expert reviews. Expert review rather than peer review may more accurately define the type of review possible in LCA.

   b. Sensitivity analysis.

   c. Front-end analysis such as life-cycle cost accounting for defense initiatives.
d. Qualitative reasoning models; fuzzy logic; artificial intelligence; process control.

e. Design for environmental methods and programs.

f. Probabilistic models for unplanned events.

g. Decision theory.

h. European methods include the Swiss method and Swedish method of aggregation into six categories.

i. Mackay unit world method.

ON IMPACT ASSESSMENT AND VALUATION OF LCA INVENTORY DATA

1. It was suggested that risk assessment models could contribute a valuable perspective regarding the safety of emissions identified in an LCA. However, the requirement for additional data to conduct sound risk assessments was viewed by some as being overly costly or logistically impractical for assessing emission impacts at multiple sites.

2. Consensus is clearly needed on the selection of methods for particular impact evaluations and the reliability of the methods within particular disciplines.

3. The recommendation was made that emissions data should be aggregated according to the type of potential impact produced.

4. The tractability of LCA inventory data was viewed as being inversely related to the breadth of scope of the study (single plants versus industry-wide trends) and directly related to the availability of accurate models describing a particular impact.

5. Certain impacts cannot currently be quantified precisely, such as habitat quality. Overzealous attempts to reduce all parameters to numerical terms may be counterproductive. Qualitative descriptions should be used where the type of impact or level of technology does not allow precise quantitative measurement.

6. It was suggested that some product improvement decisions don’t necessarily require valuation. Valuation may be most important to those decisions involving trade-offs among environmental parameters.

7. Impact assessment was seen as value laden and subject to different analyses based on the values of the assessors. Key questions identified: Who makes
the value judgments? What role should society play? It was recognized that valuations performed by potentially impacted populations could introduce bias.

8. The decision criteria for weighing LCA parameters both during scoping and in the interpretation process are fundamental to the outcome of the impact assessment. In a sense, failure to consider LCA-related issues and parameters would represent a decision in itself.

9. The possibility of using a hierarchy of confirmed social values in impact assessment was raised. An approach that considers the common good over individual or sector benefits may be optimum.

10. The cumulative nature of many impacts, particularly those on a global scale, was recognized. Process improvements might be "contagious", such that a requirement by one entity could trigger a chain of improvements.

11. The suggestion was raised that restrictions should be placed on allowing companies to perform their own impact assessment for external LCAs, either by limiting their ability to conduct them in-house or by requiring adherence to a GLP-like system in their performance.
CHAPTER 8.0

RESEARCH NEEDS

The purpose of this chapter is to present the major areas of research needed to enhance the life-cycle impact assessment methodology. To facilitate the refinement and evaluation of research needs, the needs have been grouped into categories. These categories are aligned to the extent possible with the topics discussed during the workshop.

8.1 OVERALL IMPACT ASSESSMENT FRAMEWORK

The workshop defined what can best be termed a conceptual framework for life-cycle impact assessment. This framework embodies the general notion that stressors, caused by the resource/energy use and environmental releases of the product system, lead to various impacts. These impacts fall into four broad categories: Human Health, Ecological Health, Resource Depletion, and ecological, human health, and resource depletion impacts associated with Social Welfare activities. Within each of these broad categories, impact assessment takes place through the application of approaches for converting the inventory outputs into measures of stressors. Characteristic methods, evaluation methods, and cause-effect consequences networks are several of the terms used to describe the overall process by which activities within the product system get transformed into measures of impact on the environment. The Social Welfare category was judged to be largely outside the scope of the workshop. Considerable research is necessary to better understand the relationship of the Social Welfare category to a life cycle.

Three areas are worthwhile to explore for future technical framework research:

1. An explicit description of the scoping process is needed to assist in outlining boundaries for impact assessment at the outset of the project.

2. Hierarchical definitions relating cause-effect consequences networks to different applications of a life cycle are required.

3. Interactions among LCIs, impact assessments, and other activities, such as product risk/safety assessment and waste minimization audits, need to be defined.

8.1.1 Scoping Process

The scoping process was identified as an important component in conducting an adequate and cost-effective life cycle, particularly one incorporating impact assessment. A critical analysis of scoping methods used for other processes, such as Environmental Impact Statements (EISs), would enable us to judge the extent to which existing processes and policies can be used directly, or the nature of modifications that may be necessary. Based
on prior experience and the comments made during the workshop, it would appear that the
similarities outweigh the differences. However, the nature and extent of the scoping
procedures should be matched against types of life-cycle applications to ensure that the
appropriate impact categories are considered and that the level of detail in the analysis
matches the intended use of the study.

### 8.1.2 Cause-Effect Consequences Networks

It would appear worthwhile to develop a set of networks at varying levels of
complexity and completeness to describe what issues would have to be dealt with in a life-
cycle impact assessment. One possible approach to such a network was presented at the
workshop, but additional conceptual and technical development is needed before it is
possible to fully define what data are needed and which specific impact assessment methods
are optimal for a given application. Definitions for each of the impact descriptors within
a life-cycle framework (as opposed to the existing usage framework) are needed. This is
analogous to a more detailed definition of what constitutes an impact within the context of
a life cycle. Some positions were identified during the workshop, but further elucidation and
a benefits/limitations assessment should be done. As one example, a preference was stated
for the impact descriptors to be as close as possible to the releases to minimize the
judgmental nature of the impact assessment versus an alternative position that stated impact
descriptors should move as far as necessary toward ultimate impacts to adequately capture
the magnitude and direction of the consequence.

### 8.1.3 Component/Activity Interrelationships

Related to the research need is the necessity to determine what additional data for
impact assessment will be developed in the overall process. Should additional demands be
placed on the inventory to incrementally incorporate impact assessment requirements? If
so, what modifications are needed in the scoping/inventory process? Should life-cycle
impact assessment trigger additional data collection outside of the life-cycle process? If so,
how can the quality and relevance of this information be controlled so that it has the
maximum utility? How should internal (through the inventory component) or external
(through other corporate processes) data collection be optimally balanced?

### 8.2 ECOLOGICAL HEALTH

Within the ecological health area, a three-step process of increasingly extended and
more realistic models was proposed. The models are: (1) hazard profile/characterization;
(2) unit world or canonical environment model; and (3) "reality" or risk model. These
models all identify stressor processes to varying levels of specificity. The two higher levels
of complexity also explicitly specify exposure pathways and exposure concentrations, the
latter being generally site specific. Only the first two levels have ever been attempted within
a life-cycle framework.
8.2.1 Hazard Profile/Characterization Approach

The hazard profile/characterization approach has two levels of application. The simpler level is to construct mobility, persistence, and effects profiles for each of the environmental loading factors listed in the inventory. This method has been used in one form or another by several organizations. There is no consensus on what indices or measures are best to use, whether estimated values (from structure-activity modeling, for example) are of acceptable accuracy for use in a life cycle, and how this information should be interpreted in a life-cycle impact assessment. The proposed relationship between this approach and the more complex methods was that exceeding some kind of threshold in the hazard profile triggers the next level of analysis. Research on appropriate parameters, thresholds, and single versus multiple exceedance triggers is needed.

The simple hazard profile does not attempt to combine the loading estimates from the inventory with the intrinsic properties profile, research is needed to develop and validate a hazard matrix approach combining these. Finally, it is not clear how this approach would be applied to the nonchemical stressor aspects of ecological health. While it seems possible to identify parameters and thresholds, no attempt has been made to do so.

8.2.2 Unit World Model Approach

The unit world concept has been developed by Professor Donald Mackay of the University of Toronto. This is a generic fate and exposure computer model incorporating an average volume of the physical world (1 km³, 10% water/90% soil in terrestrial component as well as a biotic component (fish). The model may be used at several levels. A similar concept has been used by the U.S. EPA Office of Toxic Substances to evaluate pesticide fate in the Exposure Analysis Modeling System (EXAMS).

Research projects to evaluate these two generic exposure models for application to life-cycle would be useful. One critical evaluation issue is whether any product system itself could place a sufficient amount of a toxic material into a given world volume to exceed a toxic threshold, or whether all life-cycle applications would have to incorporate a baseline ambient concentration to which the product system contribution was added. As was the case with the hazard profiles, the proposed concept would require a set of triggers to decide when the generic approach might need to be augmented with site-specific exposure/effects modeling.

8.2.3 Site-Specific Exposure/Risk Assessment Approach

The application of this approach would only be selective. Because of the time and expense of performing this type of analysis, its use would be restricted to either life-cycle assessment dealing with a reduced spatial scope or after being triggered by the outcome of the generic modeling efforts described previously. There are many models and specific approaches for a site-specific ecological risk assessment. SETAC has sponsored a Pellston-type workshop (SETAC, 1987) on this topic, and it is an area of active research. A study
to evaluate methods and models in detail for their applicability to LCAs would allow linkages to be defined. This research need, however, is not considered to be of high priority because (1) the methods are already being refined for other purposes; (2) the use of this level of detail in an LCA would be rare; and (3) companies and agencies performing site-specific risk assessments are already familiar with the basic methods.

Additional research needs in the ecological health area follow:

1. A critical need exists for methods and approaches for factoring uncertainty into the impact assessment process. Users need to communicate how much uncertainty exists in all steps of the analysis.

2. The inventory frequently contains information on chemical loadings to the environment. Analysts attempting to perform a life cycle need an organized reference (Source) so that these loadings can be placed in perspective with loadings from other natural and manmade sources.

3. A critical need exists for case studies that demonstrate the application and utility of the various methods being developed for ecological impact assessment in relation to a life cycle.

4. Consideration should be given to developing a technical support document for an LCA. Currently, guidance on the various methods exists as a diffuse body of knowledge. This guidance needs to be pulled together into a technical support document.

5. Workshops should be held to train individuals on how to conduct an LCA.

6. Because of the complexity of LCAs and the impact assessment component, expert systems need to be developed to guide analysts.

7. The use of decision analysis methods needs to be further explored as applied to LCAs.

8. Chemicals in the inventory component need to be specifically identified instead of grouped in such categories as TOC, VOCs, and suspended solids. Regulatory chemical listings under SARA Title III, NPDES, and RCRA should be made available. Analytical research should be conducted on methods to achieve a complete material balance on TOC.

9. Methods to conduct toxicity tests on aquatic and terrestrial organisms on unknown mixtures of chemicals in air, water, sediments, and soil are needed. Concurrently, we will have to arrive at an understanding of how to interpret these data in the context of an LCA.
10. A need exists to continue to field validate our many short-cut predictive models of fate and toxicity tests. We cannot always rely on the generally very expensive field studies.

11. There is a need for information on the influence of habitat disturbance on maintenance of habitat fragments.

12. A distinction should be made between impact assessment methods, which deal with potential impacts, and those that attempt to estimate actual impacts.

13. At present, it is recommended that impact assessment concentrate on methods that deal with potential impacts and, in particular, on the impact equivalency method.

14. High priority should be given to the development of equivalency factors for a number of impact categories, such as resource depletion, human and ecological toxicity, and environmental disturbances.

15. There is a need for techniques for quickly estimating biodiversity changes and alterations in habitats.

16. The use of generic approaches has the potential to reduce data needs to a significant extent, but the generic analytical tools are not as well developed as site-specific measures.

17. For areas of significant impacts, there is a strong need for data capable of linking particular releases to particular impacts in a quantitative cause-and-effect relationship within an acceptable level of uncertainty.

8.3 HUMAN HEALTH

Both the data and the models likely to be used in the human health impact assessment component of life cycles have considerable research needs. Among the areas needing attention are the refinement of existing models, the filling of data gaps, and the development of entirely new models.

8.3.1 Model Improvements

Considerable scientific controversy exists over key issues associated with the assessment of certain health risks. Disputes range from conflicting opinions over fundamental issues (whether threshold doses exist for some carcinogens) to disagreement on the impacts of specific stressors (does TCDD produce chronic health effects at ambient exposure levels?). Among the improvements needed in our current assessment models are:
• Refinement and validation of "biologically based" dose-response models (such as cell kinetics models), especially for compounds thought to be non-genotoxic.

• Development of models for the possibly quite substantial natural variations in human susceptibility to cancer.

• Improvement in the ability of models to account for the cumulative, synergistic, and antagonistic interactions of various health stressors.

• Improvement in the ability of models to handle very large numbers of stressors. It is difficult enough to generate a valid risk assessment for a single chemical. Product LCAs may involve an assessment of hundreds, if not thousands, of discrete stressors.

• Development of models to account for variations in human susceptibility to chemical stressors resulting from compromised health status.

• Improvement of the models available for predicting human exposure to environmental loadings in the context of a life cycle.

8.3.2 Filling of Data Gaps

Among the many types of data that need to be collected to improve life-cycle health analysis are:

• Exposure data on many of the health stressors of concern, including ambient monitoring data and personal exposure data outdoors and indoors.

• Short- and long-term bioassay data on the majority of substances for which no such data exist.

• Speciation of effluent data into individual substances. Data such as "pounds of hydrocarbons" or "pounds of kraft mill effluent" have little if any value in health impact assessment.

• Data on unintended use of products that might present significant health risks. Typical examples of the unintended use of a product are the use of a plastic container to hold food after the container's original purpose has been fulfilled, or the use of gasoline to clean metal parts in a home workroom or garage.

• Data on human exposure to health stressors from nonmanufacturing, non-point sources. For example, there are relatively few data on human exposure to gases or other chemicals released from products and product packaging.
8.3.3 Development of New Models

In some instances, entirely new models will probably need to be developed to improve health assessment:

- For estimating the linkages among resource depletion, ecosystem damage, and human health impacts.

- For predicting the physical and other health impacts imposed by the fear of a product or its manufacture or disposal, whether those fears are real or imagined (e.g., sickness induced by the fear of radiation from a nuclear plant or toxics from a nearby incinerator).

- Predicting the indirect health effects of individual changes in welfare resulting from a product decision. An example of this type of impact is increased alcoholism induced by boom and bust cycles in mineral production.

- Evaluating social impacts. For example, estimating the possible indirect health effects of the macroeconomic impact of product decisions.

8.4 RESOURCE DEPLETION

Efforts in this category centered on two areas: definitions/depletion indices and practical measurements. Several key definitions of resource types and indices to measure consumption and renewal of resources are proposed in this report. These include careful distinctions between "flow" resources and those capable of being managed for sustainability and "stock" resources, i.e., those intrinsically incapable of being used at a low enough rate to avoid reductions in the global, regional, or local pool over time. Several gray areas were also identified. A comparison is needed of the general idealized approach proposed to the extant literature on topics such as resource sustainability and environmental economic elasticity models to identify analytical framework linkages consistent with the complexity of social, ecological, and economic concerns.

More specific measures of resource depletion will need to be developed. Each of these methods development research needs should explicitly deal with information acquisition, data quality, and verification. In particular, the following were identified:

- Develop a methodology for determining the correct basis for defining resource stock, including the issue of materials contained in products in use and the role of materials in landfills.

- Develop a methodology for linking resource development (e.g., exploration, access, and closure/post-closure) with depletion.
Research Needs

- Develop a methodology for evaluating the resource implications of certain water resources, in particular hydropower and conjunctive groundwater-surface water use (transfers), and the appropriate life-cycle spatial scale for this resource.

- Determine the applicability of economic models to specific resource depletion issues, e.g., geothermal heat recovery, landfill mining, and mineral/energy material acquisition. The objective is to define the effects of price on both total reserves (global, regional, or local basis) and economically recoverable stock.

One major research need in the application of valuation methods to life-cycle impact assessment is the development of approaches that allow for the integration of disparate valuation methodologies applied to the array of impact categories. Criteria for judging the individual methods and their amenability to integration need to be developed. Among other requirements, these methods need to be transparent and replicable.

In addition to these specific methods, there is a need to decide what kinds of impact assessment information will be maintained as a general database, what information will be collected along with the preparation of the inventory, what information will be obtained during the impact assessment step, and what will be obtained from activities outside of the life-cycle process.

8.5 RESEARCH NEEDS IDENTIFIED DURING THE OPEN FORUM

1. The presentation and evaluation of resource depletion data are critical.

2. A more streamlined method for impact assessment is needed.

3. Investigation of the use of economic models to assess impacts on social welfare.

4. It might be beneficial to identify a minimum amount of data required for an LCA.

5. A consistent approach is needed to deal with data gaps.

6. Method to objectively document both positive benefits and negative impacts of products within an LCA.

7. How to get broader participation by industries not presently involved in LCA methods development?

8. More case studies and accurate public databases are needed.
APPENDIX A

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APPENDIX B

GENERAL BACKGROUND ON VARIOUS APPROACHES TO IMPACT ASSESSMENTS

LIFE-CYCLE IMPACT ANALYSIS: AN ISSUES AND METHODS OVERVIEW

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DISCLAIMER

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LIFE-CYCLE IMPACT ANALYSIS:
AN ISSUES AND METHODS OVERVIEW

I. INTRODUCTION AND ISSUES FRAMEWORK

The nature of the impact analysis component of a lifecycle assessment (LCA) is not well defined, and procedures for conducting this portion of an LCA are still being formulated. This document describes for workshop participants some of the technical and implementation issues associated with conducting an impact analysis as part of an LCA. Responses to these issues may vary with the type of application for which the analysis is used. The document also describes some possible approaches to performing an impact analysis with the intent to stimulate discussion and generate additional ideas among the participants. Topical areas include: (1) defining what an impact is within the LCA context; (2) relating the scope and boundaries of the analysis to current or possible applications; (3) suggesting guidance for determining the feasibility of conducting an impact analysis; and (4) discussing how, or if, potential impacts can truly be measured in LCAs.

The purpose of the impact analysis component of an LCA is to measure, or at least describe, the environmental consequences of the production, use, and ultimate disposal of a product or process. Impact analysis could be used to compare two or more existing products. Impact analysis is also useful as part of the iterative improvement process in determining how alternative design modifications, material specifications, manufacturing technologies, and use/reuse scenarios would change the environmental consequences. Thus, impact analysis is appropriate when two critical conditions exist: (1) the environmental consequences are not directly specifiable given the air, water, or solid wastes generated or the amounts of resources consumed per functional product usage unit from the inventory; and (2) the desired application of the LCA requires that impacts rather than releases or energy/materials usage be measured.

Impact Definition

It is not necessary for all LCAs to include impact analysis. Its inclusion depends on the objectives of a study and the nature of the use of the information. For example, the inventory alone may be used to identify opportunities for reducing environmental releases, energy, and material use (Fava et al., 1991).

If an LCA application is to include impact analysis, however, it is necessary to clearly define what is considered an impact in the context of an LCA. The workshop participants will discuss whether there should be a consensus definition of the set of relevant impacts and acceptable measurement approaches for those impacts. Alternatively, the preparer of a study will choose a set of measures and approaches, perhaps from a group of “approved” approaches, and be required only to clearly describe what was done. The direction of this discussion will significantly affect the amount and nature of future research needed.
Previous impact definitions have mixed measurements spanning the spectrum from environmental releases or the amounts of energy/material use (source or causal level), through translation of the releases into concentrations in various environmental components (effect level), to a comparison of environmental effects to exceedance of environmental impairment thresholds (receptor level). Unless the definition or definitions being used in a given situation is clearly specified, it will not be possible for an objective party to be assured that a reliable, consistent set of impact measurements has been chosen.

For purposes of workshop discussion, one way to define impact might include a three-step process: (1) issue area identification, (2) consequence measure specification, and (3) impact analysis method selection. First, the impact areas or categories of relevance should be defined. While the workshop participants may expand or contract the categories, the following list has been suggested as appropriate starting point (Assies, 1991): Resource Depletion, Pollution, Environmental Impairment, and Other. The environmental consequences of interest would be specified for each category. Possible consequence measures and their associated impact analysis level are discussed in a subsequent section entitled Consequences Measures Organization Schemes. Some examples include air emissions, habitat destruction, and chronic respiratory impairment. Quantitative or qualitative measures must be devised to relate the information derived from the lifecycle inventory to the corresponding environmental consequence. These first two steps ask what impacts we wish to measure. The third step asks how we measure impacts. It defines a set of procedures or processes inventory data manipulation, possible augmentation by additional information, and determination of their potential contribution to the environmental impact or risk.

The degree to which various measures actually correspond to impacts is important. Consequences can be measured on several levels. For example, one could envision a gradation from the inventory data themselves (amounts of greenhouse gases emitted, expressed as CO₂ equivalents), through measures of potential temperature change associated with the emission, to ultimate impacts (habitat alteration associated with sea level changes, crop pattern shifts, and so on). Various potential applications of LCA impact analysis could have differing requirements along these gradients. For example, if an LCA is to be used in an external comparison application, it could be specified that all impact analyses use the same set of consequences measures or at least utilize a set selected at comparable levels. Otherwise it might be possible to obscure or overlook a critical consequence.

One possible six-step strategic framework for the impact analysis component of LCA is:

Specify LCA Impact Analysis Application

Select Issue Areas/Consequences Measures

Identify Analysis Methods for Consequences Measures

Quantify/Qualify Impacts
Lifecyle impact analyses could be used in a variety of ways by different users. Three aspects of boundaries and scope are discussed in this section: (1) applicability of analysis methods to varying time and space scales inherent in a product's life cycle, (2) system boundary setting for various applications, and (3) impact evaluation issues associated with related corporate activities, such as safety assessments and waste minimization programs, that may parallel or overlap with LCA. Users require definitions, and specification of advantages and disadvantages for each method applicable to the impact analysis. Some applications for internal corporate use may be easy to conduct using more qualitative or descriptive impact approaches, whereas external studies involving government policy decisions or consumer purchasing may necessitate state-of-the-art methods and may require extensive analytical procedures.

In theory, a three dimensional LCA impact analysis method could be conceived. These three dimensions would include quantity, space and time. The degree of resolution of these dimensions would then be a function of the application. Some techniques may be too temporally or spatially demanding to be feasibly incorporated into an LCA framework. At the other extreme are methods not be capable of dealing with these considerations at all. The geographic scale and location specificity of the impacts will especially affect the scope of the analysis and the feasibility of applying various techniques. The scale effects in LCA range from global through continental, national, regional, local and site-specific. Time scales are also highly variable from very short turnovers associated with single use items to highly durable, multi-use goods. Time frames may even vary across different aspects of a product. For example, a disposable item may have a consumer use lifetime of only minutes before an inventory data point for solid waste generation is created. Impact time frames for the final disposition alternatives of that same item may range from days to decades. If impact methods are to incorporate time or space considerations, then one should expect clarity and consistency throughout the analysis in including or excluding these dimensions.

In general, the scope and boundaries of the impact analyses should correspond to the scope of the inventory. If the impacts of two manufacturing processes are being compared and the resulting products are the same, the inventory could be confined to some fraction of the product life cycle. Correspondingly, the impact analysis need not extend beyond the product system boundaries defined for the inventory.

Some aspects of life-cycle impact analyses may parallel safety assessments of product-packaging materials or product ingredients and waste minimization programs. A product safety assessment is a determination that a product or its packaging does not cause unacceptably adverse human health or environmental impacts when used and disposed of
in the intended fashion. Product safety assessments typically are site-specific with regard to manufacturing releases and tend to be highly quantitative, reflecting actual exposure and risk assessment procedures.

Product safety assessments potentially overlap some of the issues of life-cycle impact analyses but are not as broad in scope. For example, a manufacturer may not include upstream activities, i.e., those performed by suppliers, or some consequences that may be at a low level when considered on a facility specific basis. Some safety assessments may be limited to areas covered by worker and consumer protection regulations, while others may be more broadly based and include a range of human and ecological effects beyond those currently regulated. A product safety assessment could potentially provide a starting point for determining additional hazard- or risk-related information needed for an impact analysis.

Industrial waste minimization program activities may also parallel certain aspects of LCAs in areas where process emission impacts are of interest. The feasibility study component in a waste minimization effort should account for facility-specific characteristics of wastes, including their intrinsic toxic properties. It is rare for a waste minimization study to incorporate hazard, risk, or exposure assessment. The engineering process analysis within a waste minimization program identifies alternatives that could result in lower quantities of waste, less toxic waste, or both.

It is important to recognize that both safety assessments and waste minimization assessments tend to address site-specific scale impacts while lifecycle impact analyses will usually be at a regional to global scale. Thus, impact analysis methods, such as risk assessment, suitable where considerable site-specific data are available, may not be feasible for many lifecycle applications.

The following discussions of categories of applications for LCA impact analyses are presented in order of increasing scope or requirement for broad, quantitative analyses and the significance of consequences of making an incorrect decision (Figure 1).

**Corporate Strategy Development/Internal Communication**

Each company that uses life-cycle assessments as part of its internal decision-making process should have (1) a policy covering the development and use of life-cycle data, and (2) a process for performing and applying the impact assessment methodology. The corporate policy should state what the company is committed to doing with the LCA information. Activities which may fall into this application category include (after World Wildlife Fund and Conservation Foundation, 1990):

- Analyzing overall impacts of generic materials.
- Evaluating resource impacts associated with current products.
- Benchmarking processes, ingredients and systems causing major impacts.
- Supplying information for product and material procurement specifications.
Providing guidance in long-term strategic planning to senior management.

For example, the policy might state that "life-cycle impact analyses will be performed on all research and development projects to ensure minimal environmental impact and ideally environmental neutrality." However, such terms as "minimal impact" and "neutrality" must also be procedurally defined to ensure that corporate policy is actually implementable by the staff. For most corporate decisions an informal approach to impact analysis may suffice. Depending on the scope of the study, the consequences evaluation criteria selected could also be quite limited, perhaps encompassing only a few criteria where the inventory loadings are proportionately high.

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Figure 1 Range of LCA Impact Analysis Applications

The scope of an impact analysis in this application category might be defined by first examining the issues investigated in a prior or concurrent risk analysis or a safety assessment for a product. The LCA could include coverage of suppliers of the raw or intermediate materials, disposal routes for the product, and cumulative global considerations difficult to address at a site-specific level.

Product Design/Modification

This category of application focuses on the process for decisions made by product designers and engineers that lead to an environmental quality advantage for a new or redesigned product by reducing adverse impacts. Activities which may fall into this application category include (after World Wildlife Fund and Conservation Foundation, 1990):
• Analyzing overall impacts of material design choices,
• Comparing and selecting process options on the basis of relative impact,
• Comparing existing to redesigned but functionally equivalent products, and
• Training designers in the use of preferred materials.

At issue from an impact context is whether decisions made regarding changes in the design of a product would go beyond resource usage and environmental loadings tabulated in the inventory. This application is part of an internal corporate activity for comprehensively dealing with regulatory compliance activities, product safety assessments, and LCA impact analyses. If a design change involves a reduction in all environmental loadings and energy/materials, no impact analysis is needed. When a design change involves a mix of reductions and increases in component weights, raw material usage, and environmental emissions, an informal impact analysis could be conducted as a first step. If a more quantitative analysis needs to follow, it could be conducted as a stand-alone effort or incorporated into a revised and expanded safety assessment encompassing local effects as well as larger-scale consequences.

Facility Siting/Operations

This application category is associated with environmental issues at one or more locations where a product may be manufactured or warehoused. The information may be used either internally (planning) or externally (communication), but would typically be more local in scope than other LCA impact applications. Models can be used to estimate the emissions from a facility and the consequent exposures. Applications for internal usage may focus on alternative manufacturing processes or final manufactured products. Lifecycle analysis may not be the optimal tool in this case since the evaluation could largely consist of comparing site-specific employee, public, and environmental exposure levels to impact threshold criteria. Discussion at a recent European LCA workshop suggests that there is some lack of agreement on inclusion of worker protection issues of this type in lifecycle impact studies (Schmidt, 1991).

If the analysis is for external purposes, e.g., to show potentially affected parties how alternative sites balance local, regional, and national impacts, the process may lend itself better to use of an lifecycle approach. Results communication may focus on geographic features and regional to local spatial impacts may be emphasized.

Public Information/External Communication

This application category addresses the use of lifecycle data for a variety of public purposes associated with marketing, competing product claims, and similar comparative statements. Activities which may fall into this category include (after World Wildlife Fund and Conservation Foundation, 1990):
• Evaluating environmental impact claims made by other manufacturers,
• Enhancing market competitiveness through product advertising,
• Providing information to consumers about resource and environmental impact characteristics of products.

This external application requires a rigorous, quantitative, and transparent approach. How the impact analysis information is conveyed to the audience is as important as the analytical process. The information should be presented in the overall context with the product or activity, i.e., not selectively presented to emphasize advantages. The scope should include all elements environmentally significant to the consequences of the product or process. The boundaries of the impact analysis must be clearly defined and available to the public. In particular, the consumer should be able to discern which environmental consequences were considered, which were not and why, and generally how the impacts were determined. The results should clearly indicate which impacts are significant, either positive or negative.

Selection and communication of appropriate consequences measures should probably not be entirely left to the LCA preparer. Even if the study will directly provide a comparison between or among products, the interpretation of the impact results should clearly state the limitations of the impact analysis with regard to the environmental consequences evaluated. For example, a checklist could be used to specify which of a comprehensive list of possible measures was used.

The LCA impact analysis scope provides two critical elements in communicating relevance to the consumer: (1) a context for evaluating the significance of the consequences of the product or process relative to other consequences, and (2) a measure of the consequences for the product or process compared with activities familiar to the consumer.

Governmental Decisionmaking/Policy Formulation

The policymaking application addresses the concerns of public decisionmakers and regulatory agencies interested in environmental impact consequences of a product or process, or alternative products and processes. The analysis needs to be highly rigorous for this application to meet the scrutiny that might be imposed by regulators. This impact analysis may be appropriate in situations where changes in a process cause emissions receptors to change, such as solid waste instead of wastewater. Activities which may fall into this category include (after World Wildlife Fund and Conservation Foundation, 1990):

• Supplying information for legislative or regulatory policy restricting use of materials based on adverse impacts or encouraging use based on impact aversion
• Supplying information to set standards regarding product advertising.
General Background

- Gathering environmental impact information about products or materials
- Identifying gaps in knowledge and prioritizing research.
- Evaluating and differentiating among products for labelling programs.
- Developing policy regarding resource conservation, and impact reduction.
- Evaluating manufacturers’ claims.
- Evaluating resource impacts of source reduction and alternative waste management techniques.

At the policymaking level, LCA impact analysis could be used to attach credits or fees to a product or process for decreases or increases in environmental impacts. The uncertainties of both the inventory and impact analysis must be kept in mind in the final evaluation where the uncertainties associated with various impacts are cumulative. Environmental consequences selected for such an application and the associated analysis methods should be consistent across studies to ensure comparability.

Consequences Measures Organization Schemes

There are several approaches to organizing consequences measures. From the aspect of areas of environmental concern, measures fall into two broad categories: (1) measures directly associated with resources (e.g., materials, water, and energy feedstocks) in manufacture, consumption, or disposal of products and packaging; and (2) measures associated with emissions or outputs into air, water, and land from the manufacture, consumption, or disposal of the products or packaging.

Resource Use Consequence Measures

The preferred context for understanding environmental consequence measures associated with resource use impacts is the degree of resource renewability, scarcity or sustainability. Sustainability is the rate of consumption or use of a resource in relation to its availability or assembly (production). An advantage of measuring the “degree of sustainability” rather than the absolute categorization of “renewable” or “nonrenewable” is that it represents a continuous parameter. In addition, renewability may not indicate whether depletion of the resource is occurring. However, due to considerations with measuring sustainability, renewable versus non-renewable may be more practical to use in the short term.

Resource use consequences have a time scale associated with the rate of renewability. For example, fuels produced from biomass crops, such as corn, versus production from wood, versus fossil sources differ in their time scales of renewability from less than a year to geologic time periods. Extractable reserves of resources are often related to economic costs which may change over time in response to world or national conditions and extraction or
exploration technologies. The impact of resource depletion is also affected by the availability of suitable substitutes.

Environmental Release and Effects Consequence Measures

As noted earlier, release consequences could be specified at various levels, starting at the point of emission and continuing through abiotic pathways to biological organisms, including humans. In addition, measures identified for one consequence may not be mutually exclusive to those for another consequence or category. Acidification potential illustrates these two points. If one considers the general consequence of acidification potential, several possible measures might be conceived. For discussion purposes these might include: acid release amounts, expressed as proton equivalents; pH buffering capacity reduction in water or soil; metal, e.g. aluminum mobilization potential; human respiratory damage and emphysema mortality. These measures range from items associated with the release itself through effects in environmental media to impacts to human receptors. One or all of these could be legitimate measures of acidification impacts. If one chooses to organize the consequences by phenomenon or source versus the receptor level, careful attention must be given to avoid missing a consequence or double counting. Alternatively, one could arrange the consequences by receptor and list the measures associated with specific causes linked back to the inventory data.

The list of possible environmental consequences measures could be exhaustive. However, tradeoffs between the ability of the measure to assess impact and the effort needed to make the measurement should result in a set of measures that address the scale and nature of LCA issues while still maintaining a base in good science.
II. IMPACT ANALYSIS METHODS

Methods for assessing impacts on the spatial scope and time frame of concern to an LCA are not well developed. Traditional risk assessment frameworks rely on two key features: (1) estimation of exposure, and (2) evaluation of consequences over some time period of interest and within a geographical space. The number of locations where environmental releases may occur is considerable; similarly, the time frame over which the activities may occur is long. Further, it is unlikely that any impact analysis methods implemented for LCA will require detailed quantitative estimation of consequences for each individual event because of time and cost demands. The following discussion gives some criteria and presents some example methods that are believed to be good candidates for discussion.

Evaluative Criteria

Several preliminary criteria have been developed to help guide the selection of the LCA impact analysis methods and their application has yielded only preliminary findings. Historically, impact analyses have been used in a variety of ways. The problem is how to recognize the more appropriate methods for different LCA applications. The following criteria help to define appropriate LCA impact methodologies.

- **Compatibility** – The data requirements of the life-cycle impact analysis component are of the same genre as the data produced by the life-cycle inventory component and the data can be used without significant conversion or extrapolation.

- **Comprehensive** – The impact analysis method accommodates all five stages (i.e., raw material acquisition, material manufacture, product fabrication, filling/packaging/distribution, and consumer use and disposition) of the life cycle of a product, process, or activity.

- **Flexibility** – The method can accommodate varying levels of quality of inventory data, and may even be able to accommodate missing data from the inventory analysis.

- **Simplicity** – The method and procedure have been tested and verified and are easy to use; computer software may also exist to facilitate analyses of the inventory data. Using expert review procedures for conducting impact analysis does not require highly specialized training.

- **Credibility** – The results are representative of real-world cause-and-effect relationships, they are believable, and they do not require complex explanations for the intended audience to understand the findings.

- **Cost-Sensitive/Time-Sensitive** – The method provides information relatively quickly and easily for a small amount of effort (person-hours).
• Acceptance/Relevance — The method meets the real or perceived needs of the user of the analysis, and it is relevant to the LCA process and purpose.

These criteria were used to recognize the methods identified in the following discussions.

Impact Method Families

Although numerous assessment methods and variations have been developed for assessing impacts of various types, few LCAs include an impact analysis component. The following discusses some of the approaches to LCA impact assessment that have been used in practice or developed conceptually. Three families of impact analysis methods are suggested on the basis of similarities in requirements and approaches used in conducting the impact analyses.

Intrinsic Properties and Use Indices Impact Analysis Family

This family of impact analysis methods is based on the characteristics of the inventory data, such as numerical toxicity of emissions (e.g., mg/kg) or volume of material or effluent produced (e.g., ton/yr). The term “intrinsic properties” refers to the characteristics of the environmental release that influence impacts, such as mobility or toxicity, and not to the actual impacts. The term “use indices” refers to a mathematical expression relating the supply and demand of a resource, e.g., board-feet of timber available on a continuing basis from an area versus the board-feet needed from that area to meet demands. In this example, the actual impact, such as deforestation resulting in erosion or loss of wildlife, is not evaluated. Four examples of impact analysis approaches identified in this family are discussed below. These methods are not exhaustive of the possibilities; additional modifications and variations of them are expected to be discussed at the workshop.

Less is Best. Although not an impact analysis approach per se, the Less is Best approach uses comparative measurements to detect the direction and magnitude of change of given inputs or outputs associated with product or process alternatives. This method characterizes the majority of LCAs conducted in North America at this time. Essentially there is no further analysis beyond the LCA inventory, except for the explicit comparison and discussion of the environmental loadings and resource consumption estimates. The alternative with the lower loadings for a given equivalent usage is considered preferable.

This approach uses data directly produced by the life-cycle inventory to produce a list of inputs and outputs in which one product or process is compared with a functionally equivalent alternative. Quantities of inputs and outputs of the alternative product or process relative to the first product or process are indicated on the list by “+”, “-”, or “=“; these symbols would indicate whether the second item has fewer, more, or a similar amount of requirements or emissions. The approach might be used to compare products from different manufacturers or different processes for manufacturing a single product. Its application to LCA may be illustrated by the recent introduction of concentrated “ultra” laundry detergents that have fewer fillers and require less packaging. In this case, the ultra laundry detergent may have been considered superior because it got a minus in several
inventory categories relative to a traditional detergent. This approach treats all environmental loadings and resource consumption equally.

Solid waste is an area where the use of inventory data alone may suffice for impact analysis. Based on the USEPA waste management hierarchy (reduce, reuse, recycle, dispose), an argument can be made that the volume of solid waste requiring disposal can be used as a measure of impact. Possible candidate measures are:

- Total solid waste volume generated per year divided by the total volume of industrial production per year.
- Related but more specific measures (e.g., an average solid waste generation rate per unit production).

The Less is Best approach assumes that no changes occur in the product life cycle to achieve the higher detergent concentration to offset the reduced packaging materials and energy requirements. This approach further assumes that any reduction in loadings into one medium – air or water or land – has a positive effect. While the Less is Best approach is able to accommodate both qualitative and quantitative data, like most methods, its comprehensiveness is based on the extent and quality of the inventory data available. The approach does not permit the effects of trade-offs of one emission or compound for another to be assessed. This limits the method’s credibility. For example, there is no way to express whether loadings of dioxin to the air are worse than loadings of methylene chloride. Levels or degrees of impacts are not measured with this approach, but it is useful in making relative comparisons between similar products and processes that have common types of loadings. Direct use of the inventory data for impact analysis is limited to those activities where any environmental release or resource allocation is itself deemed to be undesirable.

Yes/No (Checklist). The Yes/No (Checklist) impact approach considers the presence or absence of a given category of impact based on a type of input or output in an inventory analysis. For example, if the LCA inventory lists CFC emissions to the air, the atmospheric ozone depletion impact area would appear affirmative on the checklist. This approach suggests cause/effect relationships in which a loading or resource consumption does or does not contribute to an impact. The results of such an impact analysis would likely take the form of a list of possible impacts, such as heated wastewater/thermal effects on biota or carbon dioxide emissions/greenhouse effect and a column for applicability, e.g. “yes” or “no”. The checklist would be compiled for all effect areas agreed upon in advance of the impact analysis. Inventory data from all lifecycle stages can be used without modification in compiling the checklist.

The Yes/No (Checklist) approach is straightforward, easy to use and it yields an overall list of categories that is consistent across product types; these in turn, facilitate comparisons. The approach can use or accommodate data of varying quality because it relies only on the presence or absence of a release or resource consumptive activity and not on the data quality. This approach, however, does not discriminate between high or low quantities of
levels in a category, and the effects of trade-offs are not quantified. Its flexibility for impact analysis is dictated by the extent of data available from the inventory. Variations on this approach relate primarily to the choice and level of impact measures included in the checklist.

**Relative Magnitude.** The Relative Magnitude approach is similar to the Yes/No (Checklist) approach, except it relies on ranges of input/output data from the inventory that are organized into subranges. Scores for the subranges may be set either subjectively or objectively using the inventory data directly for the range boundaries. For example, the range of quantities of SO₂ emitted to the air or the volume of solid waste generated may be divided into five subranges with each subrange receiving a score of one to five; the subrange with the lowest quantity would receive a score of one, whereas the subrange with the highest quantity would get a score of five.

The procedure for establishing relative magnitude is relatively straightforward, but rules would need to be developed to establish the significance of a given subrange relative to other subranges of a category. As a hypothetical example, the quantity of SO₂ emitted into the air might be broken down into five equal, but increasing, subranges (i.e., if the total possible range of SO₂ emitted per 1000 product units is 0 to 100 lbs per day, then appropriate subranges might be 0-20, 21-40, 41-60, 61-80, and 81-100). The higher the score, the greater the potential for acid rain and human health impact due to SO₂ emissions. Qualitative inventory data are usable in this approach, but missing data are difficult to accommodate. Subjective decisions may be necessary to determine the subrange boundaries and scores for both qualitative and quantitative data. Choosing the boundaries is one source of variation within this method. This approach allows credible evaluations of products and comparisons among products and product types, but impacts associated with a particular product or process are not measured. As with the Yes/No (Checklist) approach, variations on the Relative Magnitude approach relate primarily to the choice and level of impact measures included.

**Resource Consumption Ratio.** The Resource Consumption Ratio impact analysis approach measures impacts on natural resources by comparing the magnitude of energy and material consumption with the available supplies or reserves (Resource Consumption Ratio = consumption per unit of use per unit time/supply per unit time). Consumption, or input values, for each stage of the life cycle can be taken from the inventory data. Information on the available supply may be obtained from other sources e.g., company data, government publications. The measurement units may need to be normalized to a standard production time unit (e.g., annual). The inputs can have various measures for yields (e.g., annual acre-feet of water) or reserve use rate (e.g., mmton coal/yr). For example, the tons of pulpwood needed to produce 50000 tons of bleached, white paper per year would be divided by the long term supply in tons of pulpwood available to the mill per year, in order to calculate the consumption ratio for wood.

The Resource Consumption Ratio method can also be used to evaluate the impacts of emissions on natural resources. An example of assessing the impact of emissions would be
measuring the oxygen depletion in a receiving stream relative to the supply of oxygen through reaeration, or the amount of an oxygen depleting substance discharged relative to the environment's assimilation capacity. For some releases, however, the assimilation capacity and adverse effects thresholds can be very difficult to assess.

The consumption ratio approach is a conceptually simple process and can be used to create a single measure of impact (Figure 2). However, it may require considerable effort to develop the input values and to document the consequences of various pollutants. The significance of a ratio is not clear. A decision will need to be made whether to calculate the ratio of the individual product or the incremental total demand, i.e., product requirement relative to total national or world demand. Additional work is required to determine the significance of those values greater than 1 (where consumption exceeds supply) as well as reaching consensus on a weighting system.

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<tr>
<th>RESOURCE</th>
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<th>R₂</th>
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\[ C_{11} = \text{Consumption per unit of use per unit time/supply per unit time} \]
A variation of the resource consumption approach has been used in Switzerland for packaging evaluations and in the United Kingdom for washing machines (Schmidt, 1991).

This method, known as the Emission Threshold or MIK Value Method, determines the volume of air, water, or soil required to dilute a release from the lifecycle inventory to the relevant regulatory ambient standard. The "critical volumes" are determined by computing each of the pollutant loading ratios as the quotient of the release (mass per unit of product) and the allowable input to the respective environmental medium (mass per unit of environmental volume) and summing over all pollutants discharged to that medium. The environmental impacts of packaging materials were assessed using these criteria plus two other criteria not involving consumption ratios.

Advantages of this version of the Resource Consumption Ratio approach include:

- Application in several practical case studies and international recognition.
- Method transparency for pollutants which have ambient environmental standards.
- Ambient standards are set on a national basis; larger spatial scales require aggregation of various values.
- In principle, critical volumes could be determined for degradation products.
- Straightforward interpretation of the critical volume concept.
- Separation of the critical volume for each medium, facilitating transparency.

Disadvantages of this version of the Resource Consumption Ratio approach include:

- Non-chemical stressors, such as noise, radiation, and physical changes, are not considered.
- Emission thresholds reflect evolving scientific knowledge, hence, must be updated.
- Ambient environmental standards are not based on science alone, but include economic and political considerations.
- Lack of ambient standards for all pollutants in each medium.
- Pollutant dispersion and deposition subsequent to initial release are not considered.
Another variation on the Resource Consumption Ratio approach is part of the Environmental Priority Strategies in Product Design (EPS) system developed by Volvo and reviewed by Schmidt (1991). In the EPS method, the total impacts of a product from cradle to grave are evaluated using an aggregation of four impact elements (similar to the Ecofactor approach discussed below), including consumption of natural resources (raw materials, energy, and land). The consumption of natural resources is evaluated based on three factors: amounts (e.g., kg/unit of product), known resource reserves (kg/capita), and an assessment factor that accounts for the ecological scarcity of the resource. In this case, ecological scarcity refers to the relationship between the limited carrying capacity of the natural environment, defined as the existing legal limit for a pollutant, and the actual anthropogenic emissions of the pollutant.

**Exposure/Hazard Assessment Impact Analysis Family**

This family of impact analysis methods is based on calculations of hazards/risks to environmental components using data from the inventory analysis. Three approaches in this family are discussed below.

**Consequences Network.** The Consequences Network impact analysis approach is a comprehensive evaluation of the various causes and interrelated effects and impacts associated with a product action. This approach uses branching pathways to delineate cause-effect relationships (Figure 3). The branching pathways indicate how closely the impact is tied to a product action (e.g., primary, secondary and tertiary impacts). Primary impacts result directly from product actions (e.g., release of inorganic effluents from a manufacturing facility, which degrade water quality in a receiving stream). Secondary impacts are one step removed from the product action (e.g., the inorganic effluents from the facility to the stream have an impact on fish health or survival). Tertiary impacts are two steps removed from the product action (e.g., water effluents released to receiving stream contaminate or kill fish and impact the fishing industry). An example network approach (Figure 4) evaluates 19 components, 64 parameters, and many measurements that are arranged in hierarchical order and are associated with one of four types of environmental categories (i.e., physical/chemical, ecological, aesthetic, and socioeconomic). Impacts at the component level (e.g., air quality, population, noise) are quantified by a multivariate statistical program that collapses measurements or estimates of several parameters (e.g., S02, NOx) into a single component-level environmental quality index. The impacts for an individual component are determined by the following formula:

\[
EIU = ciu \times eq \text{ (with product)} - ciu \times eq \text{ (without product)}
\]

where:
- \( EIU \) = environmental impact units (units of impact per unit product)
- \( ciu \) = component importance unit (relative weighting) and
- \( eq \) = environmental quality factor for a single parameter.
Environmental Quality (EQ)

Figure 3 Example Environmental Assessment Tree

Figure 4 Hierarchical Structure of Consequences Network Approach
General Background

The EIUs calculated for each parameter are summed first for each component level and then for all components in all environmental categories, in order to determine overall impacts. The CIU is a relative weighting determined by quantifying the research team's subjective value judgments using sociopsychological scaling techniques and the Delphi procedure. The weighting process consists of ranked pairwise comparisons and controlled feedback.

The Consequences Network approach has several advantages. It is comprehensive, replicable, interdisciplinary, objective, based on explicitly defined criteria, able to assess total impact, able to detect environmentally sensitive areas, and thorough. It is thorough in that it accounts for interrelationships of the various effects resulting from project actions and sensitive system connections. The method uses explicitly defined criteria and procedures so that evaluation criteria and values are not arbitrarily assigned. However, a difficulty with this system is the time and effort required for such a thorough impact analysis. Another problem is the difficulty in implementing the approach without a developed environmental assessment tree. Additional work is needed in the development of criteria values and the simplification of major chains in a network to prepare this method for use as an LCA tool.

Variations on this approach would include the possible environmental assessment trees that could be developed; these environmental assessment trees could vary in the choice of impact criteria as well as the way in which they are organized in the assessment tree.

**Hazard Ranking.** The Hazard Ranking approach prioritizes the hazards associated with the pollutants resulting from a product type, process, or activity. The list of pollutants and the quantities required for the hazard ranking process can be obtained directly from the inventory. The methodology encompasses several steps, including data collection on energy and material inputs and outputs, determining the impacts attributable to these inputs and outputs, and ranking the impacts. At its simplest level, a Hazard Ranking approach can be based on two indicators of toxicity:

- **Reference Dose** — an indicator of chronic toxicity which is the estimate of maximum daily oral exposure that does not cause harm.
- **Cancer Potency Factor** — an indicator of carcinogenicity which is an indication of the strength of the cancer-causing potential of a chemical.

A hazard value, such as "low," "medium," or "high," can be assigned to each pollutant based on chronic toxicity, cancer potency, or similar indicator. These values can be referenced to baseline health benchmarks such as the Occupational Safety and Health Administration's Permissible Exposure Limits (PELs). One then infers a relationship between the pollutants (using the PELs, for example) in order to rank the hazards of the compounds.

The hazard ranking value associated with the quantity of a pollutant may be based on actual toxicity or carcinogenicity data or it may be a subjective value based on available information. The method focus is primarily from the human-health standpoint. The
General Background

approach requires a substantial amount of time to develop and validate the category boundaries, i.e., what constitutes high, medium, and low within the range of possible hazard values. However, once a framework has been established, application of the method is rapid where data exist. Use of the method is limited for categories of pollutants whose toxicities have not been determined, and the approach assumes that all health endpoints are equal in severity.

Non-health impacts are not currently included the in ranking procedures, and thus broad environmental impacts such as socioeconomic or ecological consequences are not addressed. However, the Hazard Ranking approach need not be used only to assess human health effects. It can be applied to environmental toxicology or any measure where unit toxicity or potency can be defined.

A variation of the basic Hazard Ranking Approach has been used by Jacobs Engineering in conjunction with the Environmental Defense Fund for evaluating impacts of cleaning agents. This approach ranks the impacts for two alternatives relative to one another by assigning plus, minus, or zero values to the various hazards.

The Swiss method (Schmidt, 1991) discussed above also has elements of the Hazard Ranking Approach. An improved version of this method associates damages with the environmental loading through the use of an Ecofactor Equation:

\[
\text{Ecopoints} = \frac{1}{F_k} \times \frac{F}{F_k} \times C
\]

where, \(F_k\) = "Critical flux" which is the maximum loading at which the respective ecosystem does not show the adverse effect,

\[F = \text{current mass flux; the ratio of current to critical is termed "ecological scarcity"},\]

and

\[C = \text{dimensionless factor \(10^{12}\) to avoid large negative exponent values.}\]

Since all of the hazards are expressed as ecopoints, it is a simple matter to sum them.

Advantages of this version of the Hazard Ranking approach include:

- Air, water, and soil emissions, resource consumption, and certain non-chemical stressor impacts (e.g., radiation loadings) can be aggregated in a single index.

- The Ecofactor equation can be formulated to measure ecological scarcity in a nonlinear fashion if desired, i.e., greater weight can be given to penalizing loadings above the critical values.

- Regional as well as national or international spatial scales may be used.
Disadvantages of this version of the Hazard Ranking approach include:

- Ecofactors are not easily determined and obtaining approval by all parties will be difficult.
- Ecofactors need to be updated at regular intervals as new scientific knowledge is gained.
- Dispersion and degradation are not considered, although, in principle, they could be.
- The aggregation process reduces transparency and there may be an impression of more objectivity than is warranted.
- Health and occupational safety are not included in the current system.

Sima-Pro, a Dutch software package, is similar in methodology to the Swiss method but substitutes the relevant Dutch Threshold Limit Values and Acceptable Daily Intake Values in computing the hazard levels. However, in this variation, a single score is not calculated. Instead, the program uses the scores for each of the assessment elements individually to rank each product.

**Hazard Matrix.** The Hazard Matrix approach to impact analysis attempts to evaluate, without an explicit exposure analysis, the potential hazards to human health and the environment that result from environmental loadings and resource consumption. Estimates of these loadings and consumption are taken from the inventory analysis. Levels or types of effects would have to be agreed upon in advance of preparing the matrix. For example, an effect area might be human health. Within this area, there could be several subareas such as cancer, reproductive toxicity, and so on. The data from the inventory on environmental loadings from each LCA stage to various environmental compartments could be matched with potency information for any effect area. In the human health example, one could value the cancer potency of a chemical based on the EPA cancer unit risk values and weight-of-evidence. This information, coupled with information on the quantity of a carcinogen released to any medium, e.g. air, would yield a cancer hazard score. This process would be performed for each human health effect. Next, each of the human health effects scores (e.g. cancer, reproductive effects, and so on) would be summed to develop an overall human health score for the LCA within a single medium.

This process would be repeated for all media and all agreed upon effects areas. A simplified example of this matrix appears in Figure 5. The final result would be a matrix of media and effects for the product life cycle. Each cell within the matrix would have a representation of the magnitude of the "score" within the cell relative to those in other cells. This means that the contents of a score could be normalized and represented as either high, medium, or low relative to the other scores calculated in the same effects area.
The Hazard Matrix approach would be applied at each stage of the LCA and summed for the entire product. The matrices produced for each LCA stage would be useful to determine the appropriate area to focus improvement efforts. The overall product matrix would be valuable for comparisons between different products.

In addition, this approach would allow the application of weighting factors for each effect within an effects area. For example, cancer may be weighted higher than other human health effects in developing the human health effects score. The development of these weighting factors would be largely a subjective exercise and could be mediated by an expert panel. This method, however, as currently construed, does not permit the relative weighting of different effects areas. For example, in the presentation of the matrix, no value judgments are made between effects to human health versus effects to the environment.

The Hazard Matrix approach is simple to use in that it does not require the evaluation of exposure. The approach assumes that the combination of high potency and high loadings is proportionate to a high probability of effect (i.e., hazard). However, the lack of an explicit consideration of exposure precludes the possible identification of a cause-and-effect
relationship. It is possible that the loadings could be spread over a considerable time and distance and therefore not result in an actual impact. Therefore, this approach is not intended to convey specific estimates of risk, and in some cases may over- or under-estimate the true risk of an effect. This approach, however, does provide an objective valuation of the hazard of various effects and, as such, can be used to differentiate between the environmental and health impacts in the context of an LCA.

Several European methods are examples of hazard matrix systems. The EPS system (Volvo Method), discussed previously, is based on an aggregation of four impact elements: resources, scarcity, health, and environment. For consumption of natural resources, three factors are used: amount consumed, known resources, and an assessment factor to account for scarcity. For the effect indices, a combination of factors are used: intensity or frequency of effect, distribution of effect, contribution of product system to total effect, costs for decreasing the effect by a unit weight, and an assessment factor. The scores for each impact element can be transformed into Environmental Load Units (ELU) that can be aggregated into one overall score. This method is an attempt to put environmental and health effects into one perspective.

The DTI (Danish Technological Institute) approach (Schmidt, 1991) has been used to compare alternative materials. This approach examines the following elements at each stage of the life cycle: resource consumption, worker exposure, occupational health effects, environmental exposure, environmental effects, and consequences of accidents (specifically fires). This approach differs from others in that it looks at upset conditions. The results are presented in a specific matrix combining lifecycle stages and environmental assessment elements for a given material, and in a relative matrix where alternatives are compared to one another. The approach does not use a detailed lifecycle inventory, but it does examine in depth the problem areas of the materials used.

Subjective/Perceptual/Monetarization Impact Analysis Family

This family of impact analysis methods is based on the different knowledge bases and levels of understandings that individuals and groups bring into the impact analysis process. Some of the methods involve the assignment of monetary values to the correction or avoidance of impacts.

Willingness to Pay. The Willingness to Pay impact approach is a subjective method that involves asking individuals directly how much they would be willing to pay for a product or service to improve or to prevent deterioration of public health or the environment. Estimating the values people place on these qualities presents a difficult problem primarily because these are not commodities that have market-determined prices. They are what economists call "public goods" -- things that provide value to individuals and to the community, but are not bought and sold. Government action provides some public goods (e.g., flood control and public health). Nature provides others, including air and water. Because these goods are not traded in markets, sophisticated methods are needed to infer their values.
One general approach for assigning values is to ask people what they would be willing to pay for the good. Contingent valuation (CV) is the most widely used technique for doing this. In the CV approach, a sample population is surveyed through personal or mail interviews and polled on how much they would be willing to pay. The responses, which are contingent on the description of the condition to be rectified, can then be analyzed. For example, a representative sample of the public may be surveyed to determine how much additional they would be willing to pay for electricity to have improved visibility obtained by decreasing SO$_2$ emissions from electric utilities.

There are numerous points where errors or flaws can enter the CV process. The analyst must define a hypothetical transaction that precisely defines the impact avoidance that people can buy and illustrate it in such a way that the respondents will understand. The transaction has to be framed in such a way that encourages the respondents to take the question seriously. Care must be taken to avoid the appearance of giving clues to people about what answer is expected. CV involves a multitude of research disciplines (e.g., economics, psychology, in addition to the technical area of expertise to which the valuation is addressed), and the results must be integrated into the various disciplines for interpretation.

The Consequences Network approach has several advantages. It is comprehensive, replicable, interdisciplinary, objective, based on explicitly defined criteria, able to assess total impact, able to detect environmentally sensitive areas, and thorough. It is thorough in that it accounts for interrelationships of the various effects resulting from project actions and sensitive system connections. The method uses explicitly defined criteria and procedures so that evaluation criteria and values are not arbitrarily assigned. However, a difficulty with this system is the time and effort required for such a thorough impact analysis. Another problem is the difficulty in implementing the approach without a developed environmental assessment tree. Additional work is needed in the development of criteria values and the simplification of major chains in a network to prepare this method for use as an LCA tool.

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General Background

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where, \( F_k \) = "Critical flux" which is the maximum loading at which the respective ecosystem does not show the adverse effect,

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Since all of the hazards are expressed as ecopoints, it is a simple matter to sum them.

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Disadvantages of this version of the Hazard Ranking approach include:

- Ecofactors are not easily determined and obtaining approval by all parties will be difficult.
- Ecofactors need to be updated at regular intervals as new scientific knowledge is gained.
- Dispersion and degradation are not considered, although, in principle, they could be.
- The aggregation process reduces transparency and there may be an impression of more objectivity than is warranted.

Combinations of Impact Analysis Methods

Selection of a methodology for a specific study may be influenced by the fact that the choice of analysis methods is not mutually exclusive. An analyst could use two or more methods in combination to obtain the best mix of advantages and disadvantages. For example, one could combine an index-based approach for global effects (global warming potential, ozone depletion potential), because the index value is easy to determine and is easily understood, with an exposure-based toxicity approach, if intrinsic toxicity indices do not adequately capture the impact potential of the product. In addition, as long as careful explanation of the approach is provided, it could be possible to mix methods addressing the same general consequences category. For example, one could use a mixture of checklists, hazard indices, and consequences networks to quantify effects of heavy metal releases from a product to humans.

Two of the impact assessment applications discussed by Schmidt (1991) at the SETAC-Europe Workshop on LCA involved combinations of two of the generic methods described earlier in this document. The Swiss impact analysis method includes components of both the Resource Consumption Ratio and Hazard Ranking approach. The EPS system includes
components of both the Hazard Matrix and the Resource Consumption Ratio approaches
discussed previously.

This paper has attempted to identify issues for the participants to discuss. These include:
(1) identification of strategies for the impact analysis process, (2) clarification of applications
of impact analysis, (3) development of impact measures, (4) development of a
methodological framework for assessing impacts, and (5) validation of criteria to be used
in selecting impact analysis methods.

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MULTI-ATTRIBUTE UTILITY THEORY (MAUT)

For each attribute, a utility function is estimated. This is accomplished through a set of questions posed to the decision maker concerning attitude toward risk. The questions are expressed in terms of simple hypothetical lotteries, and the responses are checked for consistency. An equation (the utility function) can then be fitted to the responses, relating utility to the value of an attribute. Once all of the individual utility functions are estimated, they are combined.

As a brief example, Figure C-1 identifies environmental and other objectives and measurable/meaningful attributes for water pollution effects. The analyst must first establish that the attributes are pairwise preferentially independent and utility independent through a series of questions posed to the decision maker. Assuming that independence is
established, the analyst must define the individual utility functions. For example, assume that the attribute for ecosystem effect is the results of effluent toxicity testing. The following dialogue might lead to the development of the needed utility function:

Analyst: What is the reasonable range of possible outcomes of effluent toxicity testing? (These are assigned utilities (U) of one and zero, respectively.)

Decisionmaker: I would say that 10% and 75% are reasonable values.

Analyst: Okay, now at what value of this proxy attribute would you be indifferent to a 50-50 lottery involving the extreme values or achieving the chosen value with certainty? (The chosen value is called a "certainty equivalent.")

Decisionmaker: Probably 30%.

Analyst: That is the value assigned a utility of 0.5 (U=0.5).

A question similar to the second question posed by the analyst is asked to determine the certainty equivalent to two other 50-50 lotteries, one to find the certainty equivalent for U = 0.75 and one for U = 0.25. These five points (at U = 0, 0.25, 0.5, 0.75, and 1.0) begin to describe the utility function. After checks for consistency, the analyst could express the utility function mathematically, perhaps using regression analysis. The utility functions are then combined into a single multi-attribute function (one multi-attribute utility function per decisionmaker).

To combine the single attribute utility functions, the analyst begins by posing questions to the decisionmaker concerning preference between or among attributes. For example, Keeney and Raiffa (1976) advocate a procedure involving the evaluation of pairwise tradeoffs between any two attributes while all other attributes are held fixed. This leads to scaling constants that are then used in either an additive or multiplicative multi-attribute utility function. It is important that the weights or scaling constants be checked for consistency (see Keeney and Raiffa) before the multi-attribute utility function is applied.
APPENDIX D

STEPS OF THE ANALYTIC HIERARCHY PROCESS

Below are the steps to follow in using the Analytic Hierarchy Process (AHP). Particular steps may be more emphasized in some situations than in others, and as noted, interaction is generally necessary.

1. Define the problem and determine what you want to know.

2. Structure the hierarchy from the top (the objectives from a general viewpoint) through the intermediate levels (criteria on which subsequent levels depend) to the lowest level (which usually is a list of the alternatives).

3. Construct a set of pairwise comparison matrices for each of the lower levels — one matrix for each element in the level immediately above. An element in the higher level is said to be a governing element for an element(s) in the lower level since it contributes to it or affects it. In a complete simple hierarchy, every element in the lower level affects every element in the upper level. The elements in the lower level are then compared to each other based on their effect on the governing element above. This yields a square matrix of judgments. The pairwise comparisons are done in terms of which element dominates another. These judgments are then expressed as integers (see table for judgment values). If element A dominates element B, then the whole number integer (or exact value with decimals if it is known) is centered in row A, column B and the reciprocal (fraction) is entered in row B, column A. Of course, if element B dominates element A, then the reverse occurs. The whole number is then placed in the B, A position with the reciprocal automatically being assigned to the A, B position. If the elements being compared are equal, a one is assigned to both positions.

4. There are n(n-1)/2 judgments required to develop the set of matrices in step 3. (Remember, reciprocals are automatically assigned in each pairwise comparison.)

5. Having made all the pairwise comparisons and entered data, the consistency is determined using the eigenvalue. \( Aw = \lambda_{max} w \) is determined. The consistency index then using the departure of \( \lambda_{max} \) from n compared with corresponding average values for random entries yielding the consistency ration CR).

6. Steps 3, 4, and 5 are performed for all levels and clusters in the hierarchy.
Hierarchical composition is now used to weight the eigenvectors by the weights of the criteria and the sum is taken over all weighted eigenvector entries corresponding to those in the next lower level of the hierarchy.

The consistency of the entire hierarchy is found by multiplying each consistency index by the priority of the corresponding criterion and adding them together. The result is then divided by the same type of expression using the random corresponding to the dimensions of each matrix weighted by the priorities as before. Note first that the CR should be about 10 percent or less to be acceptable. If not, the quality of the judgments should be improved, perhaps by revising the manner in which questions are asked in making the pairwise comparisons. If this should fail to improve consistency, then it is likely that the problem should be more accurately structured; that is, grouping similar elements under more meaningful criteria. A return to step 2 would be required, although only the problematic parts of the hierarchy may need revision.

To perform absolute measurement in order to preserve the rank of the alternatives to satisfy expectations and prior commitments, each lowest level subcriterion is divided into a complete set of intensities so that an alternative always reflects one of these intensities. Then the intensities are pairwise compared according to perceived importance or priority with respect to that criterion. Finally, the alternatives are rated one at a time, the intensities for each criterion and the ratings weighted and added to obtain its overall rank on a ratio scale. Unlike paired comparisons, this process requires expert knowledge to rate the intensities. In most decision problems about the future, there is no such expert knowledge. Also experts have been known to be biased and misjudge the importance of the intensities. In that case, paired comparisons must be used.
APPENDIX E

ADDITIONAL INFORMATION ON AHP EXAMPLES USED IN CHAPTER 6 (TEXT BOX 4)

By taking the priority of the intensity each of the two alternatives \( \frac{1}{2} \) full (V) and full (F) shown next to each intensity, weighting it by the importance of its criterion, and then adding over the criteria, we obtain the following overall ratio scale weights:

\[
\begin{align*}
\frac{1}{2} &= .417 \quad \text{normalized to .594} \\
\text{Full} &= .285 \quad \text{normalized to .406}
\end{align*}
\]

In this case, absolute ranking of the alternative gives the same outcome as relative ranking.

Assume that the foregoing decision were to be expanded by demanding that an alternative, relatively empty dam, be considered as yet another possibility for a dam that has been around for sometime. It can happen that the mere presence of this alternative in the comparisons, keeping the judgments on the others to be the same would cause the overall ranking of the other two possibilities to change. In other words for some decision problems the number and the quality of alternatives considered can change the ranking. There are many counterexamples offered in the literature by people who work in utility theory which show that a condition of regularity assumed in that theory is often violated. There, as in absolute measurement, alternatives are always rated one at a time and the ranking of one of them cannot affect the ranking of the others and the old rank, is never changed. When alternatives are ranked one at a time, their overall weights are determined by the absolute weight of the alternative (a scale with a unit) for each criterion, multiplied by the weight of the corresponding criterion and added over the criteria. Since their absolute weights are not changed, if a new alternative is added, the only way that their overall weights can change is if the weights of the criteria or the number of criteria is changed.

The absolute mode of the AHP also does this when, to honor commitments, we wish that there should be no change in rank to honor prior commitments. A classic example where the number of alternatives matters has been offered by Corbin and Marley. A lady goes to buy a hat and finds two hats A and B, preferring the style of A over B but finding them to be equally unique on the basis of her previous knowledge about hats, she selects A but changes her mind and buys B when she finds another copy of A. In this problem no new criterion is introduced and the importance of style and uniqueness are well established in her mind from frequent shopping for hats. There are other examples which occur in the field of marketing where the presence of an irrelevant "phantom" alternative C (a less preferred one) could cause rank reversal of more preferred alternative. These and many other examples are real life occurrences that are totally unacceptable in utility theory, but they do happen both in theory and in practice. How can an important theory let such examples that violate its assumptions go unanswered? There are times when we want to
preserve rank particularly when we have expectation and need a way to do so and times when we must allow it to reverse and need a way to do so. We cannot legislate such rank reversal away by an axiom. In relative, measurement, the weight on an alternative involves the relative, not absolute weight of the alternative for all the criteria. The normalization factor involves the measurement of all the alternatives and their number for each criterion. Any change in the alternative can change the rank and cause rank reversal.
### Additional Information on AHP Examples

Used In Chapter 6 (Text Box 4)

<table>
<thead>
<tr>
<th>Eco. Hum. R.D.</th>
<th>Priority</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ecological</td>
<td>1 1/3 3</td>
</tr>
<tr>
<td>Human</td>
<td>3 1 5</td>
</tr>
<tr>
<td>Resources Depletion</td>
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**Inconsistency = .037**

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<td>M/P</td>
</tr>
<tr>
<td>Risk of Disaster</td>
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</tr>
<tr>
<td>Mental/Physical</td>
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**Inconsistency = .000**

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</tr>
<tr>
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<td>3 1 3</td>
</tr>
<tr>
<td>Vegetable Species</td>
<td>1 1/3 1</td>
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</tbody>
</table>

**Inconsistency = .000**

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<tr>
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<td>Priority</td>
</tr>
<tr>
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</tr>
<tr>
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**Inconsistency = .062**

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<td>Physical Factors</td>
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**Inconsistency = .000**

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<tr>
<td>Ability for Flood Control</td>
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**Inconsistency = .000**
### Additional Information on AHP Examples Used In Chapter 6 (Text Box 4)

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<tr>
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<td>0.750</td>
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<td>0.833</td>
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<tr>
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<td>1</td>
<td>0.167</td>
</tr>
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<th>Priority</th>
</tr>
</thead>
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<td>1</td>
<td>5</td>
<td>0.833</td>
</tr>
<tr>
<td>Full</td>
<td>1/5</td>
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<td>0.167</td>
</tr>
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<tr>
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<td>1</td>
<td>1</td>
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<tr>
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<table>
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<tr>
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<tr>
<td>1/2</td>
<td>1</td>
<td>5</td>
<td>0.833</td>
</tr>
<tr>
<td>Full</td>
<td>1/5</td>
<td>1</td>
<td>0.167</td>
</tr>
<tr>
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<table>
<thead>
<tr>
<th>Soil Erosion</th>
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<th>Priority</th>
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</thead>
<tbody>
<tr>
<td>1/2</td>
<td>1</td>
<td>3</td>
<td>0.750</td>
</tr>
<tr>
<td>Full</td>
<td>1/3</td>
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<td>0.250</td>
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<table>
<thead>
<tr>
<th>Collapse of Dam</th>
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<th>Priority</th>
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</thead>
<tbody>
<tr>
<td>1/2</td>
<td>1</td>
<td>7</td>
<td>0.875</td>
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<td>1/2</td>
<td>1</td>
<td>1/5</td>
<td>0.167</td>
</tr>
<tr>
<td>Full</td>
<td>5</td>
<td>1</td>
<td>0.833</td>
</tr>
<tr>
<td><strong>Inconsistency</strong></td>
<td><strong>0.000</strong></td>
<td></td>
<td></td>
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</tbody>
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<table>
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<tr>
<th>Pollution Reduction</th>
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<th>Full</th>
<th>Priority</th>
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</thead>
<tbody>
<tr>
<td>1/2</td>
<td>1</td>
<td>1/5</td>
<td>0.167</td>
</tr>
<tr>
<td>Full</td>
<td>5</td>
<td>1</td>
<td>0.833</td>
</tr>
<tr>
<td><strong>Inconsistency</strong></td>
<td><strong>0.000</strong></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
To obtain the overall priority of an alternative we come down the hierarchy from the top, weight the priorities of the subcategories by the priority of their parent category and then use this result to further weight the priorities of the criteria and use these in turn to weight the priorities of the two alternatives at the bottom and add over all the criteria. We have:

\[
\begin{align*}
1/2 & \text{ Full } = 0.655 \\
\text{ Full } & = 0.345 \\
\text{ overall inconsistency } & = 0.03
\end{align*}
\]

The following table shows how these final priorities would change with a change in the relative priorities of the 1st level categories:

<table>
<thead>
<tr>
<th>Priorities of Categories</th>
<th>Priorities of Outcomes</th>
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<tbody>
<tr>
<td>Ecological</td>
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</tr>
<tr>
<td>.198</td>
<td>.689</td>
</tr>
<tr>
<td>.258</td>
<td>.637</td>
</tr>
<tr>
<td>.358</td>
<td>.551</td>
</tr>
<tr>
<td>.718</td>
<td>.242</td>
</tr>
<tr>
<td>.301</td>
<td>.682</td>
</tr>
<tr>
<td>.179</td>
<td>.405</td>
</tr>
<tr>
<td>.087</td>
<td>.196</td>
</tr>
<tr>
<td>Aesth. Quality</td>
<td>E</td>
</tr>
<tr>
<td>----------------</td>
<td>---</td>
</tr>
<tr>
<td>(F) Exc.</td>
<td>1</td>
</tr>
<tr>
<td>(¾) V.G.</td>
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</tr>
<tr>
<td>Good</td>
<td></td>
</tr>
<tr>
<td>Fair</td>
<td></td>
</tr>
<tr>
<td>Poor</td>
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Inconsistency = .065

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<tr>
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<th>Btr</th>
<th>Mts</th>
<th>Bel</th>
<th>Priority</th>
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<tbody>
<tr>
<td>Better</td>
<td>1</td>
<td>3</td>
<td>7</td>
<td>.649</td>
</tr>
<tr>
<td>(¼,F) Meets</td>
<td>1</td>
<td>5</td>
<td></td>
<td>.279</td>
</tr>
<tr>
<td>Below</td>
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</table>

Inconsistency = .062

<table>
<thead>
<tr>
<th>Dam Collapse</th>
<th>VL</th>
<th>L</th>
<th>U</th>
<th>Priority</th>
</tr>
</thead>
<tbody>
<tr>
<td>V.Likely</td>
<td>1</td>
<td>1/5</td>
<td>1/7</td>
<td>.072</td>
</tr>
<tr>
<td>(F) Likely</td>
<td>1</td>
<td>1/3</td>
<td></td>
<td>.279</td>
</tr>
<tr>
<td>(¾) Unlikely</td>
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Inconsistency = .062

<table>
<thead>
<tr>
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<th>Ex</th>
<th>Eq</th>
<th>LT</th>
<th>Priority</th>
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</thead>
<tbody>
<tr>
<td>(F) Exceeds</td>
<td>1</td>
<td>3</td>
<td>7</td>
<td>.649</td>
</tr>
<tr>
<td>(¾) Equal</td>
<td>1</td>
<td>5</td>
<td></td>
<td>.279</td>
</tr>
<tr>
<td>LessThan</td>
<td>1</td>
<td></td>
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Inconsistency = .062
<table>
<thead>
<tr>
<th>Soil Erosion</th>
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<th>I</th>
<th>Priority</th>
</tr>
</thead>
<tbody>
<tr>
<td>Signif.</td>
<td>1</td>
<td>1/5</td>
<td>1/7</td>
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Inconsistency = .062

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<th>Bel</th>
<th>Priority</th>
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<td>.649</td>
</tr>
<tr>
<td>(½) Meets</td>
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<td>5</td>
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<td>.279</td>
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<tr>
<td>Below</td>
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Inconsistency = .062

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<td>7</td>
<td>.649</td>
</tr>
<tr>
<td>(F) 100</td>
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<td></td>
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<tr>
<td>No</td>
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Inconsistency = .062

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<th>Con</th>
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<tr>
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Inconsistency = .000
REFERENCES


