Chapter 6
Strengths and Limitations of Life Cycle Assessment

Mary Ann Curran

Abstract This chapter discusses strengths and limitations of Life Cycle Assessment (LCA) not by linear analysis but by elucidating limitations embedded in strengths. It elaborates perceived and real limitations in LCA methodology grouped by research need, inherent characteristic or modeling choice. So, LCA practice continues to suffer from variations in practice that can result in different LCA results. Some limitations, such as modeling missing impact indicators and making life cycle inventory more readily-available, will be addressed through continued research and development of the tool. Other modeling choice-related limitations, such as matching goal to approach setting a proper functional unit or appropriately scoping the assessment, need to be addressed through continued education and training to assist users in the proper application of the tool. Still other limitations in LCA practice would benefit by the development of harmonized guidance and global agreement by LCA practitioners and modelers.

However, despite these variations, LCA offers a strong environmental tool in the way toward sustainability.

Keywords Attributional modeling · Consequential modeling · Data uncertainty · Decision making · Functional unit · Goal and scope definition · ISO series of standards 14000 · LCA · Life cycle assessment · Life cycle impact assessment · Life cycle inventories · Life cycle sustainability assessment · Life cycle thinking · Midpoint impact categories · Modeling · Normalisation · Risk assessment · Scale · System boundaries · System expansion

1 Introduction

The last few decades have seen a marked rise in the application of life cycle assessments in virtually all countries around the world. This growing interest can be attributed to the powerful support the tool provides to decision makers. To date,

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M. A. Curran (✉)
BAMAC Ltd., 5137 Oak Brook Drive,
Cincinnati, Ohio 45244, USA
e-mail: macurran@cinci.rr.com

W. Klöpffer (ed.), *Background and Future Prospects in Life Cycle Assessment,*
LCA Compendium – The Complete World of Life Cycle Assessment,
DOI 10.1007/978-94-017-8697-3_6,

Life Cycle Assessment (LCA) is a method defined by the international standards ISO 14040 and 14044 to analyse environmental aspects and impacts of product systems. In the introduction to the International Standard ISO 14040, serving as a framework, LCA is defined as follows:

> LCA studies the environmental aspects and potential impacts throughout a product´s life (i.e. cradle-to-grave) from raw material acquisition through production, use and disposal. The general categories of environmental impacts needing consideration include resource use, human health, and ecological consequences.

A similar definition of LCA was adopted as early as 1993 by the Society of Environmental Toxicology and Chemistry (SETAC) in the 'Code of Practice' document (SETAC 1993). Similar definitions can be found elsewhere. A consequence of those deliberate limitations to the analysis and interpretation of **environmental impacts** was the creation of a method that is restricted to only quantifying the **ecological** aspect of sustainability. The exclusion of economical and social factors was a deliberate choice intended to avoid method overload, while being well aware that any decision in the development of sustainable products, etc., cannot and must not neglect these factors (Klöpffer and Grahl 2014).

Among the many strengths of LCA are the following:

- *LCA is a comprehensive assessment*
  LCA is a cradle-to-grave analytical tool that captures the overall environmental impacts of all the life cycle stages associated with a product, process or human activity from raw material acquisition, through production and use phases, to waste management. This comprehensive view makes LCA a unique approach in the suite of environmental management tools available to decision makers. Without **life cycle thinking**[1], we risk focusing on the environmental issues that demand our immediate attention, and ignoring or devaluing issues that may occur either in another place or in another form (impact). Such focused assessments can lead to decisions that are based on incomplete information.
- *LCA highlights potential environmental tradeoffs*
  The broad scope involved in conducting LCA makes users more aware of the complexities of integrated industrial systems and ecosystems, and the appropriate corresponding remedy for a given situation. LCA encompasses all the interacting activities, media, and impacts and the identification of potential tradeoffs from one phase of the life-cycle to another, from one region to another, or from one environmental problem to another that may occur as a result of a decision (that is, resulting from a change to a system or from choosing between systems).
- *LCA provides structure to an investigation*
  The ISO series of standards developed in the 1990s provides us with a definition of LCA along with a general framework for conducting an assessment in four inter-related phases (goal and scope, inventory analysis, impact assessment,

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[1] Life cycle thinking is a fundamental prerequisite towards understanding impact mechanisms along value chains in complex product or production systems. It is the indispensable approach to support sustainable development (De Schrynmakers 2009).

interpretation) (ISO 2006). LCA has developed into an important tool to capture information for analysis, discussion, actions and regulation in a variety of areas (Ngo 2012). LCA also assists decision makers in recognising when they intentionally or unintentionally place high value on some environmental aspects and little or no value on others.

- *LCA can challenge conventional wisdom*
  The most important aspect of LCA is that it helps people incorporate whole-system thinking in terms of impact assessment. In getting away from the disconnected, stove-piped way of thinking that has led us to where we are today, LCA can bring to light data and information that makes us question what is commonly held as environmentally preferable (Ngo 2012). Bio-based materials and products, for example, have long been given preferred status. Only more recently with the reporting of LCA studies have degraded quality of water and soil resulting from biofeedstock production been brought into the discussion (von Blottnitz and Curran 2006).
- *LCA advances the knowledge base*
  Taking into account the full and complete analysis of a system's environmental impacts is likely a more complicated (i.e. costly) endeavor than many organizations are willing to undertake. It is anticipated that the continued conduct of LCAs will make organisations and consumers more aware of the interconnections of operations, while providing producers, consumers and regulators with the necessary baseline information and data to move forward (Ngo 2012). The challenge now is to find an affordable, efficient way to share this growing database of knowledge with users across the globe.
- *LCA fosters communication and discourse*
  The LCA methodology, originally developed to provide environmental information for distinguishing between products or between services, has evolved as a basis to communicate the overall environmental performance of products and processes to stakeholders. For example, developing environmental product declarations (EPDs) based on LCA is an effective way to communicate credible information about the environmental performance of products (Del Borghi 2012).

2 Strengths and Limitations—Perceived and Real—in Life Cycle Assessment

As with all complex assessment tools, the LCA methodology has its limitations as well as strengths. Although the ISO standard gives a consensus definition for LCA and provides a general framework for conducting an assessment, it leaves much to interpretation by the person conducting the assessment[2]. As a result, LCA studies

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[2] ISO 14040 did not intend from the beginning to standardize LCA methods: "there is no single method for conducting LCA" (Heijungs and Guinée 2012).

have been criticized for producing different results for seemingly the same product. The vagueness of the ISO standard along with a growing desire to follow a ‘life cycle approach’ with no clear definition of what that means, has led to confusion regarding what LCA can and cannot do, and how it fits within a strategic level approach to sustainability.

Furthermore, an aspect that is simply a characteristic of LCA methodology may be perceived as a limitation if it does not fulfil the user’s immediate need. For example, the present-day LCA framework does not take social welfare into consideration. Someone who is interested in understanding the social aspects of a product is recommended to apply some other tool or approach to gather information pertinent to the social (and economic) dimensions[3]. This is sometimes perceived as a missing element, or a limitation, in LCA. But it may also be viewed as an unrealistic expectation of what LCA is intended to do.

Some limitations are temporary in that the methodology could be clarified through further research and development to improve understanding of the issue and develop clear guidance. Other limitations are inherent in the design of LCA methodology and how it was intended to be conducted. Other limitations occur during application when the modeler has alternative approaches from which to choose, leading to widely varying results from case to case. In these instances, there is no ’right’ way and how to approach these modeling choices is often hotly debated. LCA practice would benefit by the development of harmonized guidance and global agreement by LCA practitioners and modelers (UNEP 2011; UNEP/SETAC 2011).

Table 6.1 lists examples of LCA limitations by the three types: (1) can be improved through research; (2) inherent in the methodology; and (3) alternate modeling choices. The following sections describe these limitations in more detail.

2.1 Matching the Goal of the Assessment to the Approach

Not long after the 1990 SETAC workshop[4], which laid the foundation for current LCA practice, it was realized that a very important aspect had been overlooked, i.e. setting the goal for the study at the outset of the effort. Subsequent versions of the phases of an LCA in ISO included an initial ‘goal and scope definition’ phase (Fig. 6.1) (see Chap. 2[5] of this volume).

A clearly stated goal will make defining the study scope and data collection a little easier. For example, a study with a goal to examine bio-ethanol as an automo-

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[3] Efforts to develop a Life Cycle Sustainability Assessment approach rose from the perceived need to broaden the scope of LCA from mainly environmental impacts to covering all three dimensions of sustainability (people, planet and prosperity) (CALCAS 2009). However, this broadening is at variance with ISO’s explicit restriction to environmental issues (Heijungs and Guinée 2012).

[4] A Technical Framework for Life-Cycle Assessment. August 18–23, 1990, Smugglers Notch, Vermont.

[5] The role of the Society of Environmental Toxicology and Chemistry in life cycle assessment development and application by James Fava et al.

Table 6.1 Examples of limitations in LCA methodology grouped by research need, inherent characteristic or modeling choice

Research and Development to Improve LCA
Matching the goal of the assessment to the approach
Gathering the inventory data can be very resource and time intensive
Missing impact data and models for Life Cycle Impact Assessment
Dealing with life cycle inventory and impact data uncertainty
Inherent Characteristics in LCA Methodology
Distinguishing between Life Cycle Impact Assessment and Risk Assessment
LCA Does not always (usually) declare a 'winner'
LCA results should be supplemented by other tools in decision making
Choices Available to the Modeler
Allocating environmental burdens across co-products
Assigning credit for avoided burden
Expanding the boundaries (Consequential LCA)

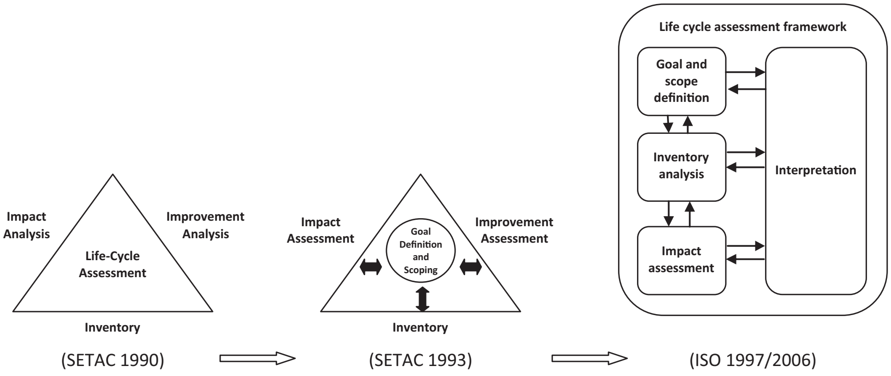

Fig. 6.1 Evolution of the LCA framework

tive fuel would lead to scoping the system around the manufacture and use of the fuel (excluding the manufacture, maintenance and end of life issues of the car itself). However, these results would then not be applicable in a comparison of, say, a car with an internal combustion engine to an all-electric vehicle, since components of the vehicle may differ (especially weight and fuel efficiency).

Although goal definition is recognized as an integral step in LCA methodology, clear guidance for matching the goal with the subsequent phases of scoping, inventory analysis and impact assessment is still lacking.

Connected with goal setting is the selection of a 'functional unit', a unique feature of LCA which sets it apart from other environmental assessment approaches. The functional unit is defined by the service provided by the system being studied. It is further shaped by the goal of the study in that it forms the basis for the study to answer the question or address the concern at hand.

Table 6.2 Functional unit versus reference flow (ISO 2012)

A functional unit is a quantified description of the performance of the product systems	Example: Lighting 10 m^2 with 3000 lx for 50,000 h with daylight spectrum at 5600 K
A reference flow is a quantified amount of manufactured product necessary for a specific product system to deliver the performance described by the functional unit	Example: 15 daylight bulbs of 10,000 lm with a lifetime of 10,000 h

At times, published LCAs report the reference flow as the functional unit and use it as the starting point for building a model of the product system; however, these two terms should not be confused. The functional unit reflects the performance or the service being fulfilled by the product system. The reference flow, then, translates the functional unit into specific product flows from the processes within the industrial system, setting the basis for calculating the inventory data (Table 6.2).

The importance of setting the appropriate scale to the functional unit was discussed early on in LCA development (e.g. Guinée et al. 2002). Often, the functional unit is set at a rather small amount; thus, the LCIA has to operate on mass loads representing a small share (often nearly infinitesimal) of the full emission output from the processes (Finnveden et al. 2009). For example, a biofuel LCA may have the functional unit of the amount of fuel to operate a single car over one year. This would require a reference flow of a small amount of biofeedstock input. The resulting impacts from the acquisition of the biofeedstock, compared to a national production level, would most likely appear insignificant, even though the potential impacts from the agricultural sector, e.g., eutrophication, land use change, soil quality, etc., may be an important consideration (Notarnicola et al. 2012). Setting the functional unit at a larger scale, such yearly production, may simplify the normalization step by giving realistic numbers for a country or an economic unit.

2.2 *Gathering the Inventory Data can be Very Resource and Time Intensive*

Although LCA databases and software have become more widely available in recent years, the lack of readily available inventory data continues to be a major hurdle for LCA practice. Inventory data can be created by collecting primary data directly from the sources, such as material and product manufacturers. More often data are collected from secondary sources such as reports, publications and databases. Data are held either privately, such as in LCA practitioners' software, or in the public domain, such as government sources. Commercial tools are usually fairly simple to use, although some training may be needed before the user is adept at using them. There is usually a subscription or purchase fee associated with these products.

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

Another option for creating life cycle inventories is the use of publicly-available databases. These databases are often government-sponsored, such as the US EPA's Toxic Release Inventory (TRI) and Australia's National Pollutant Inventory (NPI). They are easily accessible and available at no cost. But these sources do not lend themselves easily to use in most life cycle studies because the data are reported for individual sites or facilities and not as industry averages for a country or a region. Often assumptions have to be made about the data in order to aggregate them to represent an industry sector. Also, data are not allocated by production; therefore, additional information is needed in order to determine releases per product. To achieve this, the most effective way to simplify the LCA process is to increase the collection, publication, and standardisation of LCI data. For example, the Europeans have been successful in creating publicly-available databases through efforts such as the ecoinvent database and more recently the European Commission's Platform on Life Cycle Assessment. The US has seen limited success in creating a national inventory database (US LCI Database 2012)[6]. As mentioned earlier, it is anticipated that the continued conduct of LCAs will lead to increased generation of baseline information and data. Participation by producers, suppliers, LCA practitioners and commissioners of LCAs, in the active sharing of raw data that are collected and transformed into useful LCI data will go a long way in expanding available foreground data into the supply chain. An affordable, efficient way to share this growing database needs to be established and fully developed for public accessibility.

2.3 Missing Impact Data and Models for LCIA

The life cycle impact assessment (LCIA) phase is intended to provide additional information to help assess the inventory results. To do this, data that link emissions and extractions to impact categories indicators are needed. The global level models related to global warming and ozone depletion have strong agreement by LCA modelers. Other impact models are still in their infancy and in need of further development, such as water use, land use, and in addressing issues such as spatial and temporal differentiation (Margni and Curran 2012). While both abiotic and biotic resources are generally considered to be equally important, modeling biotic resource use has not received as much attention (Finnveden et al. 2009).

Further yet, some impact data are yet to be generated and made publicly-available. For example, impact data for human and ecosystem health exposure to nanoproducts (products that contain a nanocomponent or produced using nanotechnol-

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[6] See the 'US LCI Database Project—Review Panel Report on the Development Guidelines' from January 2004 (www.nrel.gov/lci/pdfs/34275.pdf). NREL is a national laboratory of the U.S. Department of Energy, Office of Energy Efficiency and Renewable Energy operated by the Alliance for Sustainable Energy, LLC.

ogy) are still insufficient. Another example involves modeling the management of nuclear waste from nuclear power generation. In both cases, current LCIA models cannot fully model the inventory data for these systems; the modeler runs the risk of dropping important inventory data if it they are not otherwise retained and reported in the final analysis.

Currently, there is no one single impact assessment methodology being used by practitioners. Nevertheless, commonalities can be seen in LCA practice regarding the impact categories that are being selected for modeling. Table 6.3 lists midpoint impact categories that are being used by prominent researchers in their LCIA models[7].

2.4 Dealing with Data Uncertainty

Uncertainty analysis is the process of determining the variability of the data and the impact on the final results. It applies to both the inventory data and the impact assessment indicators and can be attributed to both errors and normal fluctuations in the data. While data variability can have a great impact on how the results are used in decision-making, the actual influence of uncertainty on decision-making has not been adequately studied. Furthermore, many LCAs are produced without reporting the uncertainty of the data. There is a need to understand the consequences of these decisions for proper transparency in the study.

Research efforts are needed to establish recommended practice for uncertainty analysis and to elaborate guidance for practitioners and method developers on how to estimate, communicate, interpret and manage uncertainty in both LCI and LCIA (Margni and Curran 2012).

2.5 Distinguishing between Life Cycle Impact Assessment and Risk Assessment

It is important to understand the difference between Life Cycle Impact Assessment (LCIA) methodology and traditional Risk Assessment (US EPA 2004). The general approach to risk assessment is a complex process, requiring the integration of data and information across a broad range of activities and disciplines, including source characterisation, fate and transport, modeling, exposure assessment, and dose-response assessment. On the other hand, in an LCA the product system is extended in space and time, and the emission inventory is often aggregated in a form which restricts knowledge about the geographical location of the individual emissions. The

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[7] While midpoint modeling is most common in LCA practice, some methods model past the midpoint to the endpoint level (e.g., from an ozone depletion indicator to increased incidents of skin cancer). These damage models can be reported in units of Disability Adjusted Life Years (DALYs), an aggregation of environmental impacts, monetary value, or other aggregated damage units.

Table 6.3 Midpoint Impact Categories included in ReCiPe, IMPACT World+, TRACI and LIME (Margni and Curran 2012)

ReCiPe		IMPACT World+	TRACI	LIME
http://www.lcia-recipe.net/		http://www.impactworldplus.org	http://www.epa.gov/nrmrl/std/traci/traci.html	http://www.jemai.or.jp/lcaforum/index.cfm
INPUTS	Mineral Resource Depletion Fossil Fuel Depletion	Resource Use	Resource Depletion Fossil Fuel Use	Resource Consumption
	Agricultural Land Occupation Urban Land Occupation Natural Land Transformation	Land Use	Habitat/T&E Species	Land Use
	Water Depletion	Water Use	Water Use	
OUTPUTS	Climate Change	Global Warming	Global Warming	Global Warming
	Ozone Depletion	Ozone Layer Depletion	Ozone Depletion	Ozone Layer Depletion
	Particulate Matter Formation			Urban Air Pollution
	Human Toxicity	Human Toxicity	Human Health – Cancer – Non Cancer – Criteria Pollutants	Human Toxicity
	Terrestrial Ecotoxicity Freshwater Ecotoxicity Marine Ecotoxicity	Ecotoxicity	Ecotoxicity	Ecotoxicity
	Photochemical Oxidant Formation	Photochemical Oxidation	Smog Formation	Photochemical Oxidant
	Terrestrial Acidification	Acidification	Acidification	Acidification
	Freshwater Eutrophication Marine Eutrophication	Eutrophication	Eutrophication	Eutrophication
	Ionising Radiation			Waste

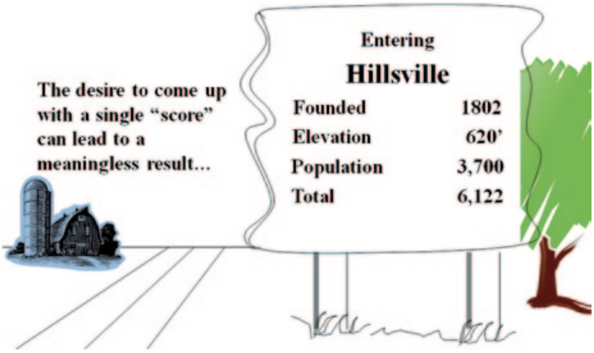

Fig. 6.2 Collapsing different impact category indicators into a single score is a subjective process involving weighting and normalization

LCI results are also typically unaccompanied by information about the temporal course of the emission (some environmental impacts may occur in the future) or the resulting concentrations in the receiving environment (Finnveden et al. 2009). With the inherent uncertainty in modeling environmental impacts, an impact indicator is the outcome of a simplified model of a very complex reality, giving only an approximation of the quality status of the affected entity. If not sufficient for absolute predictions of risk, LCIA models and LCA results are suitable for assessing relative comparisons.

2.6 LCA Does not Always (usually) Declare a 'Winner'

Converting impact results to a single score is a subjective process requiring value judgments[8], which cannot be based solely on natural science. All assumptions or decisions made throughout the study must be reported. If not, the final results may be taken out of context or misinterpreted (Fig. 6.2).

The interpretation phase of LCA entails the evaluation of the results of the inventory analysis along with the results of the impact assessment to aid in the decision making process, whether it is to select the preferred product, improve a process or service, etc. with a clear understanding of the uncertainty and the assumptions used to generate the results. Very seldom will the results of an LCA identify a clear 'winner' between alternatives. In some cases, it may not be possible to state that one alternative is better than the others because of the uncertainty in the final results. This does not imply that efforts have been wasted or that LCA is not a viable tool for decision makers. The LCA process will still improve understanding of the environmental and health impacts associated with each alternative, where they occur

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[8] Value judgments include the application of weighting (assignment and calculation of different impact categories and resources reflecting their relative importance) and normalisation (calculation of the magnitude of the category indicator results). In the ISO standard, normalisation is allowed for comparative assertions intended to be made available to the public, but not weighting due to its inherently subjective nature (ISO 14040+44).

(locally, regionally, or globally), and the relative magnitude of each type of impact in comparison to each of the proposed alternatives included in the study. This information more fully reveals the pros and cons of each alternative.

LCA can be used to establish a baseline of a product's environmental profile. But it is best used as a relative tool intended for comparison, and not absolute evaluation, thereby helping decision makers compare all major environmental impacts when choosing between alternative courses of action.

2.7 LCA Results should be Supplemented by Other Tools in Decision Making

While an LCA study produces very useful information, the results should be used as one component in a comprehensive decision-making process. It may be necessary to supplement the LCA with other tools or methods to provide a basis for decision-making. These tools include risk assessment, site-specific environmental assessment, cost assessment and others. As a part of the scoping process, it is useful to identify where and how these other tools will be used to augment the findings of the LCA. Further development is needed to create an integrated framework to reduce complexity while clarifying the simplification choices which have been made in the integrative analysis (CALCAS 2008).

In addition, the nature of LCA as an iterative process is often overlooked. Interpretation of the findings is about comparing the data and results with previous findings, and putting them in the proper context of decision-making and limitations. The iterative nature of the ISO framework (see Chap. 3[9] of this volume) shows up in this context. If the uncertainties are too high, we may go back to collect better data. If the sensitivity analysis shows that some decisions are crucial, we may go back and do a more refined analysis. It is especially important to determine that if the results of the impact assessment or the underlying inventory data are incomplete or unacceptable for drawing conclusions and making recommendations, then the previous steps must be repeated until the results can support the original goal of the study.

The decision tree shown in Fig. 6.3 depicts an iterative approach to collecting information in support of the decision making process for nanoproduct development (US EPA 2011). This approach, which follows the 'three pillar' interpretation of sustainability, can be applied to any product.

2.8 Allocating Environmental Burdens Across Co-products

When a process makes multiple products, the question of how to assign material use and environmental releases to each co-product becomes relevant. The ISO standard

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[9] The international standards as the constitution of life cycle assessment: the ISO 14040 series and its offspring by Matthias Finkbeiner.

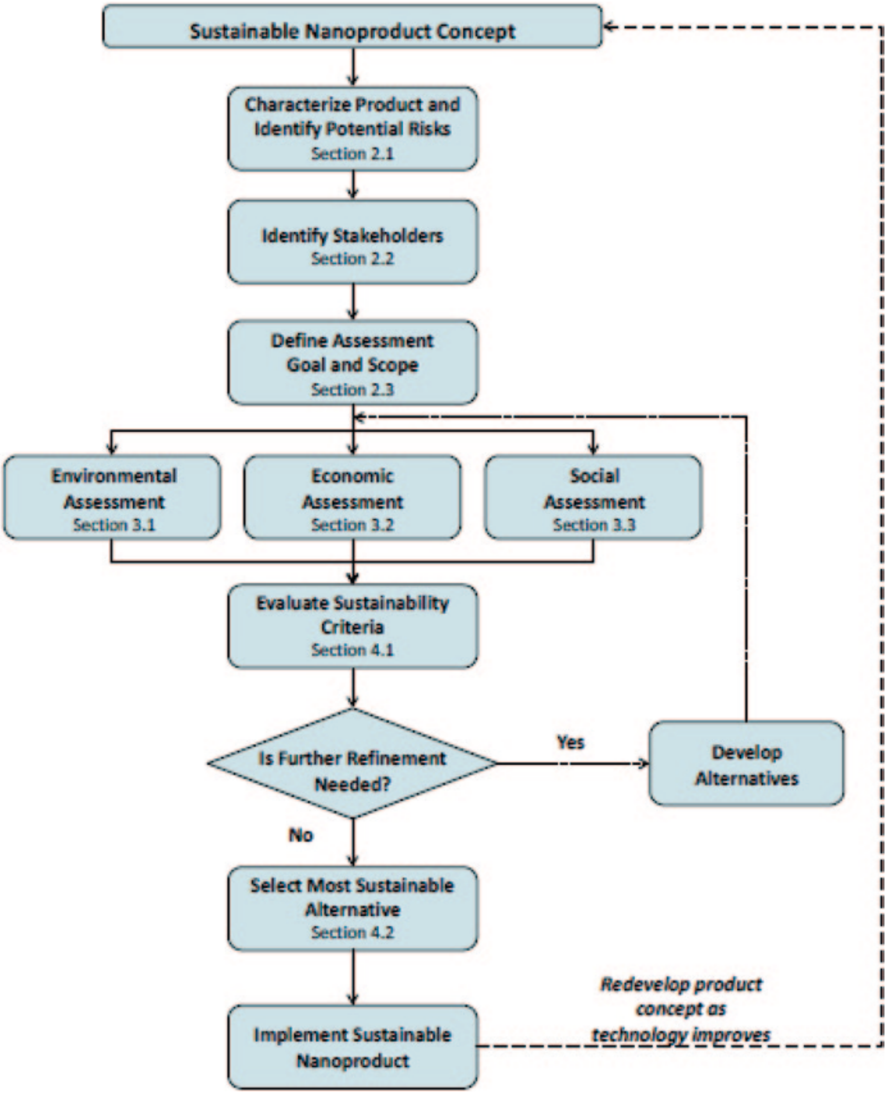

Fig. 6.3 US EPA's framework for sustainable nanotechnology. (US EPA 2011)

provides some guidance in the form of a hierarchy (Box 1), which calls for practitioners to avoid allocation if possible, by either (1) Modeling the sub-processes involved in production (i.e. collect more detailed data), or (2) Expanding the system boundaries to include additional processes that relate to the co-product(s). But much is left to interpretation in practice.

Box 1 Co-product allocation hierarchy (ISO 2006)

ISO 14041 6.5.3

On the basis of the principles mentioned above, the following step-wise procedure shall be applied.

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

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

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

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

There is general agreement that avoiding allocation through sub-process modeling and system expansion (Step 1 of the ISO hierarchy) is an appealing way to handle this seemingly intractable problem. However, both approaches cause the model to get larger and more complicated, requiring the collection of more data in order to complete the analysis. Collecting more data means more time and effort which brings the practicality of the approach into question. Also, larger systems run the risk of being less transparent in that there is more information on how the data were arrived at than can be easily communicated. So, although the answers that would be obtained through sub-process modeling would be more relevant to sustainability and more useful in helping decision-makers make better decision, allocation may not always be avoidable, especially if the data for the sub-processes or for the expanded system cannot be easily acquired.

2.9 Assigning Credit for Avoided Burden

In a system expansion approach, the boundaries are expanded to include the alternative production of exported functions. To do this, a necessary requirement of system expansion is the existence of an alternative way to produce a by-product. While this

Table 6.4 Energy ratio to produce corn ethanol calculated with co-product credit, 1996 (USDA 2002)

	Ethanol	Co-Products	Energy Use without Co-Product Credit	Energy Use with Co-Product Credit	NEV with Co-Products	Energy Ratio
	Percent	Percent	Btu/gal	Btu/gal[7]	Btu/gal[7]	Btu/gal[7]
Output weight basis:						
Wet mill	48	52	79,503	38,987	44,974	2.15
Dry mill	49	51	74,447	37,289	46,672	2.25
Weighted average	48	52	77,228	37,895	46,066	2.22

1000 Btu/US gallon = 0.279 megajoules per liter (MJ/l)
NEV Net Energy Value

concept seems reasonable on the surface, it can be controversial. It is often used to 'credit' the system with avoided burdens that are offset by the alternative process.

For example, corn mills produce both ethanol and corn oil; the corn ethanol system can be credited with the amount of energy it would have taken to make a competing product, such as soybean oil (Table 6.4). Not only does system expansion require more data to be collected, it also presents a problem with conveying the results of the study depending upon how the process in question was modeled. It is easy to see how the application of system expansion can have a significant impact on the study results.

Recycling, specifically open-loop recycling, is viewed as a special condition of allocation and is given special attention in the literature. The concern is to capture the downstream costs and benefits that post-consumer recycling may incur. Economic allocation seems to be the preferred approach and is perceived to be the best avenue to capture the downstream recycling activities. A number of allocation methods for open loop recycling are based on arguments about fairness, or accountability, so that environmental burden is appropriately assigned to the offending activity. However, it is difficult to determine which procedure is most 'fair' since this is a subjective term and depends on the perspective of the person conducting the study. The ISO 14040+44 standards stipulate the conduct of sensitivity analysis if "subjective" allocations are applied in order to show the effect the choice has on the results.

2.10 Expanding the Boundaries (consequential LCA)

By 2005, LCA practitioners began to take notice of expanded study boundaries that encompass the likely consequences of change resulting from a decision. This expanded approach to LCA became known as consequential LCA (Curran et al. 2005) to distinguish it from the more system-confined approach of attributional LCA. The change in the balance between supply and demand for a good or service can have a far-reaching impact. For example, Searchinger et al. (2008) found an attributional analysis of US corn-based ethanol resulted in a 20 % *decrease* in greenhouse gas emissions compared to conventional gasoline. However, in a consequential analysis to account for policy-driven increases in output, they predicted a 47 % *increase* in emissions compared to gasoline, due to land use changes induced by higher prices of corn, soybeans and other grains from anticipated additional demand for corn starch for ethanol production.

It is possible that the inventory results of a consequential LCA will be negative, if the change in the level of production causes a reduction in emissions greater than the emissions from the production of the product. This does not mean that the absolute emissions from the production of the product are negative, but that the production of the product will cause a reduction in emissions elsewhere in the system (Ekvall et al. 2005).

A consequential LCA is conceptually complex because it includes additional, economic concepts such as marginal production costs, elasticity of supply and demand, etc. A report prepared for the project 'Co-ordination action for innovation in life-cycle analysis for sustainability' (CALCAS 2009) outlines a four-step procedure to identify which unit processes to link:

- Identifying the scale and time horizon of the potential change studied;
- Identifying the limits of a market;
- Identifying trends in the volume of a market; and
- Identifying changes in supply and demand.

Therefore, consequential LCA depends on descriptions of economic relationships embedded in models. It generally attempts to reflect complex economic relationships by extrapolating historical trends in prices, consumption and outputs. This adds to the risk that inadequate assumptions or other errors significantly affect the final LCA results. To reduce this risk, it is important to ensure that the various results regarding different consequences can be explained using credible arguments. The main limitation for applying consequential LCA is the lack of the data in current LCA databases needed to support this type of modeling (CALCAS 2009).

There is no right or wrong choice between the attributional and consequential approaches, and the ISO standard does not offer specific guidance on how the goal of the study affects the scoping of the system boundary. While consequential modeling is relevant in most application areas of LCA, there are applications where the typical decisions studied by LCA are not of such significant size[10] and attributional modeling could be considered (CALCAS 2008). The distinction between attributional and consequential LCA is one example of how choices in the Goal and Scope Definition of an LCA influence methodological and data choices for the LCI and LCIA phases (Finnveden et al. 2009).

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[10] A decision is considered small or marginal when it does not affect the determining parameters of the overall market situation, that is, the direction of the trend in market volume and the constraints on and production costs of the involved products and technologies (CALCAS 2009).

Table 6.5 Life-cycle based approaches with a single issue focus

Title	Impact Metric
Life Cycle Greenhouse Gas (GHG) Analysis: 'Direct' as well as 'indirect' GHG emissions across the product lifecycle	Global Warming
Carbon Footprint: 'Direct' emissions of carbon dioxide (CO_2) from burning fossil fuels including domestic energy consumption and transportation as well as of 'indirect' CO_2 emissions from the product lifecycle	Global Warming
Water Footprint: Freshwater used by individuals or organisations to make goods or provide services	Water Depletion
Ecological Footprint: The amount of cropland, grazing land, forest area, and fishing grounds needed to satisfy a population's need for food, clothing, shelter, products and services, plus the amount of land required to absorb wastes	Land and Resource Use
Net Energy Balance: The overall gain or loss of energy, measured typically in Joules	Energy Production and Use
Chemical (Risk) Life Cycle: Multi-media environmental fate and transport, exposure, and effects on ecological receptors and human health across the life cycle of a chemical	Human and Ecological Health

3 Life Cycle Thinking

The preceding sections address issues related to the ISO-defined LCA methodology. In recent years the growing popularity of LCA and the life cycle concept have led to simplified approaches that focus on a single impact, thereby reducing the effort needed for data collection, impact assessment, and reporting. Table 6.5 lists several life cycle-based approaches that are commonly used to analyse and report select impact metrics.

It is clear that there is much variability in what life cycle-based tools measure and report (Curran 2013). In contrasting these approaches against LCA, it is also clear that focusing on specific issues of concern and not considering the whole suite of potential environmental concerns, risks overlooking potential burden shifting that may occur as a result of a decision. The conduct of an assessment that models only one or two pre-selected impact categories does not meet the definition of LCA, according to the ISO standards 14040+44.

4 Conclusion

Increasingly, decision makers are turning to LCA as a proven methodology to assess potential environmental impacts of products, goods and services. The ISO 14000 standard series provides a broadly accepted set of principles and the present-day

LCA framework. While LCA has come a long way in the development of methodology and continues to evolve with additional knowledge, LCA practice continues to suffer from variations in practice that can result in different LCA results. Some limitations, such as modeling missing impact indicators and making life cycle inventory more readily-available, will be addressed through continued research and development of the tool. Other modeling choice-related limitations, such as matching goal to approach setting a proper functional unit or appropriately scoping the assessment, need to be addressed through continued education and training to assist users in the proper application of the tool. Still other limitations in LCA practice would benefit by the development of harmonized guidance and global agreement by LCA practitioners and modelers.

Despite the variations outlined previously, LCA offers a strong environmental tool in our journey toward sustainability. Meeting the challenge of shifting the paradigm to one where LCA is the foundation of decision-making in regulation and commerce depends on public and private policy makers changing their belief systems and behaviors so their choices serve both current and future generations (Ngo 2012).

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