Life Cycle Sustainability Assessment for Decision-Making

Methodologies and Case Studies

Edited by

Jingzheng Ren • Sara Toniolo

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CHAPTER

1

Introduction. Life cycle thinking

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1.1 From the environmental concerns to a life cycle perspective

The issue of environmental sustainability is of great interest today (UNEP, 2011). The international community encourages companies to adopt cleaner production systems and technologies. The market seems to reward environmentally responsible organizations, and many companies around the world are increasingly becoming interested in environmental issues, introducing them as strategic variables in their businesses.

However, over the years, many environmental management tools have shown an important limit, that is the reduction of environmental impacts of an organization or a process by allocating them at other times, upstream or downstream of the supply chain, thus increasing the environmental loads of other subjects, such as suppliers, distributors, customers (O'Rourke, 2014). This is because many environmental management tools observe the environmental problem from a single point of view, the one of the single organization, while environmental problems are generated by different subjects that, together, contribute in a closely interconnected way to the overall environmental impact. With a physical point of view, the footprint of a product is the sum of the footprints of processes along the product supply chain in different times and geographical areas (Hoekstra and Wiedmann, 2014).

There are many examples of problem shifting, where solutions adopted to improve or solve a targeted problem unintentionally end up creating other problems of environmental, economic, or social nature elsewhere for other stakeholders. To solve this loop, a life cycle approach must be adopted.

Emerging interest in market concerns the green supply chain management, which explores various types of supply chain relationships and governance, encouraging a sustainable management of suppliers and distributors (Tseng et al., 2019). With a life cycle perspective, we consider the totality of the system in our analysis, including the evaluation of the product's entire life cycle, with a long-term time horizon and a multidimensional view. Life cycle

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thinking (LCT) offers this totality: a comprehensive analysis of the topic it requires, leading to solutions for reducing impacts in an absolute and not a relative way.

As shown in Fig. 1.1, a product's life cycle can begin with the extraction of raw materials from natural resources in the ground, and with energy generation. Materials and energy are then part of production, packaging, distribution, use, maintenance, and eventually recycling, reuse, recovery, or final disposal. In each life cycle stage there is the potential to reduce resource consumption and improve the product's performance.

The life cycle metaphor is borrowed from the field of biology. For example, the life cycle of a butterfly starts with an egg, which bursts and lets a caterpillar out, which then turns into a pupa, from which a butterfly emerges. The latter eventually dies after laying eggs for the cycle to be repeated. In much the same way a man-made object starts its lifecycle by the harvesting and extraction of resources, followed by production, use, and eventually management as waste, which marks the end of the life cycle (Bjørn et al., 2018a).

To minimize impacts, five levers can be used in practice, from a life-cycle perspective: lifetime extension, dematerialization, manufacturing efficiency, substitution, and recovery (Olivetti and Cullen, 2018). That's why we talk about LCT. Decisions made considering a full life cycle perspective and broader implications on the environmental, economic, and social pillars of a healthy planet, allow us to address unintended trade-offs between these pillars, and focus attention on the key drivers of change. As a result, progress towards sustainable development is faster and more efficient than when decisions are isolated (LCI, 2017).

Thinking in terms of the life cycle, businesses recognize that each choice sets the stage for not only how the product will look and function, but also for how it will impact the

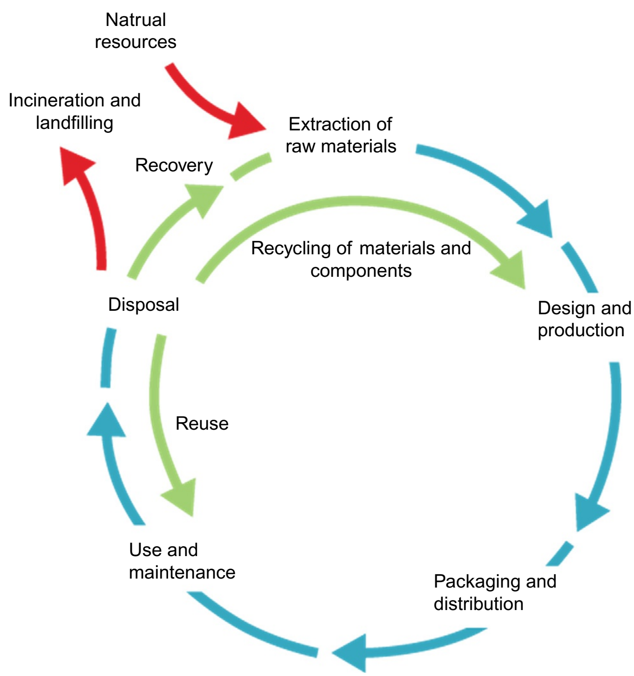

FIG. 1.1 A typical product lifecycle diagram. *Life Cycle Initiative, https://www.lifecycleinitiative.org/starting-life-cycle-thinking/what-is-life-cycle-thinking/.*

environment and the community as it is manufactured, used, disposed of, re-used, or recycled. Products can be designed so they will have less environmental impact when they are manufactured, used, and discarded. With a life cycle approach, companies are able to calculate the full life cycle cost of the goods they purchase. This includes the point-of-purchase price as well as the costs of transporting, storing, installing, cleaning, operating, repairing, and eventually discarding those goods (Hall, 2019).

As we will explore in this volume, LCT is not just a methodology of analysis; we can consider it a philosophy, a way of observing and reflecting, which leads to effective solutions for overall improvement of the sustainability of products, processes, and systems. The life cycle approach promotes relevant innovations in designing, producing and using products and services, and it brings benefits to several stakeholders along the product supply chain; we have summarized some benefits in Table 1.1.

To make choices addressed to life cycle approach, designers, manufacturers, and suppliers need tools for assessing the sustainability of alternatives, in terms of preferability and feasibility. The market too needs clear and quantified information, so that consumers and buyers are able to evaluate the sustainability of alternative products and make informed

TABLE 1.1 Main benefits of the adoption of LCT to the stakeholders along the product supply chain.

Stakeholders	LCT promotes	LCT avoids
Designers	Comprehensive, complete, and consistent analysis of all the factors that contribute to the impact of the product	partial analysis of the environmental, economic, and social impacts associated with single phases of a product's life cycle
Designers and manufacturers	Identification of solutions that improve the overall performance of the system, which includes the performance of all the actors in the supply chain	Identification of solutions that fix one environmental problem but cause another unexpected or costly environmental problem
Marketing staff and designers	Comparative evaluation of alternative business solutions in design, production, purchasing, distribution, use, and end-of-life	Inability to compare different design, production, and organizational alternatives
Consumers and market	Communication of clear and consistent information and creating awareness in the market	Communication of misleading information and disorientation in the market
Whole supply chain and community	Improvement of entire systems, not single parts of systems	Shift of problems from one life cycle stage to another, from one geographic region to another, and from one environmental item to another
Whole supply chain and community	Choices for the longer term and considering all related environmental and social issues	Short term decisions that lead to environmental degradation
Local and international governments	Investment of economic resources to support more sustainable projects	Wastage of investments in actions that do not improve the overall environmental performance or create inefficiencies

purchases. Moreover, local governments and international institutions must be able to have comprehensive and robust tools to guide companies and markets towards more sustainable production and consumption behavior. All these measurement needs find an answer in the most important operational tool of LCT: life cycle assessment (LCA). This analyses the whole life cycle of the system or product that is the object of the study and it covers a broad range of impacts for which it attempts to perform a quantitative assessment (ISO, 2006b). LCA is an important assessment tool, as demonstrated by the central role it is given in environmental regulation in many parts of the world and the strong increase in its use by companies all over the world (Hellweg and Milà i Canals, 2014). The focus of LCA has mainly been on the environmental impacts although, as we will see in following sections, both social and economic impacts can be included as well, with a more extended perspective known as sustainability assessment.

During the last 30 years, world leaders have explicitly recognized the need to change unsustainable patterns of production and consumption, and life cycle approaches play a key role. Demand for life cycle tools has increased, primarily thanks to numerous actions promoted by international initiatives to support the inclusion of life cycle approaches in governments worldwide. At the same time, in a market perspective, both companies and customers are giving increasing importance to impacts evaluation of products and services with a life cycle perspective. Today, LCT is a fundamental theme that involves multiple sectors and brings together the knowledge of many disciplines. Its current maturity is due to a progressive evolution over the years, in terms of practices, methodologies, and policies. The next section describes this evolution.

1.2 History of LCT

In the 1930s, economists begin discussing the unsustainability of welfare in an economy that uses non-renewable resources (Hotelling, 1931). In the 1960s, attention towards adverse environmental effects caused by environmental pollution increased and transparent and science-based information begin to be demanded by environmental scientists (Carsol, 1962). The first life cycle oriented study might be the one presented in 1963 by Smith in the World Energy Conference and it concerned the energy requirements for the production of chemical intermediates and products (Boustead, 2003). In this decade, the first life cycle studies in the United States and Northern Europe were conducted by some companies in the packaging sector, in order to develop production systems with energy saving and emissions reduction. These studies, carried out by large companies in an isolated manner, essentially focused on the firm's environmental management, aimed at improving internal processes, without interest in communicating to stakeholders (Hunt et al., 1992). Early methods, inspired by material flow accounting, were focused on inventorying energy and resource use, emissions, and solid waste. With more complex inventories, the focus was gradually extended with a translation from physical flows accounting into environmental impact evaluations, as contribution to climate change, eutrophication, and resource scarcity (Bjørn et al., 2018b).

In the 1970s, the concerns of the international community regarding environmental problems created by some industrial activities were growing (Meadows et al., 1972). Scientists recognized resource consumption and waste production as the main causes of environmental problems and recommended the closure of the cycle with reliability, reparability, and recyclability of products at the end of life (Singer, 1970). At the same time, in chemicals and packaging sectors, the interest in life cycle evaluation continued to grow, focusing on energy consumption, solid waste production, and air emissions. In these years, the first public and peer-reviewed LCA study was published, commissioned by the US Environmental Protection Agency with the aim of informing regulation on packaging (US EPA, 1974).

During the 1980s, the life cycle approach evolved in both applications and methodologies, thanks to companies' interest and the scientific debate. In European countries, environmental attention related to the impacts of milk packaging increases and LCA studies were conducted to compare alternative packaging systems for milk distribution to private consumers. Numerous applications of life cycle evaluation on technologies and similar products with conflicting results revealed the need for the developing of rigorous methodologies. Then, knowledge and metrics concerning cause-effect mechanisms in several environmental concerns were deepened by scientists, to define rigorous impacts quantification and avoid burden shifting. In these years, the first impact assessment method based on critical volumes was introduced (BUS, 1984) and the first two pieces of commercial LCA software were released (Gabi in 1989 and SimaPro in 1990). In line with the life cycle perspective, the United Nations published the report "our common future"—a milestone in sustainable development history—in which the importance of recycling and renewable resources is declared (UN, 1987).

In the 1990s, the life cycle approach spread. This decade marks the most important steps for the construction of LCT. The United Nations proclaimed the principles intended to guide countries in future sustainable development (UN, 1992). Meanwhile, the term "life cycle assessment" is coined (SETAC, 1993), and the first standards are published to harmonize life cycle practices (Fava et al., 1994; ISO, 1997). At the same time, several life cycle inventory databases are developed by different institutions, and new impact assessment methodologies are developed, including cause-effect-damage evaluations (Bjørn et al., 2018b). During this decade, the first scientific LCA related study is published (Guinée et al., 1993) and an academic journal fully dedicated to the LCA is born (Klöpffer, 1996).

With the beginning of the new millennium, the international community gave a fundamental role to LCT for construction of a sustainable future. In 2002, at the World Summit on Sustainable Development, world leaders recognized the need to change the unsustainable development model and subscribe common commitment to implement sustainable production and consumption "using, where appropriate, science-based approaches, such as life cycle analysis" (UN, 2002). In the same year, the United Nations Environmental Protection and Society of Environmental Toxicology and Chemistry launch the Life Cycle Initiative, focused on the dissemination of life cycle practices all over the world and, in particular, to emerging economies (LCI, 2002). In the European context, LCT receives a strong push by the European Integrated Product Policy (IPP), which supports policy instruments like environmental labeling, green public purchase, and integration of environmental aspects into standards development (EC, 2003). Moreover, in 2005, the European Commission creates the European platform on LCA to promote the life cycle perspective at both theoretical and operational level (Wolf et al., 2006). Influencing market dynamics, the European policy contributes to the spread of life cycle tools around the world.

In the 21st century, methodological approaches of LCT improve: the international standards of LCA are revised (ISO, 2006b, 2006c), and life cycle perspective is gradually applied in several sectors and integrated with other decision support tools in almost all the areas where environmental, economic, and social considerations are important. In these years, new frameworks aiming to extend LCA methodology to economic and social aspects of sustainability are elaborated (Guinée, 2016), and the concept of life cycle is adopted in several standards with different meanings and applications (Toniolo et al., 2019b).

Over the last two decades, impact assessment methods have been continuously refined and several methodologies updated; from 1999 to date, more than 20 methodologies of life cycle impact assessment have been published worldwide by several organizations (Rosenbaum, 2017). Through methodological consolidation, life cycle approach has a large and rapid spread, increasing the range of products and systems analyzed by both industries and governments. The interest in life cycle studies has increased, due to the growing public awareness of environmental issues and a widespread acceptance of sustainable development (Hou et al., 2015).

What happens next is actuality, which will be presented in the next chapters of this book. What I want to emphasize here, for an overview, is the fact that, from the 2000s, the increase in LCT initiatives around the world has gone hand in hand with increasing knowledge of environmental problems. On the one hand, greater environmental awareness pushes the scientific community to improve methods for assessing environmental impacts, while on the other, it leads the market to request more information on environmental impacts associated with products. Thus, a virtuous circuit is established, in which local governments promote LCT tools on the market, consumers are better informed and choose more consciously, companies invest in life cycle evaluations to improve their products, also communicating results to the market. To witness this virtuous circuit, we can see that, where the number of life cycle initiatives increases, available information concerning territorial environmental quality increases as well, and indicators of the overall environmental condition show a progressive improvement (Qian, 2016).

Fig. 1.2 summarizes the main evolutionary steps of the LCT along the timeline. In this graph, from 1960 to date, a progressive increase characterizing the LCT story is highlighted in four interdependent directions: life cycle practices, life cycle methods, life cycle publications, and life cycle policies. The first life cycle reasoning is done in the 1960s, when environmental degradation and limited access to resources start becoming a concern. In the following years, LCT takes shape and is gradually enriched through application, harmonization, and dissemination. Life cycle practices also started in the 1960s, as isolated experiences, recording a strong boost during the 1990s, due to the birth of standards and software to support the life cycle analyses. Since the 1990s, government initiatives supporting the life cycle approach have multiplied and scientific literature has exploded. Nowadays, the panorama of experiences, methodologies, and publications concerning LCT is enormously rich and interdisciplinary, thanks to the complicity of international policies that recommend its use in all economic sectors.

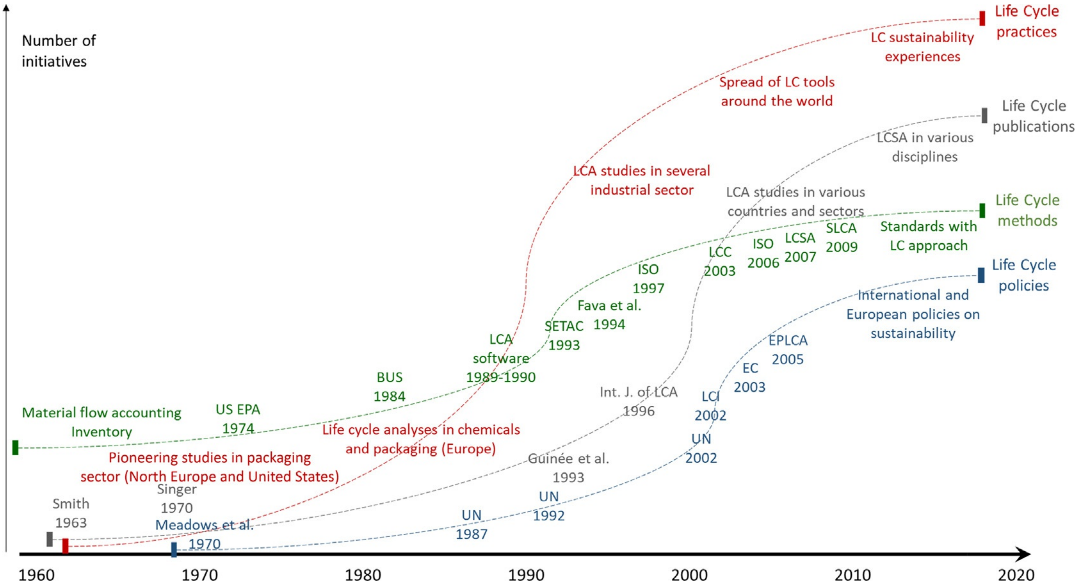

FIG. 1.2 Timeline of LCT milestones.

1.3 LCT and sustainability

The link between LCT and sustainable development is tight. On the one hand, sustainability presupposes giving an overriding priority to the essential needs coherently with environmental limits, available technologies, and socio-cultural context (UN, 1987). On the other hand, LCT aims to consider all the impacts associated to a product life cycle in order to indicate priority of interventions that are more convenient and useful (EC, 2003).

Sustainable development should ideally improve the quality of life for every individual without expending the Earth's resources beyond its capacity. Without a functioning environment we will not be able to give future generations the same possibilities for achieving the levels of welfare that current generations are experiencing. Researchers have attempted to quantify carrying capacities of the ecosystem that must not be exceeded to maintain functions, as well as other ecosystem aspects of interest. Planetary boundaries can be interpreted as carrying capacities for the entire Earth system towards various anthropogenic pressures, such as greenhouse gases and interference with nutrient cycles (Rockström et al., 2009). According to estimates, this exceedance has already happened for four of the nine proposed planetary boundaries (Steffen et al., 2015).

Acting to reduce the impact on the ecosystem is, therefore, necessary and urgent, but needs a collective effort. The journey towards sustainable development requires that businesses, governments, and individuals take action, changing consumption and production behaviors, setting policies, and changing practices. Human needs should be met by products and services that are provided through optimized consumption and production systems that do not exceed the capacity of the ecosystem.

Sustainability has three dimensions: economy, society, and environment. In the business community the term "triple bottom line" was coined to explain the importance of achieving sustainability; it implies that industry has to expand the traditional economic focus to include environmental and social dimensions, in order to create a more sustainable business (Elkington, 1997).

LCT expands the established concept of cleaner production to include the complete product life cycle and its sustainability. Source reduction in a product life cycle perspective is then equivalent to designing with sustainability principles in mind. In each life cycle stage there is the potential to reduce resource consumption and improve the performance of products; in order to succeed, all the stakeholders in the product chain have to be involved, using a collaborative approach and integrating efforts, with the same goal: sustainability. Overall, LCT can promote a more sustainable rate of production and consumption and help us use our limited financial and natural resources more effectively. We can derive increased value from money invested—such as wealth creation, accessibility to wealth, health and safety conditions, and fewer environmental impacts—by optimizing output and deriving more benefits from the time, money, and materials we use.

The full consistency of LCT with the sustainable development concept is therefore confirmed. Moreover, recent developments of the life cycle approach explicitly adopt sustainability as a framework: international policies have adopted the "3Ps" of sustainability, which stand for "people, planet, and prosperity", and linked LCT to sustainable development agenda (UN, 2002). Meanwhile, the scientific community has developed advanced models of LCA methodology, including the triple bottom line perspective: thus, life cycle costing (LCC) and social life cycle assessment (SLCA), as second and third pillars of sustainability, are born, distinguishing economic and social impacts of product systems along their life cycle. Moving to a more comprehensive assessment of sustainability, the life cycle sustainability assessment (LCSA) is the most modern life cycle-based approach to evaluate scenarios for sustainable futures and practical ways to deal with uncertainties and rebound effects with a comprehensive vision (Guinée, 2016).

Fig. 1.3 shows the possible link between LCT and sustainable development through the three pillars of sustainability and the multidimensionality of LCT.

In 2015, the 193 member states of the United Nations adopted 17 goals to "end poverty, protect the planet, and ensure prosperity for all as part of a new sustainable development agenda" by 2030 (UN, 2015a, b). To meet the goals and targets, sustainability must gain strong prominence in decision making support for all economic actors along the supply chain who are responsible for creating solutions for the future: all companies that design, create, supply, and buy, all consumers that choose, buy, use, and dispose, all local governments and institutions that regulate, control, and support.

To support sustainable decisions, from small- to large-scale perspective, the market needs comprehensive and robust tolls. To avoid the often-seen phenomenon of problem shifting, where the solution to a problem creates several new problems, decisions must be taken with a systems perspective. LCT aims to facilitate the application of life cycle knowledge in the global sustainable development agenda in order to achieve the sustainable development goals faster and more efficiently (Wulf et al., 2018). Through the life cycle approach, we recognize how our choices influence what happens at each phase, so we can balance trade-offs in economic and environmental consequences caused by our choices.

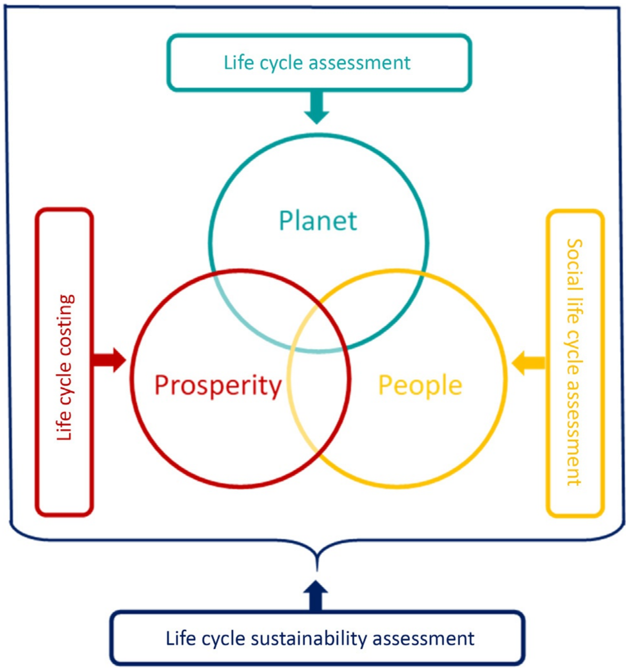

FIG. 1.3 Possible link between LCT and sustainable development in the triple bottom line perspective.

Further challenges of LCT in achieving sustainable development goals are described in Section 1.5.

1.4 Tools and actions in LCT

A life cycle approach identifies opportunities and risks of a product or technology, from raw materials to disposal, named "from cradle to grave". Consumers, companies, and governments use these various life cycle approaches for many different purposes, from day-to-day shopping, to selecting suppliers, engineering a new product design, or developing a new process, project, or business. Citizens, businesses, and governments are finding ways to promote LCT and to balance the impacts of their choices. A life cycle approach applied to community planning and development can lead to fewer environmental impacts from materials used, construction practices, and waste management, as well as energy and water used by people living and working in the community.

To support diffusion of the life cycle approach among business communities and local governments, the scientific community and international organizations promote numerous initiatives, which we can summarize in two typologies:

- Life cycle tools, which include standards and guidelines to assist researchers, practitioners, and companies in applying the principles of life cycle approach to products, processes, and projects;
- Life cycle actions, which include disseminating and supporting initiatives aimed at spreading the life cycle approach in international and local policies, as well as fostering the understanding and use of life cycle tools between companies and consumers.

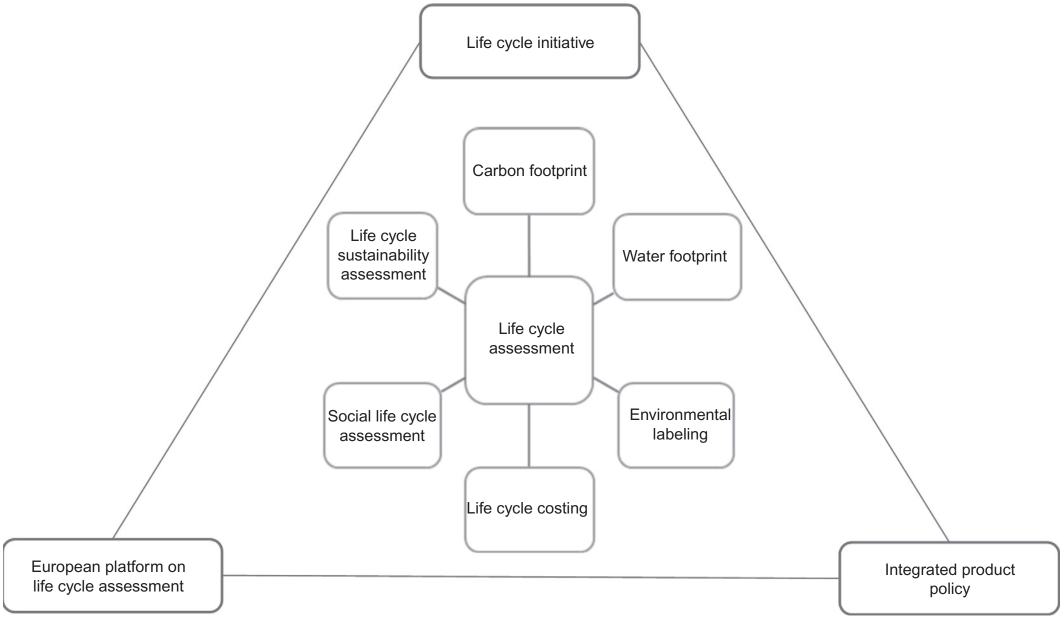

FIG. 1.4 Tools and actions in LCT.

Fig. 1.4 shows the main initiatives in LCT, as life cycle tools and actions. The following chapters of this book will describe them. Here a brief summary is given.

1.4.1 Life cycle assessment

LCA represents "the best framework for assessing the potential environmental impacts of products" (EC, 2003). It is a method to assess quantitatively the environmental impacts of goods and processes from cradle to grave. LCA models cause-effect relationships in the environment and thus helps to understand the environmental consequences of human actions.

To conduct an LCA study for products and services in many economic activities around the world, practitioners are supported by two international standards: the ISO 14040 and the ISO 14044, respectively containing general principles and specific requirements for an LCA (ISO, 2006b, 2006c). Four features of LCA make it a complete and robust tool to support companies and markets in sustainability commitments: it takes a life cycle perspective, covers a broad range of environmental issues, is quantitative, and is science-based (Bjørn et al., 2018a).

LCA is an important decision-support tool that, among other functions, allows companies to benchmark and optimize the environmental performance of products or for authorities to design policies for sustainable consumption and production. Many LCA studies are conducted to support corporate internal decision-making, such as for ecodesign of products, process optimizations, supply-chain management, and marketing and strategic decisions (Hellweg and Milà i Canals, 2014). Recent initiatives go a step further, by aiming to generalize the life cycle approach in all consumption sectors, through harmonization of life cycle-based information on a variety of impact categories to be displayed in product labeling (Toniolo et al., 2019a).

1.4.2 Life cycle costing, social life cycle assessment, and life cycle sustainability assessment

In designing, manufacturing, delivering, using, recovering, and disposing products, various requirements have to be integrated with environmental aspects: feasibility, convenience, security, acceptability; often conflicting requirements have to be fulfilled. Therefore, to support complex decisions, multidimensional approaches are necessary (Mazzi et al., 2016). Both scientists and companies have recently moved in this direction, extending the LCA model to economic and social dimensions.

To be honest, the concept of environmental LCC predates LCA: life cycle cost refers to all costs associated with the system in a defined temporal life cycle (Blanchard and Fabrycky, 1998). Recently, the LCA community has come closer to this concept with the aim of integrating financial data and cost information with environmental life cycle metrics. Then, LCC has become the acronym of the tool which, consistently with LCA model, across the product's life cycle, includes all costs borne by different actors with different perspectives and at different times (Hunkeler and Rebitzer, 2003). A code of practice for LCC has been published by the Society of Environmental Toxicology and Chemistry for evaluating decisions with consistent systems boundaries as a component of product sustainability assessments (Swarr et al., 2011). In a company perspective, LCC is a key tool for sustainable business, because it helps in giving the right signal on economic implications of sustainable production for the decision-maker as well as giving priority to the most cost-effective environmental improvements (Hannouf and Assefa, 2016).

The SLCA is a methodological approach aimed at evaluating social and socioeconomic aspects of products and their potential positive and negative impacts along their life cycle. Social impacts are those that may affect stakeholders along the product life cycle and may be linked to company behavior, socioeconomic processes, and impacts on social capital (Benoît and Mazijn, 2009). From a company perspective, one of the main added values of SLCA is the possibility to spend the results of social evaluation on the market. This could be achieved, for example, by means of a social label (Zamagni et al., 2011).

SLCA is still not widespread because it suffers from a double difficulty: definition and application. SLCA encompasses unquantifiable issues of ethics and values with holistic and personnel perspectives, such as active citizenship, well-being and happiness, preserving sociocultural diversity, and meeting basic needs (Mattioda et al., 2015). Recent efforts to facilitate the practicality of SLCA are directed to solve the lack of available data and the difficulty to evaluate immaterial impacts with undefined cause-effect relationships (Weidema, 2018).

Concerning life cycle sustainability assessment (LCSA), definitions are not yet carved in stone. Two main definitions of LCSA exist. Klöpffer and Renner (Klöpffer, 2008) propose to calculate the LCSA as the sum of the three studies: LCA, LCC, and SLCA; thus, LCSA broadens LCA methodology including economic and social aspects in the life cycle evaluation. Guinée et al. (2011) start from the previous definition and add two dimensions of evaluation, related to the external contest of organizations: technological conditions and economic state.

Moving from theory to practice, the concept of life cycle sustainability is presented in several standards with different meanings and applications. Even if all sustainability dimensions are standardized by international community, environment still remains the most considered one in a life cycle approach (Toniolo et al., 2019b).

1.4.3 Partial LCAs: Carbon footprint and water footprint

Over the last 50 years, some critical environmental issues have particularly worried the international community: the emission of greenhouse gases is the main cause of global climate change; the scarcity of freshwater availability is critical for healthy lives and a healthy planet; the energy consumption closely linked to the availability of nonrenewable resources is a dangerous brake on economic development and a threat to political and social world balance; and increasing land use and fossil fuel combustion are leading to enhanced losses of reactive nitrogen to the environment. Attention to specific environmental issues has led the scientific community to develop impact assessment tools able to go into depth on individual environmental issues. Since the 1980s, in order to know environmental impacts related to greenhouse gases emission, water consumption, energy sustainability, and nitrogen variation, among companies, new metric needs have emerged. To meet the market's needs and provide businesses and consumers with rigorous assessment methods, new standards have been published for the calculation of the so-called "partial LCAs."

To calculate the carbon footprint (CF) of a product or service, the ISO 14067 specifies methodology and requirements to measure the emissions of greenhouse gases in input and output of a product's life cycle, and the associated environmental impacts on climate change (ISO, 2018b). This result corresponds to the partial result of LCA related to the life cycle impact category indicator "global warming potential"; therefore, CF is a typical case of "partial LCA."

To support organizations in assessing the environmental profile of water footprint (WF) consumption and degradation, the ISO 14046 indicates methodology and characteristics that need to be taken into consideration when assessing the WF of a product from a life cycle perspective (ISO, 2014). WF is defined as a metric that quantifies the potential environmental impacts related to water. It includes identification and evaluation of the impacts related to consumptive water use (e.g., scarcity and availability) and related to degradative water use (e.g., eutrophication and acidification). The WF gives a profile of the impact category results that can be reported in a standalone study or as part of a more comprehensive LCA study (Mazzi et al., 2014).

The environmental profile obtained by these partial LCAs has some advantages but also limitations. From a scientific perspective, partial LCAs lack a comprehensive environmental view, because they observe inputs and outputs of the product life cycle with a partial view which, despite being important, is still relative. On the other hand, LC tools such as CF and WF may be more detailed than a complete LCA in examining specific environmental problems because, by focusing on single environmental parameters, they investigate thoroughly the cause-effect-damage relations of a single impact category.

From a market perspective, along supply chains and towards consumers, the demand for information concerning environmental footprint increases, and it guides efforts in ecoinnovation, ecomanagement, and ecolabeling. Companies and consumers undoubtedly prefer synthetic and immediate evaluation and communication tools, although in taking CF or WF as the one and only yardstick, one has to face life-threatening trade-offs (Finkbeiner, 2009).

Future perspectives to develop partial LCAs do exist; these life cycle tools can deepen the analysis of single environmental problems, enriching environmental models that support life cycle impact assessment with ad hoc quantification and regionalization (Bulle et al., 2019).

1.4.4 Ecolabeling

LCA, originally developed to be used as a decision support tool for environmental management, now has several related applications such as external communication through environmental labels and declarations. As ISO classified, three typologies of environmental labels exist: type I, II, and III; for each of them we can refer to ad hoc standards, for measuring and communicating the environmental performance of products: ISO 14024 (ISO, 2018a), ISO 14021 (ISO, 2016), and ISO 14025 (ISO, 2006a), respectively.

In recent years, the market demand for environmental product declarations, such as type III environmental labels, has increased, as well as the number of program operators (Toniolo et al., 2019a). At the same time, in the European market, the European Commission launched the Product Environmental Footprint (EC, 2013), a multicriteria method to calculate the environmental profile of products with a life cycle perspective; it is an applicable tool supporting external communication or public procurement tender requirements.

This growing number of different environmental product declaration schemes with different requirements causes confusion in the market and disorientation in purchasing decisions; consequently, an effort to make labels more reproducible, comparable, and verifiable, will be much appreciated by the market (Del Borghi et al., 2019).

1.4.5 Life cycle initiative

Hosted by the UN, thanks to the common commitments of the United Nations Environmental Protection and Society of Environmental Toxicology and Chemistry, the Life Cycle Initiative (LCI) is the interface between users and experts of life cycle approaches (LCI, 2017). LCI provides a global forum to ensure a science-based and consensus-building process to support decisions towards the shared vision of sustainability as a public good. It delivers authoritative opinion on sound tools and approaches by engaging multi-stakeholder partnership, including governments, private and public organizations, scientists, scholars, and civil society.

The LCI is a public-private, multistakeholder partnership enabling the global use of credible life cycle knowledge by private and public decision makers. It facilitates the application of principles and tools of LCT in local governments and markets. The mission of LCI has two main directions: to improve decisions that need assessment and comparison of products, technologies, lifestyles, and economy-wide choices, and to build consensus on environmental, social, and economic life cycle knowledge, with inventory data, impact assessment methods, and indicators. To pursue these commitments, LCI establishes periodic action programs and verifies the results; through numerous initiatives and its website, it provides publications and communications and promotes collaboration between stakeholders around the world.

1.4.6 Integrated product policy

IPP is a European initiative, developed by the Directorate-General for Environment, aimed at reducing the environmental burden of products and services throughout their life cycles. This can be achieved using a toolbox of policy instruments that make markets more sustainable through greening both the demand side (consumption) and the supply side (product development) (EC, 2003). It is an attempt by the European Commission to create conditions in which environment-friendly products, or those with a reduced impact on the environment, will gain widespread acceptance among the European Union Member States and the European market.

IPP, within environmentally advanced countries in Europe, is part of a growing trend towards product-oriented environmental policies. It seeks to minimize all environmental degradations caused by products throughout the entire life cycle, by looking at all phases of its life cycle and acting where it is most effective. To achieve this challenging goal and succeed in intervening on different subjects with often contradictory interests, IPP includes several measures such as economic instruments, substance bans, voluntary agreements, environmental labeling, and product design guidelines.

1.4.7 European Platform on LCA

The European Platform on LCA represents the European answer to business and policy needs for social and environmental assessments of supply chains and end-of-life waste management. It was born primarily to support the European IPP, to increase the availability of quality-assured life cycle data (JRC, 2006). The European Platform on LCA is implemented by the Joint Research Centre, in collaboration with the European Directorate-General Environment, to support business and government needs for availability, interoperability, and quality of life cycle data and studies, supplying guidelines spanning from methodological aspects to characterization models. For more than 10 years, this platform has elaborated frameworks and guidelines to support the LCA practitioners, with methodological and practical improvements of inventory databases and impact assessment methods (Sanfélix et al., 2013).

1.5 Emerging trends in LCT

Many authors in recent years have highlighted development prospects of LCT tools. Recalling some of them here, rather than to be exhaustive, I want to stimulate once again the discussion.

Several recent papers emphasize the directions in which LCT tools must evolve to meet emerging market demands. To continue building a greater demand is fundamental to redirect resources of companies and governments towards a life cycle strategy. The starting point must remain understanding, identifying, and managing risks, opportunities, and trade-offs associated with products, technologies, and services over their whole life cycle (Fava, 2016). Unlike the more traditional site-specific approaches to environmental protection, sustainability strategies have implications that extend across a product's life cycle and require engaging stakeholders who can influence the ability to manufacture and sell products around the world. Next, research developments, both in scientific and business communities, will investigate the adaptation of collaborative supply chain solutions with sustainability issues, through application of LCT in supplier management (O'Rourke, 2014). Sustainable public procurement and sustainable buildings will likely create the most immediate demand for life cycle approaches in the market, with a "domino" effect. Whether and how the financial sector incorporates life cycle approaches into their sustainability rating schemes could be a further demand that will push the diffusion of life cycle approaches.

The growing knowledge of environmental problems and cause-effect mechanisms at local and global level determines emerging needs for the assessment of environmental impacts. That translates into efforts to improve life cycle inventories, enriching available local and global data and information, and integrating the life cycle impact assessment models to include more detailed and site-specific cause-effects relationships. At the same time, the users of life cycle tools need to have available life cycle evaluations with intelligent results, which include uncertainties and knowledge limitations (Jolliet, 2006).

Climate change stresses terrestrial ecosystems, increasing seasons' length, and altering community composition; these stresses enhance productivity and water-use efficiency, but also lead to increased mortality and disturbances from wildfires, insects, and extreme meteorological events. Next, changes of life cycle tools must consider the link of climate processes with Earth system models, including vulnerability-adaptation descriptors such as atmospheric and oceanic states, land use, habitat loss, water availability, wildfire risk, air quality, crops, and fishery (Bonan and Doney, 2018). For future-oriented decisions, environmental assessment life cycle methodologies must progressively include indirect impacts, related land use, water consumption, air emissions, acidification, eutrophication, and so on. For this purpose, the traditional LCA model must be combined with other disciplines, such as general and partial equilibrium models from economic sciences (Hellweg and Milà i Canals, 2014).

Until now, the social and ethical dimensions of sustainability have not been given the same attention within the business community, since the benefits are less tangible. However, examples of positive links exist between environmental improvements and health and safety improvements in the workplace. Now, a general trend shows companies' and governmental policies to be more sensible towards integrated management systems in order to take into consideration also health and safety issues, as well as other social aspects (Zamagni, 2012).

Emerging approaches combining LCT, triple bottom line, and sustainable development goals prove that some difficulties remain to solve. Focusing on LCSA, its application requires proper and quantitative data and methods for LCSA indicators, including dealing with value choices and subjectivity and the guidelines for external communication (Guinée, 2016).

References

Benoît, C., Mazijn, B. (Eds.), 2009. Guidelines for Social Life Cycle Assessment of Products. Social and Socio-Economic LCA Guidelines Complementing Environmental LCA and Life Cycle Costing, Contributing to the Full Assessment of Goods and Services within the Context of Sustainable Development. UNEP/SETAC Life Cycle Initiative, Paris, France.

Bjørn, A., Owsianiak, M., Molin, C., Laurent, A., 2018a. Main characteristics of LCA. In: Hauschild, M.Z., Rosenbaum, R.K., Olsen, S.I. (Eds.), Life Cycle Assessment: Theory and Practice. Springer, ISBN: 978-3-319-56474-6, pp. 9–16. Chapter 2.

Bjørn, A., Owsianiak, M., Molin, C., Hauschild, M.Z., 2018b. LCA history. In: Hauschild, M.Z., Rosenbaum, R.K., Olsen, S.I. (Eds.), Life Cycle Assessment: Theory and Practice. Springer, Cham, Switzerland, ISBN: 978-3-319-56474-6, pp. 17–30. Chapter 3.

Blanchard, B., Fabrycky, W.J., 1998. Systems Engineering and Analysis. Prentice Hall, Upper Saddle River, NJ.

Bonan, G.B., Doney, S.C., 2018. Climate, ecosystems, and planetary futures: The challenge to predict life in earth system models. Science. 359 (6375), eaam8328. https://doi.org/10.1126/science.aam8328.

Boustead, I., 2003. Chapter 3: Polymers and energy. In: Andrady, A.L. (Ed.), Plastics and the Environment. Wiley Interscience, Hoboken, NJ, ISBN: 0-471-09520-6, pp. 123–135.

Bulle, C., Margni, M., Patouillard, L., Boulay, A.M., Bourgault, G., De Bruille, V., Cao, V., Hauschild, M., Henderson, A., Humbert, S., Kashef-Haghighi, S., Kounina, A., Laurent, A., Levasseur, A., Liard, G., Rosenbaum, R.K., Roy, P.O., Shaked, S., Fantke, P., Jolliet, O., 2019. IMPACT World+: a globally regionalized life cycle impact assessment method. Int. J. Life Cycle Assess. https://doi.org/10.1007/s11367-019-01583-0.

BUS, 1984. Bundesamt für Umweltschutz: Ökobilanzen von Packstoffen (Life-cycle assessment of packaging materials) (in German). In: Schriftenreihe Umweltschutz. Bern, Switzerland, p. vol. 24.

Carsol, R., 1962. Silent Spring. Houghton Mifflin Harcourt, Boston, MA.

Del Borghi, A., Moreschi, L., Gallo, M., 2019. Communication through ecolabels: how discrepancies between the EU PEF and EPD schemes could affect outcome consistency. Int. J. Life Cycle Assess. https://doi.org/10.1007/s11367-019-01609-7.

EC, 2003. Integrated Product Policy. Building on Environmental Life-Cycle Thinking. Communication from the Commission to the Council and the European Parliament COM/2003/0302 final. https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:52003DC0302&from=EN. (Accessed 19 May 2019).

EC, 2013. ANNEX II: Product environmental footprint (PEF) guide to commission recommendation on the use of common methods to measure and communicate the life cycle environmental performance of products and organisations. In: European Commission (Ed.), Official Journal of the European Union.

Elkington, J., 1997. Cannibals with Forks: The Triple Bottom Line of 21st Century Business. Capstone, New Society.

Fava, J., Denison, R., Jones, B., Curran, M.A., Vigon, B., Selke, S., Barnum, J. (Eds.), 1994. SETAC Workshop Report: A Technical Framework for Life-Cycle Assessment. Smugglers Notch, Vermont, August 18–23 1990. SETAC Press.

Fava, J.A., 2016. Will the next 10 years be as productive in advancing life cycle approaches as the last 15 years? Int. J. Life Cycle Assess. 11 (Special Issue 1), 6–8. https://doi.org/10.1065/lca2006.04.003.

Finkbeiner, M., 2009. Carbon footprinting—opportunities and threats. Int. J. Life Cycle Assess. 14, 91–94. https://doi.org/10.1007/s11367-009-0064-x.

Guinée, J.B., Udo de Haes, H.A., Huppes, G., 1993. Quantitative life cycle assessment of products: 1. Goal definition and inventory. 2. Classification, valuation and improvement analysis. J. Clean. Prod. 1 (1), 3–13. https://doi.org/10.1016/0959-6526(93)90027-9.

Guinée, J.B., Heijungs, R., Huppes, G., Zamagni, A., Masoni, P., Buonamici, R., Ekvall, T., Rydberg, T., 2011. Life cycle assessment: past, present, and future. Environ. Sci. Technol. 45 (1), 90–96. https://doi.org/10.1021/es101316v.

Guinée, J., 2016. Life cycle sustainability assessment: what is it and what are its challenges? In: Clift, R., Druckman, A. (Eds.), Taking Stock of Industrial Ecology. Springer, Cham, ISBN: 978-3-319-20570-0, pp. 45–68.

Hall, M.R., 2019. The sustainability price: expanding environmental life cycle costing to include the costs of poverty and climate change. Int. J. Life Cycle Assess. 24, 223–236. https://doi.org/10.1007/s11367-018-1520-2.

Hannouf, M., Assefa, G., 2016. Comments on the relevance of life cycle costing in sustainability assessment of product systems. Int. J. Life Cycle Assess. 21 (7), 1059–1062. https://doi.org/10.1007/s11367-016-1136-3.

Hellweg, S., Milà i Canals, L., 2014. Emerging approaches, challenges and opportunities in life cycle assessment. Science 344 (6188), 1109–1113. https://doi.org/10.1126/science.1248361.

Hoekstra, A.Y., Wiedmann, T.O., 2014. Humanity's unsustainable environmental footprint. Science 344 (6188), 1114–1117. https://doi.org/10.1126/science.1248365.

Hotelling, H., 1931. The economics of exhaustible resources. J. Polit. Econ. 39 (2), 137–175. https://doi.org/10.1086/254195.

Hou, Q., Mao, G., Zhao, L., Du, H., Zuo, J., 2015. Mapping the scientific research on life cycle assessment: a bibliometric analysis. Int. J. Life Cycle Assess. 20 (4), 541–555. https://doi.org/10.1007/s11367-015-0846-2.

Hunkeler, D., Rebitzer, G., 2003. Life cycle costing–paving the road to sustainable development? Int. J. Life Cycle Assess. 8 (2), 109–110.

Hunt, R.G., Sellers, J.D., Franklin, W.E., 1992. Resource and environmental profile analysis: a life cycle environmental assessment for products and procedures. Environ. Impact Assess. Rev. 12 (12), 245–269. https://doi.org/10.1016/0195-9255(92)90020-X.

ISO, 1997. ISO 14040:1997—Environmental Management—Life Cycle Assessment—Principles and Framework. International Organization for Standardization, Geneva, Switzerland.

ISO, 2006a. ISO 14025:2006 Environmental Labels and Declarations—Type III Environmental Declarations—Principles and Procedures. International Organization for Standardization, Geneva, Switzerland.

ISO, 2006b. ISO 14040:2006 Environmental Management—Life Cycle Assessment—Principles and Framework. International Organization for Standardization, Geneva, Switzerland.

ISO, 2006c. ISO 14044:2006 Environmental Management—Life Cycle Assessment—Requirements and Guidelines. International Organization for Standardization, Geneva, Switzerland.

ISO, 2014. ISO 14046:2014 Environmental Management—Water Footprint—Principles, Requirements and Guidelines. International Organization for Standardization, Geneva, Switzerland.

ISO, 2016. ISO 14021:2016 Environmental Labels and Declarations—Self-Declared Environmental Claims (Type II Environmental Labelling). International Organization for Standardization, Geneva, Switzerland.

ISO, 2018a. ISO 14024:2018 Environmental Labels and Declarations—Type I Environmental Labelling—Principles and Procedures. International Organization for Standardization, Geneva, Switzerland.

ISO, 2018b. ISO 14067:2018 Greenhouse Gases—Carbon Footprint of Products—Requirements and Guidelines for Quantification. International Organization for Standardization, Geneva, Switzerland.

Jolliet, O., 2006. Emergence and Future of Life Cycle Impact Assessment: Good science comes from good people. Int. J. Life Cycle Assess. 11 (special issue 1), 9–10. https://doi.org/10.1065/lca2006.04.004.

JRC, 2006. Supporting Business and Policy Making in Europe With Reference Data and Recommended Methods on Life Cycle Assessment (LCA) for Better Practice in LCA Use and Interpretation. Joint Research Center, European Commission, Varese, Italy. http://ec.europa.eu/environment/ipp/pdf/flyer_lca_0511.pdf. (Accessed 19 May 2019).

Klöpffer, W., 1996. Editorial. Int. J. Life Cycle Assess. 1 (1), 3. https://link.springer.com/content/pdf/10.1007%2FBF02978623.pdf. (Accessed 19 May 2019).

Klöpffer, W., 2008. Life cycle sustainability assessment of products. Int. J. Life Cycle Assess. 13 (2), 89–95. https://doi.org/10.1065/lca2008.02.376.

LCI, 2002. Phase I (2002–2006): Creating a Community. Life Cycle Initiative. https://www.lifecycleinitiative.org/activities/phase-i/. (Accessed 19 May 2019).

LCI, 2017. Life Cycle Initiative 2017–2022 Strategy Document. Life Cycle Initiative.

Mazzi, A., Manzardo, A., Scipioni, A., 2014. Chapter 4 - water footprint to support environmental management: An overview. In: Salomone, R., Saija, G. (Eds.), Pathways to Environmental Sustainability. Methodologies and Experiences. Springer, Switzerland.

Mazzi, A., Toniolo, S., Manzardo, A., Ren, J., Scipioni, A., 2016. Exploring the direction on the environmental and business performance relationship at the firm level. Lessons from a literature review. Sustainability 8, 1200. https://doi.org/10.3390/su8111200.

Mattioda, R.A., Mazzi, A., Canciglieri, O., Scipioni, A., 2015. Determining the principal references of the social life cycle assessment of products. Int. J. Life Cycle Assess. (8), 1155–1165. https://doi.org/10.1007/s11367-015-0873-z.

Meadows, D.H., Meadows, D.L., Randers, J., Behrens, W.W., 1972. The Limits to Growth. Potomac Associates - Universe Books, ISBN: 0-87663-165-0.

Olivetti, E.A., Cullen, J.M., 2018. Toward a sustainable materials system. Science 360 (6396), 1396–1398. https://doi.org/10.1126/science.aat6821.

O'Rourke, D., 2014. The science of sustainable supply chains. Science 344 (6188), 1124–1127. https://doi.org/10.1126/science.1248526.

Qian, G., 2016. What causes more LCA research—subjective values, objective problems, or scientific traditions? Int. J. Life Cycle Assess. 21 (7), 1049–1058. https://doi.org/10.1007/s11367-016-1111-z.

Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin, F.S., Lambin, E., Lenton, T.M., Scheffer, M., Folke, C., Schellnhuber, H.J., Nykvist, B., de Wit, C.A., Hughes, T., der Leeuw, S., Rodhe, H., Sörlin, H., Snyder, P.K., Costanza, R., Svedin, U., Falkenmark, M., Karlberg, L., Corell, R.W., Fabry, V.J., Hansen, J., Walker, B., Liverman, D., Richardson, K., Crutzen, P., Foley, J., 2009. Planetary boundaries: exploring the safe operating space for humanity. Ecol. Soc. 14(2), 32. http://www.ecologyandsociety.org/vol14/iss2/art32/. (Accessed 19 May 2019).

Rosenbaum, R.K., 2017. Selection of impact categories, category indicators and characterisation models in goal and scope definition. In: Curran, M.A. (Ed.), LCA Compendium—The Complete World of Life Cycle Assessment—Goal and Scope Definition in Life Cycle Assessment. Springer, Dordrecht, pp. 63–122. https://doi.org/10.1007/978-94-024-0855-3.

Sanfélix, J., Mathieux, F., de la Rúa, C., Wolf, M.A., Chomkhamsri, K., 2013. The enhanced LCA resources directory: A tool aimed at improving life cycle thinking practices. Int. J. Life Cycle Assess. 18, 273–277. https://doi.org/10.1007/s11367-012-0468-x.

SETAC, 1993. Guidelines for Life-Cycle Assessment: A "Code of Practice" (from the SETAC Workshop Held at Sesimbra, Portugal, 31 March–3 April 1993). SETAC, Brussels/Pensacola.

Singer, S.F., 1970. Reducing the environmental impact of population growth. Science 169 (3951), 1233. https://doi.org/10.1126/science.169.3951.1233.

Steffen, W., Richardson, K., Rockström, J., Cornell, S.E., Fetzer, I., Bennett, E.M., Biggs, R., Carpenter, S.R., de Vries, W., de Wit, C.A., Folke, C., Gerten, D., Heinke, J., Mace, G.M., Persson, L.M., Ramanathan, V., Reyers, B., Sörlin, S., 2015. Planetary boundaries: guiding human development on a changing planet. Science 347(6223), 1259855. https://doi.org/10.1126/science.1259855.

Swarr, T.E., Hunkeler, D., Klöpffer, W., Pesonen, H.L., Ciroth, A., Brent, A.C., Pagan, R., 2011. Environmental Life Cycle Costing: A Code of Practice. Society of Environmental Chemistry and Toxicology (SETAC), Pensacola.

Toniolo, S., Mazzi, A., Simonetto, M., Zuliani, F., Scipioni, A., 2019a. Mapping diffusion of environmental product declarations released by European program operators. Sustainable Prod. Cons. 17, 85–94. https://doi.org/10.1016/j.spc.2018.09.004.

Toniolo, S., Mazzi, A., Mazzarotto, G., Scipioni, A., 2019b. International standards with a life cycle perspective: which dimension of sustainability is addressed? Int. J. Life Cycle Assess. https://doi.org/10.1007/s11367-019-01606-w.

Tseng, M.L., Islam, M.S., Karia, N., Fauzi, F.A., Afrin, S., 2019. A literature review on green supply chain management: trends and future challenges. Resour. Conserv. Recycl. 141, 145–162. https://doi.org/10.1016/j.resconrec.2018.10.009.

UN, 1987. Report of the World Commission on Environment and Development: Our Common Future. United Nations, New York. https://sustainabledevelopment.un.org/content/documents/5987our-common-future.pdf. (Accessed 19 May 2019).

UN, 1992. Report of the United Nations Conference on Environment and Development. Annex I—Rio Declaration on Environment and Development. United Nations, Rio de Janeiro.

UN, 2002. Plan of Implementation of the World Summit on Sustainable Development. United Nations. UN Documents, A/CONF.199/20, http://www.un-documents.net/jburgpln.htm. (Accessed 19 May 2019).

UN, 2015a. The Millennium Development Goals Report. United Nations, New York.

UN, 2015b. Transforming our World: The 2030 Agenda for Sustainable Development, a/RES/70/1. United Nations, New York.

UNEP, 2011. Towards a Green Economy: Pathways to Sustainable Development and Poverty Eradication. Nairobi, Kenya, United Nations Environment Program, ISBN: 978-92-807-3143-9.

US EPA, 1974. Resource and Environmental Profile Analysis of Nine Beverage Container Alternatives. Final Report. EPA/530/SW-91c, U.S. Environmental Protection Agency, Washington, USA.

Weidema, B.P., 2018. The social footprint—a practical approach to comprehensive and consistent social LCA. Int. J. Life Cycle Assess. 23 (3), 700–709. https://doi.org/10.1007/s11367-016-1172-z.

Wolf, M.A., Pennington, D., Bersani, R., 2006. The European Platform on LCA—towards good practice in LCA and a system of high-quality LCI databases for industry and government. In: International Conference on Green and Sustainable Innovation 2006. JRC, Energy Management and Conservation Center, p. 116. http://publications.jrc.ec.europa.eu/repository/handle/JRC36718. (Accessed 19 May 2019).

Wulf, C., Werker, J., Zapp, P., Schreiber, A., Schlör, H., Kuckshinrichs, W., 2018. Sustainable development goals as a guideline for Indicator selection in life cycle sustainability assessment. Procedia 69, 59–65. https://doi.org/10.1016/j.procir.2017.11.144.

Zamagni, A., Amerighi, O., Buttol, P., 2011. Strengths or bias in social LCA? Int. J. Life Cycle Assess. 16 (7), 596–598. https://doi.org/10.1007/s11367-011-0309-3.

Zamagni, A., 2012. Life cycle sustainability assessment. Int. J. Life Cycle Assess. 17 (4), 373–376. https://doi.org/10.1007/s11367-012-0389-8.

CHAPTER

2

Sustainability, sustainable development, and business sustainability

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2.1 Sustainability and sustainable development: Main concepts and approaches

The world is nowadays unsustainable (WBCSD, 2010; UN News Centre, 2012). KPMG International (2012) identified 10 "mega-forces" (climate change, energy and fuel, material resource scarcity, water scarcity, population growth, wealth, urbanization, food security, ecosystem decline, and deforestation) that challenge our world and, if not adequately and urgently tackled, could undermine human life and the environment in the next 20 years. Oxfam International (2014) affirms that social inequalities are rising, despite the development of emerging countries. The Intergovernmental Panel for Climate Change (IPCC, 2015) states that extreme events related to climate change are increasing in frequency. These are only some examples of the dangers humanity is facing today and which will have to deal with in the future.

Sustainability has probably become the most used approach to development in the last three decades, entering the discourse in numerous disciplines at all levels. Researchers, political institutions at all levels, businesses, and civil society organizations, all give their own interpretation to the concept. As a consequence, sustainable development has several meanings and definitions and there is no agreement about it (Mebratu, 1998; Giddings et al., 2002). This chapter will only provide the reader with a brief overview on the origins and main characteristics of the concept.

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2.1.1 The Brundtland definition

The term sustainability firstly appeared within the report of the World Commission on Environment and Development "Our Common Future" in order to describe a new approach to development which should "meet the needs of the present without compromising the ability of future generations to meet their own needs" (Brundtland et al., 1987, p. 41). This new approach is derived from a progressive acknowledgement of the insufficient progress made to defeat poverty and to ensure well-being to all human beings as well as of the environmental boundaries given by a planet with finite resources. As a matter of fact, that was a time of oil crises and raw material price increase, the fall of the Berlin wall, and the end of the decolonization process with the consequent questioning of the bipolar/triple pole world division, the Sahel environmental crisis and the great famine, the raise of consciousness about finite fossil resources, nuclear catastrophes, and the fruits of unequally distributed growth.

Therefore, the ultimate goal of Sustainable Development is assuring wellbeing to the whole global population in the present (intra-generational equity) and in the future (inter-generational equity) at the same time. Consequently, the concept implicitly underpins the need for a long-term perspective. The World Commission on Environment and Development also acknowledges the inter-dependence between social, economic and environmental aspects. Sustainability is thus an invitation to an interdisciplinary approach while coping with development issues.

Except for these theoretical inferences, the Brundtland definition has been largely criticized for being ambiguous. According to Wackernagel and Rees, this was done on purpose in order to be widely and transversally accepted (1996 as cited in Giddings et al., 2002) through interpreting the concept in the most diverse ways (Pearce et al., 1989).

Development is seen as a broader concept than economic wealth and growth. As Sen (1999) states, development is a set of conditions that allow a subject to realize its potential: any person can function if she/he has the means (physical, psychological, social, relational, …) that release her/his ability to function. Although the concept of sustainable development has often been summarized in a the three-pillar approach (normally represented as a three-intersected-ring sector), for many authors the idea behind it is even wider: Sustainable Development is considered to be holistic (Pike et al., 2007). Thus, a three-dimensional approach does not allow us to see the real potential for inclusiveness of the concept. Development is then brought by concomitant progress of integrated dimensions.

Some of the major concepts given above can be summarized by Dyllick and Hockerts (2002, p. 1), defining sustainability as the "societal evolution towards a more equitable and wealthy world in which the natural environment and our cultural achievements are preserved for generations to come". A clear image is also given by Raworth's Doughnut Economics (2017), which represents a "sage and just space for humanity" in a doughnut shape where the outside boundary is made by environmental constraints and sustainability challenges and the inside one by 11 social elements based on fairness and the wellbeing of humanity; keeping producing and consuming within the doughnut means developing an inclusive and sustainable economy.

2.1.2 Main approaches to sustainable development

The United Nations Handbook of National Accounting (UN et al., 2003) identifies three different approaches to sustainable development: three-pillar, ecological, and capital approaches.

The Capital approach to Sustainable Development (Daly and Cobb, 1989; Pearce et al., 1989; Pearce and Turner, 1990) has been developed in order to make sustainability conceptually closer to the business sector and raise its attention on the issue (Goodwin, 2003). Ostrom and Ahn (2003) define capital as a set of resources that could be immediately consumed, but are instead allocated to future wealth generation. In this case, the economic rule of nondeclining capital, or capital maintenance (Victor, 1991), has been transposed to the concept of sustainable development. The Handbook of National Accounting (United Nations, 2003) defines sustainable development as one that ensure a nondeclining pro capita national wellness by replacing or conserving the sources of that wellness, that is to say different capital typologies' stocks. Several capital classifications exist identifying four, five, or six capital typologies considering financial, man-made, social, human, cultural, and natural capital (Forum for the Future, n.d.; Goodland, 2002; Goodwin, 2003; Hallsmith and Lietaer, 2011; International Integrated Reporting Council, 2013). Development is then sustainable if the sources of wealth (man-made, social, cultural, human, financial, and natural capital) are maintained rather than depleted or degraded in order to leave to the future generations a capital stock able to deliver the same wellbeing current generations have access to (Pearce and Atkinson, 1998; Forum for the Future, n.d.; Goodwin, 2003; Hallsmith and Lietaer, 2011).

An open discourse exists concerning the substitutability or complementarity between capitals, and especially between the natural and the other forms of capital (UN et al., 2003). On the one side, according to the weak sustainability idea, the overall capital has to be maintained over time, while the different capital types can be substituted between them. On the other side, the strong sustainability approach affirms capital nonsubstitutability (Pearce and Atkinson, 1993; UN et al., 2003; Dietz and Neumayer, 2007). This derives from the fact that different capitals are responsible for delivering different functions (Ekins et al., 2003), and some capitals actually have value only if combined together (complementarity). However, Turner (1993 as cited in Ekins et al. 2003) identifies some middle ways between the two rules, presenting four positions. On the one side, very weak sustainability consists in complete capital substitutability, whereas weak sustainability admits substitutability between natural and manmade capital with minor exceptions. On the other side, Strong sustainability affirms that substitution between natural and manmade capital could be importantly undermined by the irreversibility of certain natural capital depletion or deterioration, or by the existence of critical natural capital stocks delivering unique functions indispensable for life. Moreover, the depletion of certain natural capital stocks could have no impact until a given threshold, showing nonlinearity after passing it (Rockstrom et al., 2009; Dyllick and Hockerts, 2002). Given the uncertainty consequent to an incomplete scientific knowledge concerning nature and society-environment interactions, the precautionary principle is thus supported by numerous authors and institutions (Pearce and Turner, 1990; United Nations, 1992a, b). Lastly, very strong sustainability proposes complete nonsubstitutability between capitals but it is not taken into concrete consideration, whereas the most likely approaches seem to be weak and strong sustainability. Fig. 2.1 represents the different capital substitutability scenarios based on a very weak, strong, or very strong sustainability approach. With a very weak sustainability approach, there is a complete substitutability among different capitals (namely natural, social, cultural, human, financial, and manmade). With a strong sustainability model the concept of critical capital stock is introduced. Substitutability among different capitals can be only partially accepted until specific thresholds of critical capital stocks. Depleting critical capital stocks can be risky in terms of

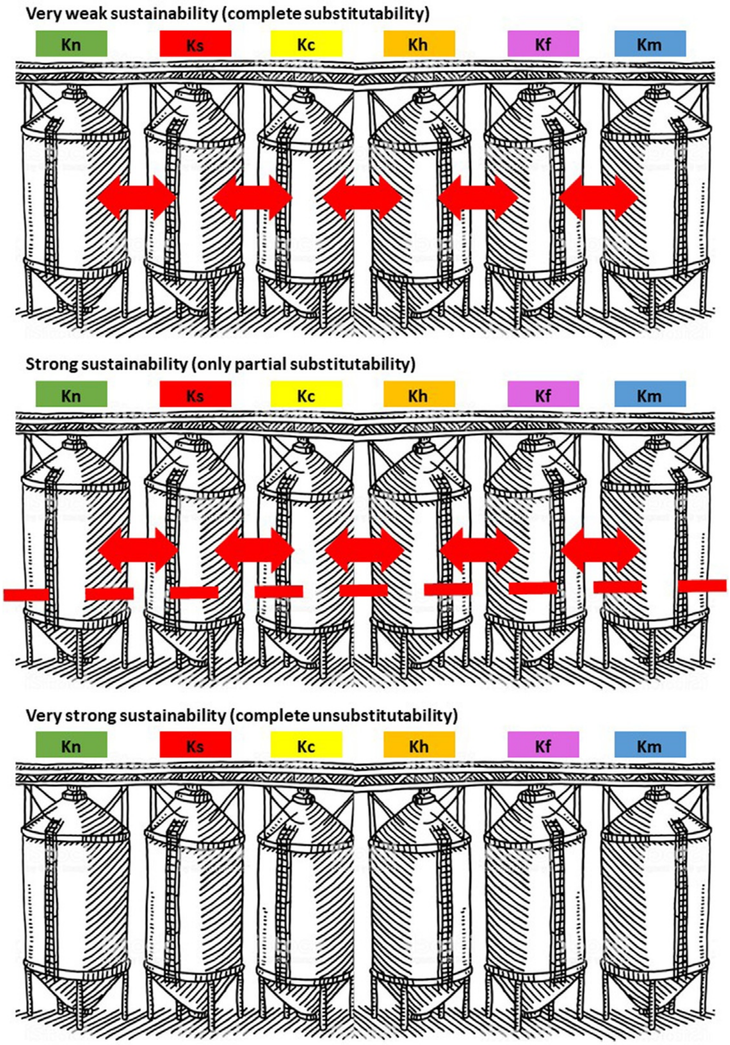

FIG. 2.1 The figure represents the different capital substitutability scenarios based on a very weak, strong, or very strong sustainability approach. *(Credits: Marta Avesani)*

irreversibility of that specific form of capital regeneration. The very strong sustainability approach is the more prudent model not allowing any substitutability.

The three-pillar approach is the most widely known one. It underlines the importance of addressing the economic, environmental, and social dimensions of sustainable development simultaneously because of their urgency, inter-dependency, and interconnection (cf. Giddings et al., 2002 for a graphic representation). As a consequence, according to this view, the three focuses are equally important and must be equally weighed. Some weak points are also acknowledged. First of all, a risk is supposed for a technical fix approach in coping with sustainable development issues (Giddings et al., 2002); that is to say that the model, though showing the interconnection about different dimensions, is unlikely to tear down the "silo thinking" in problem-solving. Secondly, giving the same importance to the three dimensions leads to the idea of trade-offs among sectors, in line with a weak sustainability approach, allowing substitutability among different capital typologies.

The ecological approach sees the environment as the priority, since the social and economic systems cannot exist without the life-support services and resources provided by nature. In that sense, compared to the three-pillar approach this one is more coherent, with a strong sustainability approach affirming a nonsubstitutability of critical stocks of natural capital (cf. Giddings et al., 2002 for a graphic representation).

2.2 Sustainability in the global public agenda

After the first appearance of the concept of Sustainable Development within the World Commission on Environment and Development in 1987, the first United Nations Conference on Environment and Development (UNCED) took place in Rio de Janeiro in 1992, also known as the Rio Conference or Earth Summit. This global meeting led to three main outputs: (i) the adoption of the Rio Declaration; (ii) the adoption of a Sustainable Development Programme called Agenda 21; (iii) the adoption of the United Nations Framework Convention on Climate Change (UNFCCC).

The Rio Declaration (United Nations, 1992a) stated the centrality of present and future human beings in any concerns and the need of an integration of the environmental protection principle within the development process, while confirming the right of sovereign states to exploit their own resources as long as they commit to warning each other about any dangerous activities. In addition, it linked environmental and poverty issues as interdependent and concerning all countries, with common though differentiated responsibilities. Lastly, it affirmed the need to stop nonviable production and consumption patterns and to encourage the viable ones. Compared to previous international policies, three new approaches were suggested for taking action: (i) active engagement and participation of citizens; (ii) identification and engagement of stakeholder groups to be engaged, such as NGOs, labor unions, local authorities, companies, the scientific community, youth groups, women, and local communities; (iii) an integrated vision of the interdependence among peace, development, and the environment.

Agenda 21 (United Nations, 1992b) was the first internationally agreed sustainable development program, which put into practice the new approaches proposed within the Rio Declaration. For instance, the subsidiarity principle is applied, envisioning a participative implementation of the agenda at the local level, as the most appropriate in terms of education, mobilization, resource exploitation, planning and economics, and social and environmental infrastructure maintenance.

Lastly, the UNFCCC (United Nations, 1992c) is the first United Nations attempt to clarify what climate change is and how to cope with it. Three main principles are proposed through this framework: (i) precaution; (ii) common though differentiated responsibilities; and (iii) right to development. The framework is not binding, but it provides countries the possibility to engage further internationally. The Kyoto Protocol (United Nations, 1998), in effect from 2005 to 2012, is the first example of this kind.

During the United Nations Conference on Sustainable Development (Rio+20) in 2012, the development of a set of goals and targets aiming at promoting sustainable development globally was included in the nonbinding document resulting from the conference called "The Future We Want" (United Nations, 2012). The Agenda 2030, based on 17 Sustainable Development Goals (SDGs), was actually adopted in 2015 by the United Nations General Assembly. Reasonably, these new goals were thought to substitute the previous Millennium Development Goals (MDGs) to be met by 2015. The former set of goals was not successful everywhere for different reasons. Firstly, the focus was given to development issues, treating the environmental dimension as a separate goal without any strong interconnection with the social one. Secondly, MDGs mainly targeted the so-called developing countries without taking into any account the strong interdependence between them and the wealthier countries for what concerns the socioeconomic and environmental dimensions. Lastly, MDGs were operationalized at the institutional and technical level, without any engagement of other involved stakeholders such as citizens and local communities or companies. On the contrary, SDGs were developed trying a different approach based on the weak points, which influenced the failure of the previous agenda. SDGs are global goals; every country in the world should work to meet them. Moreover, the silo thinking is overcome by a 17 interlinked goal structure, with goals that combine environmental, social, and economic aspects at the same time. Lastly, a multistakeholder approach is considered the best one to tackle complexity. For this reason, every level and stakeholder is called to contribute to the global goals: from states to local administrations, from NGOs to citizens, from companies to social businesses, from research institutes and universities to schools.

2.3 Business sustainability

Business sustainability has a paramount role in the context of global unsustainability. Business responsibility on this issue is largely supported by nongovernmental organizations (NGOs), such as Oxfam International. For instance, the organization affirms that the business practice of tax dodging constitutes an unjust advantage for big companies over small and medium enterprises, and deprives countries of an important income to tackle poverty and inequalities. Moreover, businesses are often responsible for poverty wages (Oxfam International, 2014).

The recent encyclical letter on the environment by Pope Francis also states that negative environmental impacts such as pollution, water shortage, natural resource depletion, and deforestation account businesses as one of the main responsible parties. It also invites companies to follow their vocation of serving the common good through positive value creation (Pope Francis, 2015).

In 2015, the IPCC affirmed that business-as-usual should be urgently left forever for global temperature increase to stop below 2°C relative to preindustrial levels, in order to prevent irreversible climate changes and unknown scenarios. The special IPCC report published in 2018, while stating that 1°C temperature growth above preindustrial level has already been reached, drew the pathway for it to be halted at a 1.5°C increase by 2030. SDG 12 "Sustainable Production and Consumption" is considered coherent with the identified pathway.

The need for a change in the business model was also supported by former United Nations Secretary-General Ban Ki-Moon (2014) in his report on the post-2015 agenda, and indirectly implied by the new 2030 Agenda through goals and targets such as #12 Sustainable Production and Consumption, target 12.C:

> Rationalize inefficient fossil-fuel subsidies that encourage wasteful consumption by removing market distortions, in accordance with national circumstances, including by restructuring taxation and phasing out those harmful subsidies, where they exist, to reflect their environmental impacts, taking fully into account the specific needs and conditions of developing countries and minimizing the possible adverse impacts on their development in a manner that protects the poor and the affected communities.

and #8 Economic Growth and Decent Work, target 8.4:

> Improve progressively, through 2030, global resource efficiency in consumption and production and endeavor to decouple economic growth from environmental degradation, in accordance with the 10-year framework of programs on sustainable consumption and production, with developed countries taking the lead.

In recent decades, companies themselves have started to acknowledge their responsibility in global sustainability issues due to their negative impacts on society and the environment together with the need for the business sector to be part of the solution in order to actually shift from business-as-usual to a sustainable world where every citizen live well within planetary boundaries (WBCSD, 2010). The needed change does not encompass only the business model but also broader economic system and consumption patterns (WBCSD, 2010; Dyllick and Muff, 2015). As a matter of fact, Gray (2010) and Gray and Bebbington (2000) accuse capitalism of contributing to unsustainability, since it is based on short term financial return on investment, consumerism, and greed. As a consequence, Townsend (2015) envisages a shift from capitalism to "capitalism 2.0," or sustainable economy. This is an economic system no longer based on resources exploitation and financial revenues but rather on prosperity for both people and firms within planetary limits.

The need for business sustainability derives from the acknowledgement of a gap existing between corporate activities and global environmental and social performances. From an environmental point of view, Rockstrom et al. (2009) states that industrialization has brought the world into a new era where human activities are responsible for major changes in the environment, which could have dangerous impacts in the future. The Millennium Ecosystem Assessment (MEA, 2005 as cited in Dyllick and Muff, 2015) demonstrated that 15 out of 24 ecosystem services have been deteriorated in the past 50 years because of human actions, while only four are in better condition. According to the United Nations Financial Initiative (UNEP-FI, 2011), in 2008, the human race cost nature \$6.6 trillion, corresponding to 11% of the world gross domestic product for that year. Similarly, the 3000 world's biggest publicly traded companies were responsible for \$2.15 trillion of environmental cost. The German footwear company, Puma, in partnership with Trucost, has been the first firm to account and monetize its hidden debt to nature for all the services the environment provides for its business activities. In 2010, the company should have paid nature for 8 million Euros, 145 million if external partners in the supply chain were also included, though the latter normally serve more than one company at a time (Puma, 2011).

Although environmental issues and impacts receive a great part of the global attention, the business world has also a relationship with the social dimension of sustainability. Azapagic and Perdan (2000) state that industry is recognized both to degrade the environment and deplete natural resources and to contribute to societal development and prosperity. For instance, business provides income, training, and social security to a large number of employees all over the world (WBCSD, 2002). Companies have also a responsibility for the safety, health, and environmental conditions of the places where they operate (KPMG International, 2014). Moreover, according to Gray and Milne (2002), social disparities are a congenital component of capitalism since they split the world into capitalists and workers.

According to Paul Polman (2015), Unilever's former Chief Executive Officer (CEO), social and environmental goals have been fixed by the 75% of the largest firms. However, although corporate commitment to sustainable development has become mainstream in the last decades, this has not effectively contributed to the reduction of the human environmental footprint or of global social problems (Dyllick and Muff, 2015). On the contrary, these appear to be exacerbated.

For all these reasons, the business sector is supposed to have a responsibility in the path toward global sustainability. In 1992, the World Business Council for Sustainable Development (WBCSD), a CEO-led organization of forward-thinking companies, was created to represent the business voice at the Rio Earth Summit. According to its founder, Stephan Schmidheiny, a Swiss entrepreneur and philanthropist, business has an unavoidable responsibility in sustainable development ("WBCSD", n.d.). Twenty-nine WBCSD members have recently worked on Vision 2050, envisaging nine billion people living well and within the limits of the planet (WBCSD, 2010, p. 4). They acknowledged the impossibility to reach the vision with a business-as-usual attitude and the need to decouple economic growth from resource depletion and environmental degradation through radical changes in governance, economic frameworks, and business and human behaviors. A similar belief is also supported by the Council of Academies of Engineering and Technological Sciences, which states that industrial processes and resource management should be modified in order to bring about Sustainable Development (Azapagic and Perdan, 2000).

2.3.1 Evolution of the concept

The previous section discussed the relevance of the Business Sustainability concept. However, the above expressed arguments are the fruits of at least two decades of discussion, research and debate. Therefore, this section will provide the reader with a short overview on its origins and evolution.

The acknowledgement of the Earth as a planet with finite resources and of human impact on the environment appeared long before 1987, in the 1960s (Elkington, 2004). Similarly, during the 1970s a rise in the attention to social issues was observed, though it soon disappeared during the 1980s and 1990s, only to come again with the new century with a new consciousness linking interdependently together the environmental, social, and economic pillars (Gray and Bebbington, 2000). Therefore, a first phase of environmental concern was raised in the 1960s based on environmental regulation by governments and a passive, compliant behavior by businesses (Elkington, 2004). However, this reactive approach by businesses was shown to lack long-term viability because of its high costs. Business risk aversion and cost minimization brought firms to act in a more active way (Azapagic and Perdan, 2000).

Elkington (2004) identifies a new phase taking place over the 1970s and the 1980s, characterized by a moment of market liberalizations and privatizations. During this period business tried to invert the legislation to its favor. At the end of the 1980s, the rise of the sustainability concept, with the Brundtland report in 1987, and several industrial accidents, gave birth to a third phase. Suspicion concerning business behavior and reporting legislation or voluntary disclosure support by governments drove business toward a more competitive behavior based on being "green" (Kolk, 2003; Elkington, 2004). It is in this period that the WBCSD was created in order to bring the business voice to the 1992 Earth Summit in Rio. During the Conference, the organization, together with the International Chamber of Commerce, acted to shift the public attention from the transnational companies' impact on the environment and the need for new forms of business social accountability. On the contrary, a role for business as part of a sustainable solution is supported, arguing the coincidence between business sustainability and good business practices (Gray and Bebbington, 2000; Gray and Milne, 2002). Therefore, business shows now a pro-active approach driven by the rising idea of the existence of a business case for sustainability (Azapagic and Perdan, 2000) and thus distancing itself from old corporate social responsibility (CSR), mainly focused on reputation (KPMG International, 2014).

A last phase took place at the end of the 20th century. Civil society raised its voice against international organizations, to which a responsibility for sustainable development is ascribed (Kolk, 2003; Elkington, 2004). It is acknowledged that sustainable development cannot be achieved through disconnected initiatives but through a new form of governance and strategy at the global level and within business. Sustainability is embedded in business strategy and it is communicated to external stakeholders via reporting (Azapagic and Perdan, 2000).

The described phases show an initial business diffidence toward business sustainability; however, according to Little (n.d. as cited in Giddings et al., 2002), the relevance of Sustainable Development is now recognized by 95% of the largest firms in Europe and the United States.

2.3.2 Many approaches to business sustainability

Having discussed its relevance and its origins and evolution, the discourse is now ripe to define business sustainability in its most common form, as well as to put the emphasis on the vast amount of different interpretations related to the concept.

The most widely used definition of business sustainability is basically an adaptation of the sustainable development definition to business. The International Institute for Sustainable Development (IISD, 1992 as cited in Labuschagne et al., 2005, p. 1) defines business sustainability as the adoption of business strategies and activities that meet the needs of the enterprise and its stakeholders today while protecting, sustaining, and enhancing the human and natural resources that will be needed in the future. Dyllick and Hockerts (2002) specify that the needs to be met are of direct and indirect stakeholders and express the futurity principle in terms of future stakeholder need rather than of solely natural resources. A slightly different, more business-focused definition is given by Niţă and Ştefea (2014, p. 2), who describe Business Sustainability as a business strategy that drives long-term corporate growth and profitability by mandating the inclusion of environmental and social issues in the business model. These different statements, though with some commonalities, give in advance an idea of the varied meanings Business Sustainability can stand for. Some of these different interpretations will be revised here below. However, some authors warn about the presence of misleading meanings of business sustainability (De Simone and Popoff, 1997 as cited in Dyllick and Hockerts, 2002). For this reason, the overview comprehends also some of the main critiques to the different concept directions.

2.3.2.1 *Triple bottom line*

The concept of triple bottom line was created by John Elkington, an expert in corporate responsibility and sustainable development, in 1997. It rises from the need for a new definition of added value, which goes beyond economic value and comprehends the environmental and social costs and benefits that business brings to society. The idea is also known as "3P", standing for "people, planet, profit", or "win-win-win" strategy, since it tries to combine together social, environmental, and economic stakes supporting the ability of business to manage them all (Elkington, 2004).

On the triple bottom line, Dyllick and Hockerts (2002) affirm that economic Business Sustainability, which requires contemporaneously enough liquidity and financial returns for investors, does not satisfy long-term sustainability alone. The satisfaction of ecological and social business sustainability is also needed. Additionally, the two authors specify that ecological business sustainability binds firms to use natural resources at a degree below their recreation or substitute development and to produce waste at a level below the ecosystem absorption capacity. Similarly, social business sustainability is related to human and social capital enhancement from the company toward the different stakeholder groups. Moreover, despite the presence of trade-offs between the groups, the community can count on a common value system.

Elkington identifies seven revolutions for moving to sustainable capitalism including:

(i) free and competitive markets;
(ii) a global shift in human and societal values;
(iii) transparency through global reporting and disclosure;
(iv) life-cycle technology making firms responsible for the product "from cradle to grave";
(v) partnerships with different organizations based on cooperation and mutual trust;
(vi) a combination of two apparently opposite time conceptions: one as fast as possible to manage properly a global market and one based on a long-term time horizon essential for sustainability; and
(vii) a corporate governance including stakeholders.

The triple bottom line approach has been criticized by several authors. Firstly, Gray and Milne (2002) warn about the fact that, in the case of trade-off between the three bottom lines, the financial aspect is given more importance than the others. It means that environmental and social issues are subordinated to their ability to bring business profit. Nevertheless, according to some authors, corporate economic sustainability should always be prioritized since, if a firm is not able to stay in business, it cannot even contribute to the external societal well-being (Labuschagne et al., 2005). Furthermore, McDonough and Braungart (2002) criticize the triple bottom line as an "end-of-the-pipe" measure for business sustainability, since it provides companies with strategies to minimize their negative impact instead of designing a sustainable process and product from the beginning, avoiding negative effects at all.

2.3.2.2 *Corporate social Responsibility*

CSR is a tool adopted by numerous companies in order to take responsibility for the detected social and environmental impacts. It normally goes beyond regulation compliance (Corporate Social Responsibility, 2007) and, therefore, it can be considered a sign of business pro-activeness. Nevertheless, CSR is meant in very different ways by companies. Pless et al. (2012) identifies two different approaches to CSR. The first one is "instrumental". It commits companies to CSR only if economically profitable. Whereas the second "multifaceted" one, aims at creating shared value for both investors and stakeholders. This second approach is the one supported by the European Commission.

According to the Prince of Wales Institute, corporate responsibility should include responsible core business activities, philanthropic investments, but also business involvement in public-private partnerships (Nelson, 2002 as cited in Labuschagne et al., 2005). Labuschagne et al. (2005) splits the "corporate responsibility strategy" into two main components: societal and operational initiatives. The first one comprehends corporate social investments related to external philanthropy, while the second one is related to business core activities. The authors underline that business sustainability performance should be assessed based on sustainability initiatives (environmental, social, and economic) related to the core business activities. This is a really relevant elucidation given that a lot of businesses tend to confuse business sustainability with their contributions to external social investments and philanthropic causes mainly enhancing their image and reputation rather than their actual operations. This is an argument supported by Porter and Kramer (2011), who highlight the risks of investing in initiatives that have almost nothing to do with the business core. In fact, there is a risk for these initiatives to be quitted as soon as they do not bring business benefit anymore. Showing limited engagement with a start and an end point, it is thus difficult to maintain sustainability in the long term. However, the two authors are criticized by Crane et al. (2014) who, though recognizing that CSR literature seldom goes beyond the business case for CSR, argue the existence of a "strategic CSR," which embeds initiatives within the business strategy in order to benefit the sustainability of the firm's core activities.

A reductionist judgment on CSR initiatives seems to be given by KPMG as well when writing:

> This investment [in people, communities and the environment] entails far more than corporate philanthropy, CSR projects or "green" initiatives—worthy and important though these may be. To do well in today's business environment, you increasingly have to measure, understand and pro-actively manage the value you create, or reduce, for society and the environment as well as for shareholders. ***(KPMG International, 2014, p. 4)***

Lastly, CSR has been criticized by Young and Tilley (2006) for referring only to socio-efficiency, that is to say, social impact minimization and social benefit maximization in relation to the created business value (Dyllick and Hockerts, 2002), instead of considering also socio-effectiveness, defined as a continual societal positive impact.

However, as mentioned at the beginning of this section, for some authors and organizations, CSR should not be a bolt-on set of initiatives put in force by companies to serve their business case. On the contrary, it should focus on shared value creation. The critiques to CSR by Porter and Kramer (2011) bring them to the elaboration of the "creating shared value" theory willing to reshape the relationship between business and society in order to ensure prosperity for both subjects. The theory suggests economic value creation by creating societal value through three different strategies. These are:

(i) rethinking products and markets based on society's needs and societal benefits;
(ii) transforming the value chain through efficiency measures and stakeholder relationship management; and
(iii) investing in local cluster development in order to strengthen business partnerships and the link between business and society (Porter and Kramer, 2011).

Nevertheless, this theory is partly criticized by Crane et al. (2014) who affirm that, though it represents a step forward involving stakeholders as value beneficiaries, corporate self-interest is not discussed and stakeholders would always come after business profit. As an alternative, they propose the adoption of multistakeholder processes as a true social perspective, where business is but one stakeholder among others, in order actually to walk toward the common good of society. The importance of cooperating in partnership with external stakeholders is also supported by Pfitzer et al. (2013) and Zimmermann et al. (2014) at all the process stages for firms willing to create shared value for business and society. In fact, companies with an insufficient comprehension of societal needs can rely on other actors in order to gain insight on their social purpose. Moreover, they can share innovation risks through the use of incubators and activating partnerships (Zimmermann, et al., 2014) and hybrid innovative business structures. Similarly, monitoring and assessment need an external view in order to catch the shared value of the enterprise (Pfitzer et al., 2013).

However, Porter and Kramer (2011) were not the first ones to focus on a broader interpretation of value creation. In fact, in their answer to Crane et al. (2014), they acknowledge the contribution of Emerson (2003 as cited by Dyllick and Muff, 2015) and his "blended value" concept, inviting businesses to seek profit, social, and environmental goals at the same time. Nevertheless, Porter and his colleague take the distance from this theory affirming that it is not meant to solve societal problems like theirs is designed for (Porter and Kramer, 2014).

In accordance with the multifaceted interpretation of CSR aiming at shared value creation and willing to highlight the distance taken from an instrumental use of the concept, some companies recently started to use "corporate sustainability" instead. The United Nations Global Compact, a voluntary initiative for business sustainability, based on corporate CEOs committed to bring about sustainability principles and UN goals (About the UN Global Compact, n.d.), defines it as the business way of contributing to sustainable development global challenges. It constitutes in moving their means and skills for economic, social, environmental, and ethical value creation, both for business and for society in the long term. This implies the incorporation of sustainability principles into core business strategies, acknowledging business transformative power (UNGC, 2013).

The presented critiques to instrumental CSR mainly propose a continuous business commitment to the outside by delivering positive value. Interestingly, Moneva et al. (2006), while agreeing on the reductionism of CSR as a set of initiatives inside the organization, points out its distance from sustainable development. In fact, the latter has a normative intention leading to deep global systemic changes, whereas the former acts within the status quo.

2.3.2.3 Eco-efficiency

According to McDonough and Braungart (1998), the concept of eco-efficiency, though not with this name, can be dated back to Henry Ford and his efforts to achieve resource minimization and recycling in the assembly line. Always indirectly, it was used in the Brundtland report (1987), envisaging more resource efficiency and less pollution and minimization of the irreversible negative impacts to society and the environment. However, its formal appearance takes place in 1991 by the just-born WBCSD.

Eco-efficiency is a concept linking together the environmental and economic dimensions and it is defined as doing more with less (McDonough and Braungart, 1998, p. 2), as a firm's economic profit in relation to its environmental impact (Schaltegger and Sturm, 1990, 1992, 1998 as cited in Dyllick and Hockerts, 2002) or as maximizing value while minimizing impact (WBCSD, 2002). However, Schmidheiny and Stigson (2000), within their report on eco-efficiency for the WBCSD, argue that these are reductionist views and invite us to see eco-efficiency also as a concept that should prompt us toward new production solutions not only within the firm's framework, but also along the whole value chain. Nevertheless, eco-efficiency is a largely criticized concept. First of all, Welford (1997 as cited in Dyllick and Hockerts, 2002) and Schaltegger and Sturm (1990, 1992, 1997 as cited in Dyllick and Hockerts, 2002) point out that eco-efficiency is often used by businesses as a synonym of sustainability, whereas this is but one measure among many of a broader concept.

Secondly, Gray and Milne (2002) argue that the absolute impact of each business on every resource base should be aggregated in order to actually measure for environmental sustainability. As a matter of fact, a company who can minimize its environmental damages is only relatively sustainable, whereas, in absolute terms, the amount of damage produced by all businesses together could still be unsustainable for the planet. Gray (2010) defines sustainability as a systemic concept, which has to be considered at the eco-systemic level. The need for absolute thresholds is also supported by Dyllick and Hockerts (2002), who affirm that irreversibility, nonlinearity, and nonsubstitutability principles applied to natural capital depletion make it unsustainable to only rely on eco-efficiency.

Young and Tilley (2006, p. 3) summarize this critique, defining eco-efficiency as an insufficient illusion of short-term relative improvements for a business willing to be truly sustainable. This illusion decreases the feeling of culpability and worry about the future without actually solving the problems since, despite the relative improvements, resources and nonrenewable energy sources continue being unsustainably used and ecosystems damaged, and what does decrease is only the rate of depletion and deterioration (McDonough and Braungart, 1998). Additionally, Gray and Bebbington (2000) argue the ineffectiveness of eco-efficiency measures also comparing sustainability indicators at 5 years of distance from the first eco-efficiency initiatives taken at the Rio 1992 conference. Their results showed that these indicators worsened during that 5 year span. Furthermore, Gray and Milne (2002) doubt that capitalistic businesses would be really interested in broadening efficiency to effectiveness measures, meant as an absolute decrease in business, social, and environmental impacts, for two main reasons. Firstly, this would probably imply a decrease in production undermining the concepts of consumerism and, ultimately, of growth. Secondly, social disparities are a fundamental capitalistic element.

Lastly, eco-efficiency, relating together only the economic and environmental dimensions, does not take into account social aspects, thus forgetting an indispensable and integral part of sustainability. This last argument is relatively common, as many businesses mean sustainability as only related to the environment. This is partly due to the difficulties in measuring the majority of social impacts.

2.3.2.4 True business sustainability

In recent years, a step forward in the concept of business sustainability was made by several authors (i.e., McDonough and Braungart, 1998, 2002; Dyllick and Hockerts, 2002; Young and Tilley, 2006; Dyllick and Muff, 2015). This concept evolution will be referred to with the expression "true business sustainability," used by Dyllick and Muff (2015). A question immediately emerges: why "true"? Was business sustainability not true? Where does the need for "true" business sustainability come from?

In fact, the adjective "true" has not only been recently used by the aforementioned authors, but companies started to apply it to other concepts such as "true cost," "true price," "true earning," "true profit," "true progress," and "true value." All these concepts were created based on the recognition that companies have impacts on society and on the environment that are not taken into account. These impacts correspond to business externalities since, being difficult to be measured, they do not have a price and they are thus considered outside the market. The ignorance of externalities leads to a narrow definition of value creation, which is challenged by sustainable value creation, implying the account of all costs and benefits (Fatemi and Fooladi, 2013). As a consequence, in recent years, several business accounting organizations and other business-related institutions started to find ways to measure, through monetization (true price), business impacts (true costs) to society and nature. This was done in order to internalize externalities and assess firms' true earnings or true profit and ultimately, their true value—that is, a value benefiting both shareholders and society (WBCSD, 2010; True Price Foundation, 2012; KPMG International, 2014). It has to be mentioned that externalities refer in general to all of what is not accounted within the corporate statement, positive or negative, meaning that externalities could also include hidden benefits given by the firm to the outside (KPMG International, 2014).

Trucost, a company helping businesses to identify their hidden costs and impacts, applies this approach in terms of risk minimization for business ("What we do", n.d.). Nevertheless, the True Price Foundation (2012) seems to be more vocal in terms of expressing the potential of externalities internalization. Firstly, monetization means fostering sustainability through the use of markets; secondly, internalization of externalities creates transparency, as it is widely questioned by consumers. Thirdly, transparency can turn into more profitability for businesses, implying more competitiveness and license to operate; and lastly, the whole process envisages multistakeholder cooperation instead of conflict leading to unpredictability.

Although these new concepts are bringing business sustainability a step forward, the abovementioned authors researching on true business sustainability seem to mean something deeper and broader than mere internalization of externalities, which could bring business sustainability to a new level. In this case, the "true" addition seems to refer to an implicit critique to the reductionist approach to business sustainability, which has characterized the discourse and the initiatives up to now.

Dyllick and Muff (2015) highlight the lack of evidence concerning an actual benefit of Business Sustainability initiatives to Sustainable Development. The assimilation of sustainability to eco-efficiency is, according to Gray and Bebbington (2000), a signal that the business-as-usual growth and profit maximization are not questioned and alleviating global issues is preferred to solving them (McDonough and Braungart, 1998). Starting from these premises, Dyllick and Muff create the concept of true business sustainability, referring to a business that designs its existence around its contribution to solving societal and environmental issues. It is also important to mention that they fully acknowledge the importance of all the steps made in terms of business sustainability concept evolution and practices until the present moment, and they see true business sustainability as a final goal to reach through a path that sees businesses at different levels of awareness and capability for change.

The two authors gave birth to the Business Sustainability Typology Framework, showing four different business models, three of which present different degrees of sustainability (1.0, 2.0 and 3.0), moving from a business-as-usual model, that is to say, a business model totally focused on profit maximization and shareholder value creation externalizing natural and social costs. Business Sustainability 1.0 uses the three-dimensional concern in order to minimize costs and maximize benefits for shareholders; therefore, a Business Sustainability 1.0 company could indirectly create value also for other beneficiaries. However, a more important change takes place with Business Sustainability 2.0. This business model acknowledges the existence of three bottom lines and Business Sustainability 2.0 firms act in order to pursue not only economic profit but also social and environmental value creation through what has become popularly known as Corporate Social Responsibility. This is a way for companies to manage their risks and opportunities. Both 1.0 and 2.0 typologies adopt an inside-out perspective. This means relying on improvements and basing on what already exists following an efficiency approach. On the contrary, Business Sustainability 3.0 embraces an outside-in perspective, finding the sense of doing business in the business contribution to solve societal issues through its own skills. This innovative approach brings to a step forward also the value creation aspect. In fact, in the case the 3.0 typology, or true business sustainability, the common good becomes also an indispensable value creation beneficiary. As Roberts (2004 as cited in Fatemi and Fooladi, 2013) states, business exists to serve human needs. Only when the shift to outside-in perspective takes place can a business be considered truly sustainable.

The authors are self-critical about the feasibility of Business Sustainability 3.0, since this implies profit-driven companies focusing on sustainability and the common good. Nevertheless, they support the model as far as sustainability is embedded as the core of the business strategy through an outside-in approach. Moreover, they are doubtful about the ability of big companies to reach true business sustainability, and they see the issue of ownership as the biggest obstacle. In fact, stock-quoted corporations, having to do with the financial markets, are far more dependent on their shareholders and on their financial performance (Muff and Dyllick, 2014). According to the Economy for the Common Good movement, business revenues should be used for investment in the company and providing owners and employees with an income, whereas they should not pay interest to external investors, so that the company can aim at the common good without pressure for income maximization (Felber, 2012). Nevertheless, the impossibility for big corporations to be truly sustainable has not been demonstrated and the authors are currently searching for examples of businesses that could match the model, as well as for strategies to engage businesses for a further shift. Lastly, the authors acknowledge that starting a new business under the Business Sustainability 3.0 model (i.e., benefit corporations, social entrepreneurship, common good companies, etc.) is easier than shifting from Business Sustainability 2.0 to Business Sustainability 3.0. However, they also think that a shift of big corporations to true business sustainability is indispensable for planetary and societal sustainability (Muff and Dyllick, 2014).

In order actually to solve societal issues, sustainability should be at the center rather than business itself (Gray and Bebbington, 2000). However, according to a research run by Gray and Bebbington (2000) on transnational companies and sustainability, an important part of these corporations does not, cannot, or will not support sustainability if it endangers their financial return and, ultimately, their existence. In fact, a change in business sustainability model from business-centered to sustainability-centered could actually challenge the core of current business. Muff and Dyllick (2014) envisage new business models supported by a different idea of business and by suitable legal frameworks. However, Dyllick and Muff (2015) underline that the economic model and consumer behavior require changes as well in order for true business sustainability to work out.

References

About the UN Global Compact, n.d. UN Global Compact. Retrieved from, https://www.unglobalcompact.org/about on 25 June 2015.

Azapagic, A., Perdan, S., 2000. Indicators of sustainable development for industry: a general framework. Process Saf. Environ. Prot. 78 (4), 243–261.

Brundtland G, Khalid M, Agnelli S, Al-Athel SA, Chidzero B, Fadika LM, et al. (1987). Our Common Future. World Commission on Environment and Development. Retrieved from https://sustainabledevelopment.un.org/content/documents/5987our-common-future.pdf on 06/10/2019.

Corporate Social Responsibility, 2007. Investopedia. Retrieved from, http://www.investopedia.com/terms/c/corp-social-responsibility.asp. (Accessed 10 June 2019).

Crane, A., Palazzo, G., Spence, L.J., Matten, D., 2014. Contesting the value of "creating shared value". Calif. Manag. Rev. 56 (2), 130–153.

Daly, H.E., Cobb, J.B., 1989. For the Common Good: Redirecting the Economy Toward Community: The Environment and a Sustainable Future. Beacon Press, Boston, MA.

Dietz, E., Neumayer, E., 2007. Weak and strong sustainability in the SEEA: concepts and measurement. Ecol. Econ. 61 (4), 617–626.

Dyllick, T., Hockerts, K., 2002. Beyond the business case for corporate sustainability. Bus. Strateg. Environ. 11, 130–141.

Dyllick, T., Muff, K., 2015. Clarifying the meaning of sustainable business, introducing a typology from business-as-usual to true business sustainability. Organ. Environ. 1–19. Retrieved from, https://www.bsl-lausanne.ch/wp-content/uploads/2015/04/Dyllick-Muff-Clarifying-Publ-Online.full_.pdf. (Accessed 10 June 2019).

Ekins, P., Simon, S., Deutsch, L., Folke, C., De Groot, R., 2003. A framework for the practical application of the concepts of critical natural capital and strong sustainability. Ecol. Econ. 44 (2), 165–185.

Elkington, J., 2004. Enter the triple bottom line. In: The Triple Bottom Line: Does It All Add Up, pp. 1–16.

Fatemi, A.M., Fooladi, I.J., 2013. Sustainable finance: a new paradigm. Glob. Financ. J. 24 (2), 101–113.

Felber, C., 2012. L'Economia del Bene Comune. Un modello economico che ha futuro. Tecniche Nuove.

Forum for the Future, n.d. The five capital model: a framework for sustainability. Retrieved from https://www.forumforthefuture.org/project/five-capitals/overview on 15 Apr 2015.

Giddings, B., Hopwood, B., O'brien, G., 2002. Environment, economy and society: Fitting them together into sustainable development. Sustain. Dev. 10 (4), 187–196.

Goodland, R., 2002. Sustainability: human, social, economic and environmental. In: Encyclopedia of Global Environmental Change.vol. 5, pp. 481–491.

Goodwin, N.R., 2003. Five kinds of capital: useful concepts for sustainable development. Working paper, Tufts University. Retrieved from, http://www.ase.tufts.edu/gdae/publications/working_papers/03-07sustainabledevelopment.pdf. (Accessed 10 June 2019).

Gray, R., 2010. Is accounting for sustainability actually accounting for sustainability…and how would we know? An exploration of narratives of organisations and the planet. Acc. Organ. Soc. 35 (1), 47–62.

Gray, R., Bebbington, J., 2000. Environmental accounting, managerialism and sustainability: is the planet safe in the hands of business and accounting? Discussion paper. Retrieved from, https://cgi.st-andrews.ac.uk/media/csear/discussion-papers/CSEAR_dps-sustain-envaccman.pdf. (Accessed 10 June 2019).

Gray, R., Milne, M., 2002. Sustainability reporting: who's kidding whom? Chart. Accountants J. N. Z. 81 (6), 66–70.

Hallsmith, G., Lietaer, B., 2011. Creating Wealth: Growing Local Economies with Local Currencies. New Society Publishers.

International Integrated Reporting Council, 2013. International Integrated Reporting Framework.

IPCC, 2015. In:Pachauri, R.K., Meyer, L.A. (Eds.), Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. IPCC, Geneva, Switzerland Core Writing Team. 151 pp. Retrieved from, https://www.ipcc.ch/site/assets/uploads/2018/02/SYR_AR5_FINAL_full.pdf. (Accessed 10 June 2019).

IPCC, 2018. Special Report: Global Warming of 1.5°C. Summary for Policy-makers. Retrieved from, https://www.ipcc.ch/site/assets/uploads/sites/2/2019/05/SR15_SPM_version_report_LR.pdf. (Accessed 10 June 2019).

Ki-Moon, B., 2014. The road to dignity by 2030: ending poverty, transforming all lives and protecting the planet. Synthesis report of the Secretary-General on the post-2015 sustainable development agenda. Retrieved from, https://www.un.org/disabilities/documents/reports/SG_Synthesis_Report_Road_to_Dignty_by_2030.pdf.

Kolk, A., 2003. Trends in sustainability reporting by the fortune global 250. Bus. Strateg. Environ. 12 (5), 279–291.

KPMG International, 2012. Expect the Unexpected: Building Business Value in a Changing World. KPMG International, The Netherlands. Retrieved from, https://home.kpmg/content/dam/kpmg/pdf/2012/08/building-business-value-part-1.pdf. (Accessed 10 June 2019).

KPMG International, 2014. A New Vision of Value. Connecting Corporate and Societal Value Creation. KPMG International, The Netherlands. Retrieved from, https://assets.kpmg/content/dam/kpmg/pdf/2014/10/a-new-vision-of-value-v1.pdf. (Accessed 10 June 2019).

Labuschagne, C., Brent, A.C., Van Erck, R.P., 2005. Assessing the sustainability performances of industries. J. Clean. Prod. 13 (4), 373–385.

McDonough, W., Braungart, M., 1998. The next industrial revolution. Atl. Mon. 282(4).

McDonough, W., Braungart, M., 2002. Design for the triple top line: new tools for sustainable commerce. Corp. Environ. Strateg. 9 (3), 251–258.

Mebratu, D., 1998. Sustainability and sustainable development: historical and conceptual review. Environ. Impact Assess. Rev. 18 (6), 493–520.

Moneva, J.M., Archel, P., Correa, C., 2006. GRI and the camouflaging of corporate unsustainability. Account. Forum 30 (2), 121–137.

Muff, K., Dyllick, T.L., 2014. An Organizational Roadmap of Business Sustainability. Retrieved from, http://papers.ssrn.com/sol3/papers.cfm?abstract_id=2442211. (Accessed 5 August 2015).

Niţă, C.G., Ştefea, P., 2014. Cost control for business sustainability. Procedia Soc. Behav. Sci. 124, 307–311.

Ostrom, E., Ahn, T.K., 2003. Foundations of Social Capital. Edward Elgar Publishing Ltd.

Oxfam International, 2014. Even it up. Time to end Extreme Inequality. Retrieved from, https://www.oxfam.org/sites/www.oxfam.org/files/file_attachments/cr-even-it-up-extreme-inequality-291014-en.pdf. (Accessed 10 June 2019).

Pearce, D.W., Atkinson, G., 1998. The concept of sustainable development: an evaluation of its usefulness ten years after Brundtland. Swiss J. Econ. Stat. 134 (3), 251–269.

Pearce, D.W., Atkinson, G., 1993. Capital theory and the measurement of sustainable development: an indicator of "weak" sustainability. Ecol. Econ. 8 (2), 103–108.

Pearce, D., Markandya, A., Barbier, E., 1989. Blueprint for a Green Economy. Earthscan, London.

Pearce, D., Turner, R.K., 1990. Economics of Natural Resources and the Environment. Johns Hopkins University Press, Baltimore, MD.

Pfitzer, M., Bockstette, V., Stamp, M., 2013. Innovating for shared value. Harv. Bus. Rev. 91 (9), 100–107.

Pike, A., Rodriguez-Pose, A., Tomaney, J., 2007. What kind of local and regional development and for whom? Reg. Stud. 41 (9), 1253–1269.

Pless, N.M., Maak, T., Waldman, D.A., 2012. Different approaches toward doing the right thing: mapping the responsibility orientations of leaders. Acad. Manag. Perspect. 26 (4), 51–65.

Polman, P., 2015. Redefining business purpose. Driving societal and systems transformation. Buil. Sustain. Leg. 5, 9–16.

Pope Francis, 2015. Encyclical Letter Laudato si' of the Holy Father Francis on Care for our Common Home. Retrieved from, http://w2.vatican.va/content/francesco/it/encyclicals/documents/papa-francesco_20150524_enciclica-laudato-si.html. (Accessed 10 June 2019).

Porter, M., Kramer, M., 2011. Creating shared value. How to reinvent capitalism—and unleash a wave of innovation and growth. Harv. Bus. Rev. (Jan-Feb), 62–77.
Porter, M., Kramer, K., 2014. A response to Andrew Crane et al.'s article. In: Crane, A., Palazzo, G., Spence, L. J., & Matten, D. (2014). Contesting the value of "creating shared value". Calif. Manag. Rev. 56 (2), 130–153.
Puma, 2011. PUMA's Environmental Profit and Loss Account for the year ended 31 December 2010.
Raworth, K., 2017. Doughnut Economics: Seven Ways to Think Like a 21st-Century Economist. Chelsea Green Publishing.
Rockstrom, J., Steffen, W., Noone, K., Persson, A., Chapin, F.S., Lambin, E.F., Foley, J.A., 2009. A safe operating space for humanity. Nature 461 (7263), 472–475.
Sen, A., 1999. Development as Freedom. Oxford University Press, Oxford.
Schmidheiny, S., Stigson, B., 2000. Eco-Efficiency: Creating More Value with Less Impact. World Business Council for Sustainable Development.
Townsend, M., 2015. Editorial. Sustainable Economy Special Edition. Buil. Sustain. Leg. 5, 5–8.
True Price Foundation, 2012. True Price. The Concept Explained. True Price Foundation, The Netherlands.
UNEP-FI, 2011. Universal Ownership. Why Environmental Externalities Matter to Institutional Investors. UNEP Finance Initiative/PRI Association.
UNGC, 2013. Architects of a better world: Building the Post-2015 business engagement architecture. UN Global Compact Rep. 5 (1), 29–49.
United Nations, 1992a. Rio Declaration on Environment and Development 1992. Retrieved from, http://www.jus.uio.no/lm/environmental.development.rio.declaration.1992/portrait.a4.pdf. (Accessed 10 June 2019).
United Nations, 1992b. Agenda 21. Retrieved from, https://sustainabledevelopment.un.org/content/documents/Agenda21.pdf. (Accessed 10 June 2019).
United Nations, 1992c. United Nations Framework Convention on Climate Change. Retrieved from, https://unfccc.int/resource/docs/convkp/conveng.pdf. (Accessed 10 June 2019).
United Nations, 1998. Kyoto Protocol to the United Nations Framework Convention on Climate Change. Retrieved from, https://unfccc.int/resource/docs/convkp/kpeng.pdf. (Accessed 10 June 2019).
United Nations, 2003. Handbook of National Accounting: Integrated Environmental and Economic Accounting. United Nations Publications.
United Nations, 2012. The Future We Want. Outcome Document of the United Nations Conference on Sustainable Development. Rio de Janeiro, Brazil, 20–22 June 2012. Retrieved from, https://sustainabledevelopment.un.org/content/documents/733FutureWeWant.pdf. (Accessed 10 June 2019).
UN, Eurostat, IMF, OECD, World Bank, 2003. Integrated Environmental and Economic Accounting 2003—Handbook of National Accounting. Final Draft Circulated for Information Prior to Official Editing.
UN News Centre, 2012. Despite Agreed Environmental Goals, World still on Unsustainable Path. Retrieved from, http://www.un.org/apps/news/story.asp?NewsID=42166#.VXwutvntmko. (Accessed 10 June 2019).
Victor, P.A., 1991. Indicators of sustainable development: Some lessons from capital theory. Ecol. Econ. 4 (3), 191–213.
"WBCSD", n.d.. World Business Council for Sustainable Development. Retrieved from http://www.wbcsd.org/about.aspx on 13 June 2105.
WBCSD, 2002. The business case for sustainable development: making a difference toward the Johannesburg Summit 2002 and beyond. Corp. Environ. Strateg. 9 (3), 226–235.
WBCSD, 2010. Vision 2050. The New Agenda for Business. WBCSD, Geneva.
"What we do" (n.d.). Trucost. Retrieved at http://www.trucost.com/what-we-do on 06/16/2015.
Young, W., Tilley, F., 2006. Can businesses move beyond efficiency? The shift toward effectiveness and equity in the corporate sustainability debate. Bus. Strateg. Environ. 15 (6), 402–415.
Zimmermann, A., Gomez, P., Prosbst, G., Raisch, S., 2014. Creating societal benefits and corporate profit. MIT Sloan Manag. Rev. 3, 18–23.

Further reading

"Our ten guiding principles", n.d.. Economy for the Common Good. Retrieved from, https://www.ecogood.org/en/general-information/ecg-idea/our-ten-guiding-principles on 19 June 2015.

CHAPTER

3

Life cycle thinking tools: Life cycle assessment, life cycle costing and social life cycle assessment

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3.1 Life cycle assessment methodology

Life Cycle Assessment (LCA) methodology was born in order to face the need for methods for understanding and addressing environmental protection and the impacts of products. In other words, it was born to provide information to show the effects of an activity on the environment and to identify opportunities for making changes to reduce the environmental impacts (Perriman, 1993). Currently, the LCA methodology is standardized by two international standards, namely ISO 14040:2006 and ISO 14044:2006.

According to a well know definition given by SETAC, LCA is a methodology to evaluate the environmental burdens associated with a product, process, or activity. It identifies and quantifies energy and materials used and waste released to the environment; it assesses the impact of energy, materials, and releases to the environment; it identifies and evaluates opportunities for environmental improvements. LCA embraces the entire life cycle of a product, process, or activity, encompassing extraction and processing of raw materials; manufacturing, transportation, and distribution; use, reuse, maintenance; recycling, and final disposal (SETAC, 1993). Thus, it aims at assessing the environmental burdens through the identification and quantification of energy and materials consumed, waste produced, and possible environmental improvements at various points in the life cycle of products, processes, and activities.

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The currently accepted definition of LCA is "compilation and evaluation of inputs, outputs, and potential environmental impacts of a product system throughout its life cycle," which typically occurs in four steps (ISO, 2006a,b), as shown in Fig. 3.1.

The first phase is the description of the goal and scope, which includes defining the objectives of the study and setting the system boundaries. The second phase, called inventory analysis, compiles inputs and outputs for each process in the life cycle and sums them across the whole system. Typically, several hundreds of emissions and resources are quantified. In the third phase, known as life-cycle impact assessment (LCIA), emissions and resources are grouped according to their impact categories and converted to common impact units to make them comparable. The final phase is the interpretation of the inventory and impact assessment results in order to answer the objectives of the study (Hellweg and Milà i Canals, 2014). A scheme representing the application of the LCA methodology is shown in Fig. 3.2.

Thanks to its characteristics, LCA is also used to inform decision-makers in industry, government, or nongovernment organizations; to select indicators of environmental performance; and to implement eco-labeling and make environmental claims. As a decision support tool, LCA is generally applied to a product, but also to a system or service (Tillman and Baumann, 2004). For instance, Liamsanguan and Gheewala (2008) used LCA as a decision support tool for environmental assessment of municipal solid waste management systems. Ramasamy et al. (2015) used LCA as a tool to support decision making in the biopharmaceutical industry, revealing considerations and challenges; whereas Means and Guggemons (2015) developed a framework for environmental decision-making based on LCA for commercial buildings.

Dong et al. (2018) analyzed the need and obstacles for integrating LCA into decision analysis, whereas Zanghelini et al. (2018) studied how multicriteria decision analysis is aiding LCA in results interpretation. LCA is also used as a support for decision making in the public sector; for instance, Guérin-Schneider et al. (2018) focused on how better to include environmental assessment in public decision-making in the case of wastewater treatment, and Jouini et al. (2019) developed a framework for coupling a participatory approach with LCA for public decision-making in rural territory management.

A detailed literature review on sustainable evaluation for energy systems carried out by Campos-Guzmán et al. (2019) revealed that LCA and multicriteria decision-making techniques, when used in combination within the same methodological framework, can be an effective tool for sustainable evaluation. In particular, the combination of LCA and analytic hierarchic process is often used for its simplicity and robustness for sustainable evaluation in energy systems.

3.1.1 Goal and scope definition

The goal definition comprises the identification of the intended application, the reasons for carrying out the study, the stakeholders involved, and how the results are intended to be used; i.e., if they are intended to be used in comparative assertion or if they are intended to be disclosed to the general public. The scope instead defines the dimension and detail of the study to reach the goal. In the scope, the following items have to be defined (ISO, 2006a):

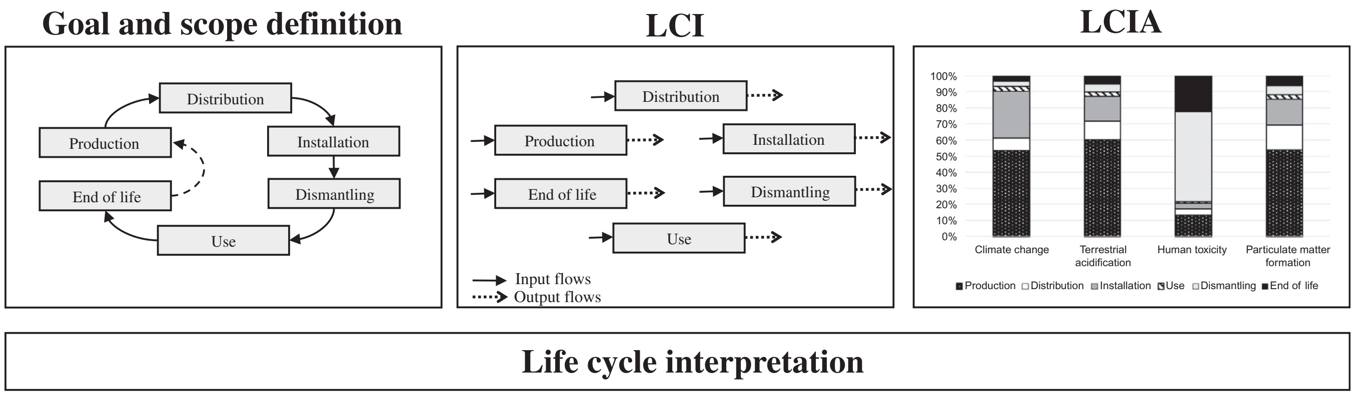

FIG. 3.1 Phases of the LCA methodology.

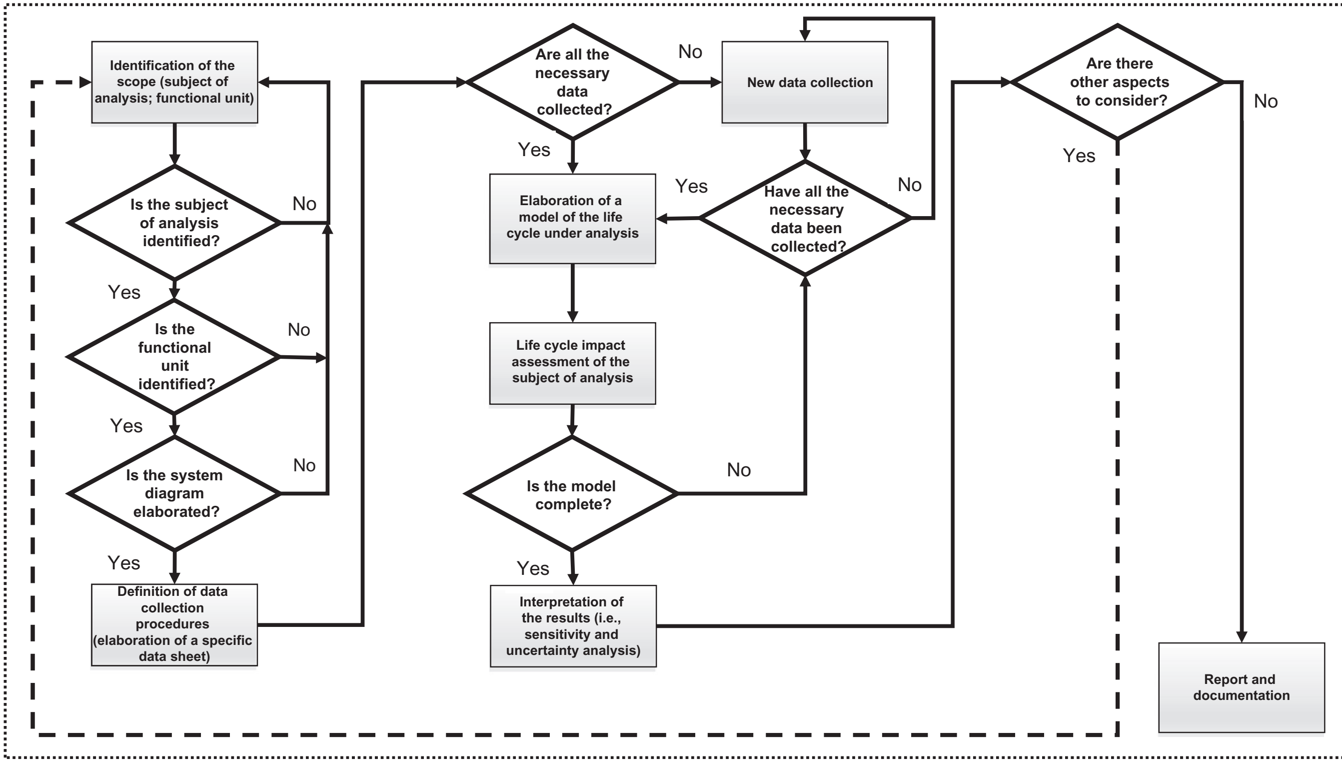

FIG. 3.2 Scheme representing the application of the LCA methodology.

– product system;
– functions of the product system;
– functional unit;
– system boundary;
– allocation procedures;
– impact category and methodologies of impact assessment;
– data requirements;
– assumptions;
– limitations;
– initial data quality requirements;
– type of critical review; and
– type and format of the report.

The main item is the functional unit, which allows quantifying of the identified functions of the system or product, is the reference unit for the calculation of the inputs and outputs, and ensures comparability in case of comparative studies. An LCA study has to be conducted by defining product system and the system boundaries, which are necessary to establish the functions to be considered. Ideally, inputs and outputs should be elementary flows. However, the choice of the elements of the system depends on the goal and scope of the study, the intended application, the audience, the assumptions made, data, cost and cut-off criteria (ISO, 2006a), namely the criteria to establish the threshold under which it is possible to exclude not significant environmental burdens.

3.1.2 Life cycle inventory (LCI)

This stage covers data collection and calculations to quantify the relevant inputs and outputs of the system. It is an iterative step and thus, further data requirements or limitations may be identified to meet the goal of the study during the conduction of the analysis. The main data required to conduct an LCA study are (ISO, 2006b):

– consumption of inputs;
– products, co-products and waste flows;
– emission to air, water, and soil; and
– other environmental aspects.

Input and output data have to be organized as usage of raw material, water usage, energy consumption, emission into water, air, and soil, and waste. In addition, the following items have to be indicated (ISO, 2006a):

– data sources;
– reference process
– reference technology
– geographical area;
– monitoring details;
– measuring methods; and
– specific units of measurement.

After data collection, a calculation procedure to validate the collected data has to be implemented; data have to be connected to the unit process and to the reference flow of the functional unit. These actions are necessary to generate the results of the inventory phase.

In this phase, a required item concerns allocation procedures. The main problem is which flows and environmental interventions must be allocated to the functional unit, and which should be allocated to other product systems. Within LCA studies, two different cases have to be distinguished for the application of allocation procedures (Toniolo et al., 2017a). The first case occurs when simultaneous products are manufactured and thus, different inputs and outputs shall be allocated to different products, whereas the second case occurs when subsequent products are realized in recycling or reuse systems. In general, almost all of the industrial processes produce more than one product or recycle a portion of the waste material (Frischknecht et al., 2005, 2007; Frischknecht, 2010).

However, even if in general allocation procedures represent a critical point (Ardente and Cellura, 2012), this distinction is not deeply investigated in ISO 14040 and ISO 14044. Anyway, it is possible to appeal to ISO TR 14049 where some examples are described and some considerations are added. Other considerations can be found in the ILCD (International reference Life Cycle Data system) handbook. If the market value of the waste or end-of-life product at its point of origin is above zero, in LCA perspective, it is a co-product and the multifunctionality has to be solved by allocation. However, the case of recycling is insofar different from the general case of multifunctionality, as the secondary good is not only a co-function of the system, but is itself recycled again and again (while each time at lower amounts and/or quality, considering losses of each loop) (EC-JRC, 2010).

3.1.3 Life cycle impact assessment

In this phase, the effects of the substances on the selected impact categories and the processes that generated them are analyzed (Toniolo et al., 2017b). Inventory data are associated with environmental impact categories and category indicators. The elements within this phase are (ISO, 2006b):

- Classification. Classification assesses which global/local impact the input/output is contributing to. There are input-relating categories and output-related categories. There are several categories that are commonly used, such as climate change, ozone layer depletion, eutrophication, acidification, particulate matter formation, and several impact categories under development, such as acoustic impact.
- Characterization. Impacts are quantified within given categories with the general Eq. (3.1) (Goedkoop et al., 2013):

$$EP(j)i = Q \times EQ(j)I \tag{3.1}$$

where $EP(j)i$ is the environmental impact of substance i with reference to the impact category j, Q is the quantity of substance I, and $EQ(j)i$ is a factor representing the substance i contribution to the impact j. Different substances contributing to an environmental impact are aggregated considering their substance-specific effect. Scientific models are used, therefore characterization could be considered objective. Fig. 3.3 shows an example of characterized results of an LCA study.

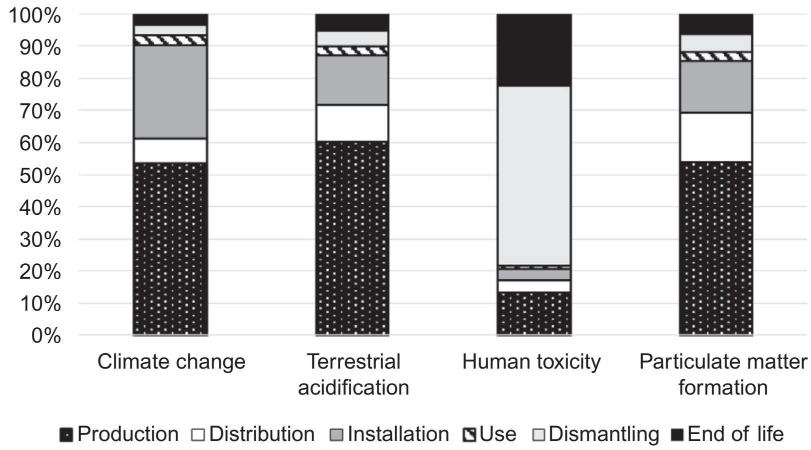

FIG. 3.3 Example of LCIA results.

– Normalization. The aim of normalization is to clarify the relative importance of the indicator results. Values are divided with reference to a standard value.
– Weighting. The categories results are assigned numerical factors in accordance with their importance, then multiplied by these factors and finally aggregated in a single impact score.

Classification and characterization are mandatory elements, whereas normalization and weighting are discretionary (ISO, 2006a).

3.1.4 Life cycle interpretation

In this phase, the results of the LCI and LCIA are considered together with reference to the objective of the study. The interpretation is comprised of several elements (ISO, 2006a,b):

– Identification of the significant issues based on the results of the LCI and LCIA phases. The objective of this step is to analyze the results from the LCI or LCIA phases in order to help determining the significant issues, in accordance with the goal and scope definition.
– An evaluation that considers completeness, sensitivity, and consistency checks. During this evaluation, the following techniques should be used: completeness check, sensitivity check, consistency check. The results of uncertainty analysis and data quality analysis should supplement these checks. The completeness check is performed to control that all the needed data and information are available and complete; the sensitivity check is performed to evaluate the reliability of the results; and the consistency check is conducted to determine whether assumptions, methods, and data are coherent with the goal and scope defined.
– Conclusions, limitations and recommendations.

3.2 Life cycle costing methodology

The use of life cycle costing (LCC) was reported for the first time in a tractor delivery contract in the 1930s in the United States (Ciroth et al., 2011); it was also used in the US Department of Defense in the mid-1960s for the acquisition of high-cost military equipment (Gluch and Baumann, 2004; Hoogmartens et al., 2014). Some attempts were made in the mid-1980s to adapt LCC to building investments and several research projects have been developed to adapt the LCC methodology for the construction industry and for sustainable public procurement, placing LCC in an environmental context (Gluch and Baumann, 2004). Therefore, we can say that LCC is not a new concept (Heijungs et al., 2013).

The LCC technique is often used to examine the preferable alternative of products and services from an economic point of view (Dragos and Neamtu, 2013), to ensure the ranking of different investment alternatives, and the adoption of the best solution, moving beyond the purchase price of a good or a service, and using a long-term approach for the decision-making process (Woodward, 1997). It has also become an important economic tool for decision-making, as it is used to evaluate the costs associated with an item in its whole life cycle, from its design through its production, transport to its end of life, and it is often applied in combination with LCA (Di Maria et al., 2018; Buyle et al., 2019).

For instance, Choi (2019) applied LCC and LCA in the case of maintenance and rehabilitation of highway pavement; whereas Xue et al. (2019) applied them for urban water system. Several combined applications exist for the building sector; i.e., Auer et al. (2017) conducted a case study on the performance of a modernized manufacturing system for glass containers. Schmidt and Crawford (2017) developed a framework for the integrated optimization of the life cycle greenhouse gas emissions and cost of building for buildings. Balasbaneh et al. (2018) analyzed the choice of different hybrid timber structures for low medium cost single-story residential buildings. Mah et al. (2018) studied the application of LCA and LCC for the management of concrete waste generated during the construction and demolition stages, and Hong et al. (2019) for building design.

In addition, several authors combined LCC with LCA and multicriteria decision analysis methods, among which Miah et al. (2017) proposed a novel hybridized framework combining integrated methods for LCA and LCC to provide decision-makers a comprehensive method to investigate environmental and economic aspects. They used a hybrid method combining the technique for order of preference by similarity to ideal solution (TOPSIS) and analytical hierarchy process (AHP). Harkouss et al. (2018) applied a multiobjective optimization methodology for net zero energy buildings using multicriteria decision making and LCC; whereas Invidiata et al. (2018) proposed a method that combines adaptive thermal comfort, climate change, LCA, LCC, and multicriteria decision making to identify the best design strategies for improving buildings.

Kouloumpis and Azapagic (2018) presented a new model, which integrates LCA, LCC, and Social LCA into a fuzzy inference framework; while Rocchi et al. (2018) conducted a sustainability evaluation of retrofitting solutions for rural buildings through LCA and multicriteria analysis.

The main difference between other traditional investment calculus methods and LCC is that LCC has an expanded life cycle perspective. The life cycle cost of an item is the sum of all costs expended from its conception and production, through its operation, to the end of its useful life (Woodward, 1997). LCC helps shifting from the best value for money to the best value across the asset life cycle (Perera et al., 2009), and includes a comparison between options or an estimation of future costs at portfolio, projects, or components over a defined period of analysis (ISO, 2011). It allows evaluation of the cost of acquisition, development, operation, management, repair, disposal, and decommissioning (Langdon, 2006; Reidy et al., 2015).

Therefore, LCC can be used both by private and public organizations to optimize the cost of acquiring, owning, and operating physical assets over their useful lives, trying to evaluating all the significant costs involved in the life cycle (Woodward, 1997). According to Woodward (1997), the costs of an item can be comprised of engineering and development costs, production and implementation costs, operating costs, and end of life costs. For instance, production and implementation costs comprise the initial capital costs, namely purchase costs, which include assessment of goods like land and buildings; they can be obtained through quotations from suppliers. There are also acquisition/finance costs, which include the cost effect of alternative sources of funds and regulations and installation/commissioning/training costs, which include the installation of machines and the training of the workers. An important concept is the life of the asset, which defines its life expectancy and decisive factors considering functional life, physical life, technological life, economic life, and social and legal life (Woodward, 1997). Associated with the concept of the life of an asset, there is the concept of the discount rate. The selection of the discount rate is a fundamental phase in LCC application. A high discount rate will tend to facilitate options with low capital cost, short life, and high recurring cost; whereas a low discount rate will have the opposite effect.

A way to define the discount rate in LCC studies was proposed by Islam et al. (2015). They calculated future costs, for instance for operation, maintenance, and demolition, using Eq. (3.2); then they discounted them using Eq. (3.3). Because of future risk, the discount rate exceeds the inflation rate.

$$FC = PC \cdot (1 + f)n \tag{3.2}$$

where FC = future cost, PC = present cost, f = inflation rate, and n = number of years.

$$DPV = (1 + d) \tag{3.3}$$

where DPV = discounted present value, FC = future cost, d = discount rate, and n = number of years.

In the scientific literature, three possible types of LCC emerge, namely conventional LCC, environmental LCC, and societal LCC (Hunkeler et al., 2008). The conventional LCC is the assessment of all the costs associated with the life cycle of a product. The focus of the evaluation is on real, internal costs and sometimes the costs of the end of life are not included. The environmental LCC is the evaluation of all the costs associated with the life cycle of a product covered by the actors in the product life cycle, for instance suppliers, manufacturers, users or consumers, and end of life actors. However, the environmental problems are simplified, since it assumes that everything can be expressed as a one-dimensional unit, such as monetary flows (Gluch and Baumann, 2004). The societal LCC includes all the costs that are associated with the entire life cycle of a product. These costs are covered by anyone in the society, today, or in the long-term future (Hunkeler et al., 2008).

Contrary to LCA methodology, which is standardized by two ISO standards, LCC is not structured by a specific international standard. The standard ISO 15686-5:2008 provides the instructions and the guidelines for the application of this methodology in the building sector, thus it cannot be applied to other contexts. However, some authors propose a methodology comprised of 10 steps to conduct an LCC study. All the 10 phases are required; they can be implemented in sequence, but also out of sequence, or sometimes simultaneously (Dhillon, 2010). The 10 steps are as follows (Dhillon, 2010):

1. Determine the purpose of the LCC analysis.
2. Define and scope the system/support system.
3. Select the appropriate estimating methodology/LCC model.
4. Gather data and make the appropriate inputs to the methodology/model.
5. Perform sanity checks of input and outputs.
6. Perform sensitivity analysis and risk assessment.
7. Formulate the results of the LCC analysis.
8. Document the LCC analysis.
9. Present the LCC analysis.
10. Update the LCC analysis/baseline.

The steps proposed by Greene and Shaw (1990) can be grouped in four phases, in line with LCA methodology, as reported in Fig. 3.4.

3.2.1 Goal and scope definition

The identification of the purpose for conducting an LCC study is the first necessary step. In some cases, the purpose may be obvious or predetermined, as in a source selection LCC analysis. Nevertheless, in other cases, when the purpose is not sufficiently clear, considerable efforts can be made before understanding the direction of the study. In this phase, the required issues to conduct the study are defined, including the criteria to be used for selection of alternatives. The goal of an LCC analysis may be a comparative analysis of a new system versus an existing system, or provisioning purposes (Greene and Shaw, 1990). The scope definition includes the system units to be included in the study, the definition of the subject of the study, the definition of assumptions, and the identification of limitations.

Usually, the system and subsystems are not completely defined until the final design, and the scope need to be revised. In the beginning of the study, the system definition and the scope may be vague. If the system under analysis is replacing or is similar to an existing system, it is important to include similarities and differences. This step is fundamental to ensuring a credible LCC analysis (Greene and Shaw, 1990). The selection of an appropriate LCC model depends on several factors, such as the type of system/support system/subsystem to be analyzed, the system units included in the life cycle and the type of analysis to be conducted, as defined in the first step.

The amount of data available to conduct an analysis is determined by the phase of development of the product or process under analysis. Only limited data may be available during the research and development phase, and so for instance parametric cost estimating models can be appropriate. If the product is under production, or a process is operative, there may be

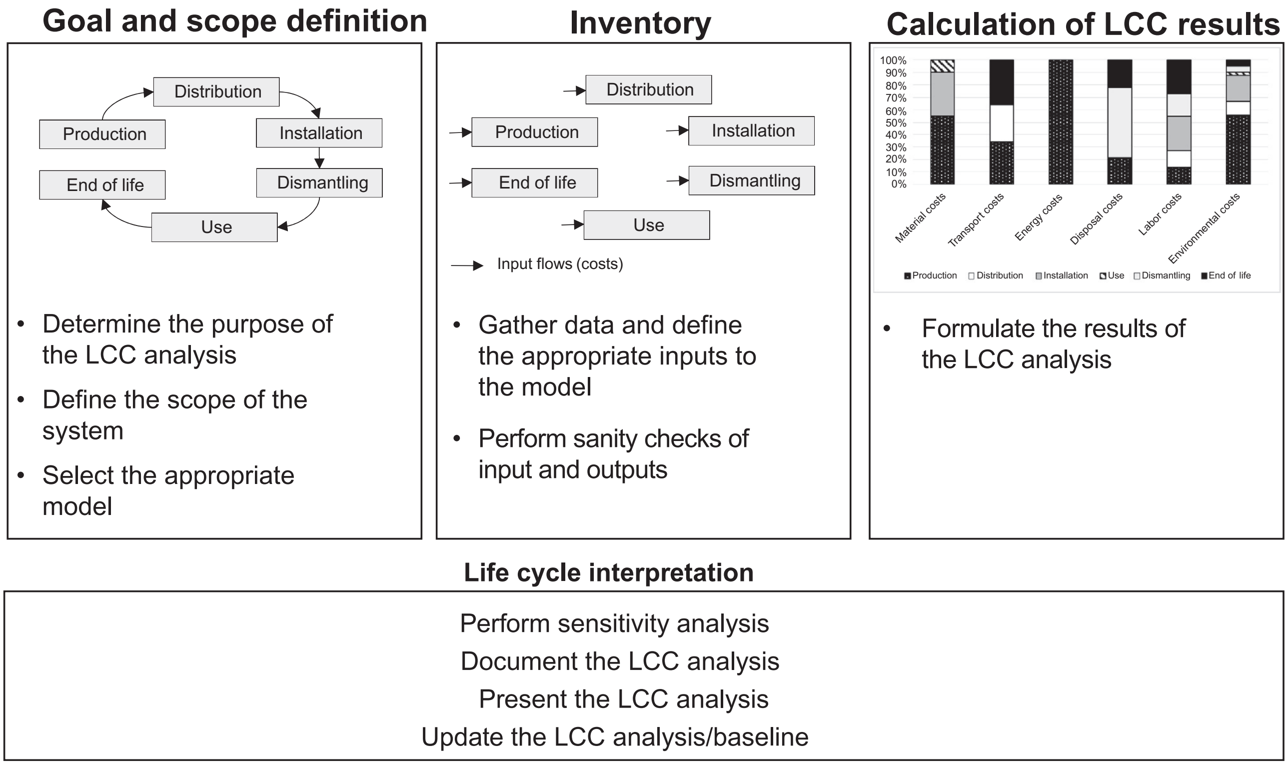

FIG. 3.4 The steps proposed by Greene and Shaw (1990) to perform an LCC study grouped in four phases.

enough information for an engineering bottoms-up estimation model. The model could refer to the number of operating hours a system accumulates. Different models are appropriate for each situation, for instance it is possible to use an operating hour driven model, or a periodic inspection, or a periodic exercise driven model. For a system or subsystem trade study, it can be appropriate to use a bottom-up rather than a top-down model. This means that a perfect LCC methodology/model that fits all applications does not exist. In any case, the selected model has to be documented properly, has to be verified and validated, and has to contain all the elements that need to be covered (Greene and Shaw, 1990).

3.2.2 Inventory

Data collection is a significant, time-consuming phase. Data can be collected from different sources, it is possible to collect data in the plant where the product is realized or the process occurs, directly. However, sometimes, it is necessary to make use of data from technical reports, from the scientific literature, or it can be necessary to do some estimations. The input and output data have to be checked in terms of consistency, accuracy, validity, and completeness to make sure that erroneous information is not present in the analysis or that required information has not been neglected. This step is important in order to avoid misinformation coming out of the analysis and to improve the credibility of the analysis.

3.2.3 Calculation of LCC results

After the collection of all the needed data, the implementation of the LCC model, and the making of the sensitivity or risk assessment, the results of the LCC study can be calculated. If there is the need to revise some methodological choices, it is possible to revise previous phases, such as the purpose of the study, and adjust the study. In this step, it is necessary to evaluate possible alternatives and identify the factors that significantly affect the LCC study. During this step, the results of LCC analysis can be inflated or discounted.

3.2.4 Interpretation

The input parameters with significant risk and high potential for impacting cost need to be varied over a reasonable range. It is possible to use the "best case," the "worst case," or something in between. LCC evaluations are estimations regarding the cost range, which can be expressed in a statistical way, or through a model with a limited number of parameters. The results of the LCC study have to be documented, along with the information to support the analysis.

3.3 Social life cycle assessment methodology

Social life cycle assessment (S-LCA) is a novel methodology to address the social impacts of products and services along their life cycle. It is based on LCA methodology, with some adaptations, and was developed in accordance with the ISO 14040 and 14044 standards (Ekener Petersen, 2015). It has been applied in different sectors, such as food, biofuels, materials, technology, and services (Vasta et al., 2015).

According to the definition given by UNEP/SETAC (2009), S-LCA is an assessment technique to evaluate the social and socio-economic aspects of products and their potential impacts along their life cycle from extraction and processing of raw materials to final disposal, passing through manufacturing, distribution, use, reuse, maintenance, and recycling. This technique tries to assess the social impacts of a product or service where social impacts are mainly understood as the impacts on human capital, human well-being, cultural heritage, and social behavior (Sala et al., 2015). These impacts can be associated with the behaviors of enterprises or with their processes and can be positive or negative. They are consequences of social interactions raised during an activity, such as production, consumption, or disposal (UNEP/SETAC, 2009). The basic idea is that social impacts could be embodied in products and related to supply chains (Sala et al., 2015). S-LCA can be a profitable tool to give answers to the following questions: what is the social value of products? How to define it, and to quantify it? (Russo Garrido et al., 2018).

S-LCA can be applied on its own or in combination with LCA, using generic and site-specific data (UNEP/SETAC, 2009). However, the level of methodological development, application, and harmonization of S-LCA is still at a preliminary stage (Sala et al., 2015). Contrary to other social assessment techniques, it takes into consideration the entire life cycle of a product or a service and helps in evaluating the social impacts that directly affect stakeholders. The stakeholders considered are clustered in five categories based on shared interests, namely workers/employees, local community, society, consumers, and value chain actors. Each category of stakeholders is associated with specific subcategories, for instance workers/employees are linked to the subcategories "freedom of association," "child labor," "fair salary," "working hours," "forced labor," "discrimination," "health and safety," and "social benefits." Consumers are linked to the subcategories "health and safety," "feedback mechanism," "consumer privacy," "transparency," and "end of life responsibility" (UNEP/SETAC, 2009).

In general, there are two methodological approaches to conduct S-LCA, called "performance reference point" methods and "impact pathways" methods. Performance reference point methods take into consideration living and working conditions of workers at different life cycle phases; whereas impact pathways methods evaluate the social impacts using characterization models with indicators similar to LCA (Sala et al., 2015). Different S-LCA methodologies have been proposed in several case studies and discussions are still open in the research community regarding the role of local stakeholders and the need of a common social theory as base to develop S-LCA (Ekener Petersen, 2015). It is still under debate whether qualitative or quantitative assessment methods are more suitable for S-LCA; indeed, a certain level of subjectivity cannot be avoided (Sala et al., 2015) and social issues are influenced by the subjectivity of researchers and the social context (Soltanpour et al., 2018). Despite this, S-LCA should be used to support decision making by different actors, identifying how to reduce the social hotspots along the supply chain (Sala et al., 2015), and can support the organizations within decision-making processes by optimizing the efforts and resources in order to achieve social sustainability (D'Eusanio et al., 2018). Currently, S-LCA is used as a business-oriented methodology, where the social assessment is based on the behavior of the organizations that are involved in the processes under study (Arzoumanidis et al., 2018).

Kolotzek et al. (2018) developed a model combining LCA and S-LCA for the assessment of raw material supply risks and used the analytic hierarchy process to weight the indicators. Santos et al. (2017) performed an S-LCA of school buildings for higher education, focusing on the criteria of health and comfort. They used analytic hierarchic process to obtain the weighting scheme to rate social performance. Chandrakumar et al. (2017) elaborated a multicriteria decision support system based on an S-LCA framework for evaluating three sanitation system designs. They applied the analytic hierarchy process to solve their proposed model. Halog and Manik (2011) proposed a framework adopting LCA, LCC, S-LCA, and stakeholders analysis supported by multicriteria decision techniques for the assessment of the development of biofuel supply chain networks.

Currently, new guidelines are under development for the application of S-LCA, they will consider and incorporate methodological advancements and recent practical experiences. They will also deal with harmonization of S-LCA methods, specification of application of S-LCA for organizations, and scale up of scientific debate (Benoit Norris et al., 2018). The following phases are usually conducted to develop an S-LCA according to the current guidelines (UNEP/SETAC, 2009).

3.3.1 Definition of goal and scope

The first phase of an S-LCA study is the definition of the goal and scope of the study. A clear statement of purpose, namely the goal of the study and the intended use, is needed. Based on the goal, a critical review may be planned. It is important to take into consideration that the ultimate objective is improving of social conditions and of the socio-economic performance of a product throughout its life cycle for all of its stakeholders (UNEP/SETAC, 2009). Successively, the scope has to be defined; the function of the product under study, its utility, and the functional unit, defined in time and space need to be determined. To define the functional unity, the following properties need to be considered: functionality, technical quality, additional services, aesthetics, image, costs related to purchase, use, and disposal (UNEP/SETAC, 2009). The definition of the functional unit is a key issue; indeed, in some cases, it is difficult to conceptualize it (Sala et al., 2015), and even if it is required, it does not seem to be a common practice to define it (Arzoumanidis et al., 2018). In addition, the following actions need to be conducted (UNEP/SETAC, 2009):

– Determine the unit processes to be included in the assessment, namely the system boundaries.
– Organize data collection; identify which data will be collected, for instance generic or specific data.
– Specify impact categories and subcategories.
– Define the stakeholders involved and the type of critical review, if needed.
– Define the types of impact to be evaluated and the related indicators and methods.
– Define allocation procedures.
– Plan the interpretation and identify assumptions, limitations, analyze data quality.

3.3.2 Social life cycle inventory analysis

The life cycle inventory phase can be conducted performing the following actions: collecting data on unit processes and redefining the selected system boundaries if needed. Data to be collected may be primary or secondary data and data for characterization. Primary data sources can be audits of enterprise documentation and documentation of authorities, making use of participative methodologies, interviews, focus groups, questionnaires, and surveys (UNEP/SETAC, 2009; Arcese et al., 2013; Trevisani Juchen et al., 2018). Secondary data sources can be scientific literature, web search, and databases. Collected data and functional unit have to be related and aggregated when applicable (UNEP/SETAC, 2009). Collected data should meet a list of quality criteria, such as validity—data have to provide information on what is intended to be measured; relevance; completeness—data have to cover the needs of the study; and accessibility—data collection has to be well documented. Then, uncertainty analyses should be performed and the measurement methods to generate the data have to be analyzed in order to define if they are appropriate (UNEP/SETAC, 2009).

3.3.3 Social life cycle impact assessment

This is the third phase of an S-LCA. Its purpose is to aggregate inventory data within categories and subcategories and to make use of additional information to help in understanding the significance of the collected information (UNEP/SETAC, 2009). This phase can be conducted through some actions: selection of the impact categories, subcategories, and characterization models, classification, namely associating inventory data with categories and subcategories, and characterization, namely calculating the impacts for the subcategories indicators (UNEP/SETAC, 2009). Unlike in LCA, where impacts are mostly negative, social impacts can also be positive (Sala et al., 2015). Indicators for S-LCA can be quantitative or qualitative depending on the goal of the study (UNEP/SETAC, 2009). Contrary to LCA, where impacts are calculated through a multiplication between the inventory data and a characterization factor recognized by international scientific community, S-LCA can express the impacts through a scoring system, providing as estimation of the impact (UNEP/SETAC, 2009).

3.3.4 Social life cycle interpretation

During this phase, the significant issues are identified and consideration about completeness and consistency of the study are drawn. Finally, the level of engagement with stakeholders is evaluated; conclusions and recommendations are reported (UNEP/SETAC, 2009).

References

Arcese, G., Lucchetti, M.C., Merli, R., 2013. Social life cycle assessment as a management tool: methodology for application in tourism. Sustainability 5, 3275–3287.

Ardente, F., Cellura, M., 2012. Economic allocation in life cycle assessment. The state of the art and discussion of examples. J. Ind. Ecol. 16, 387–398.

Arzoumanidis, I., D'Eusanio, M., Raggi, A., Petti, L., 2018. Functional unit definition criteria in LCA and social LCA: a discussion. In: Social LCA People and Places for Partnership 6th SocSem Pre-Proceeding, Fruitop Thema, Pescara. pp. 121. ISBN: 978-2-9562141-1-3.

Auer, J., Bey, N., Schäfer, J.-M., 2017. Combined life cycle assessment and life cycle costing in the eco-care-matrix: a case study on the performance of a modernized manufacturing system for glass containers. J. Clean. Prod. 141, 99–109.

Balasbaneh, A.T., Bin Marsono, A.K., Khaleghi, S.J., 2018. Sustainability choice of different hybrid timber structure for low medium cost single-story residential building: environmental, economic and social assessment. J. Build. Eng. 20, 235–247.

Benoit Norris, C., Russo Garrido, S., Traverso, M., Ekener, E., Valdivia, S., Lehmann, A., Finkbeiner, M., 2018. A global effort: 2019 S-LCA guidelines. In: Social LCA People and Places for partnership 6th SocSem Pre-proceeding, Fruitop Thema, Pescara, pp. 134–139. ISBN: 978-7-2-9562141-1-3.

Buyle, M., Galle, W., Debacker, W., Audenaert, A., 2019. Sustainability assessment of circular building alternatives: consequential LCA and LCC for internal wall assemblies as a case study in a Belgian context. J. Clean. Prod. 218, 141–156.

Campos-Guzmán, V., García-Cáscalesa, S., Espinosa, N., Urbina, A., 2019. Life cycle analysis with multi-criteria decision making: a review of approaches for the sustainability evaluation of renewable energy technologies. Renew. Sust. Energ. Rev. 104, 343–366.

Chandrakumar, C., Kulatunga, A.K., Mathavan, S., 2017. A multi-criteria decision-making model to evaluate sustainable product designs based on the principles of design for sustainability and fuzzy analytic hierarchy process. In: Campana, G., Howlett, R., Setchi, R., Cimatti, B. (Eds.), Sustainable Design and Manufacturing 2017. SDM 2017. In: Smart Innovation, Systems and Technologies, vol. 68. Springer, Cham. ISBN: 978-3-319-57077-8.

Choi, J.-H., 2019. Strategy for reducing carbon dioxide emissions from maintenance and rehabilitation of highway pavement. J. Clean. Prod. 209, 88–100.

Ciroth, A., Hunkeler, D., Klöpffer, W., Swarr, T.E., Pesonen, H.-L., 2011. Life Cycle Costing – a Code of Practice. Key messages and critical evaluation. LCA XI Chicago, Chicago. Available at: http://www.greendelta.com/uploads/media/LCAXI_LCC.pdf.

D'Eusanio, M., Lehmann, A., Zamagni, A., Finkbeiner, M., Petti, L., 2018. How experiences and existing data of companies can be used to define the goal and scope in a social organisational life cycle assessment (SO-LCA). In: Social LCA People and Places for Partnership 6th SocSem Pre-proceeding, Fruitop Thema, Pescara. ISBN: 978-2-9562141-1-3, pp. 208–214.

Dhillon, B.S., 2010. Life Cycle Costing for Engineers. CRC Press, Boca Raton, FL.

Di Maria, A., Eyckmans, J., Van Acker, K., 2018. Downcycling versus recycling of construction and demolition waste: combining LCA and LCC to support sustainable policy making. Waste Manag. 75, 3–21.

Dong, Y., Miraglia, S., Manzo, S., Georgiadis, S., Danielsen Sørup, H.J., Boriani, E., Hald, T., Thöns, S., Hauschild, M.Z., 2018. Environmental sustainable decision making—the need and obstacles for integration of LCA into decision analysis. Environ. Sci. Pol. 87, 33–44.

Dragos, D., Neamtu, B., 2013. Sustainable public procurement: life cycle costing (LCC) in the new EU directive proposal. Eur. Procurement Public Private Partnership Law Rev. 8, 19–30

EC-JRC, European Commission—Joint Research Centre—Institute for Environment and Sustainability, 2010. International Reference Life Cycle Data System (ILCD) Handbook—General guide for Life Cycle Assessment—Detailed guidance, first ed. Publications Office of the European Union, Luxembourg. March 2010. EUR 24708 EN.

Ekener Petersen, E., 2015. State of the art on social LCA. In: Sala, S., Vasta, A., Mancini, L., Dewulf, J., Rosenbaum, E. (Eds.), Social Life Cycle Assessment—State of the Art and Challenges for Supporting Product Policies. Publications Office of the European Union, Luxemburg, pp. 27–31. European Commission, Joint Research Centre, Institute for Environment and Sustainability.

Frischknecht, R., 2010. LCI modelling approaches applied on recycling of materials in view of environmental sustainability, risk perception and eco-efficiency. Int. J. Life Cycle Assess. 15, 666–671.

Frischknecht, R., Jungbluth, N., Althaus, H.J., Doka, G., Dones, R., Heck, T., Hellweg, S., Hischer, R., Nemecek, T., Rebitzer, G., Spielmann, M., 2005. The ecoinvent database: overview and methodological framework. Int. J. Life Cycle Assess. 10, 3–9.

Frischknecht, R., Jungbluth, N., Althaus, H.J., Doka, G., Dones, R., Heck, T., Hellweg, S., Hischer, R., Nemecek, T., Rebitzer, G., Spielmann, M., Wernet, G., 2007. Overview and Methodology, Ecoinvent Report No. 1. Swiss Centre for Life Cycle Inventory, Dübendorf, Switzerland.

Gluch, P., Baumann, H., 2004. The life cycle costing (LCC) approach: a conceptual discussion of its usefulness for environmental decision-making. Build. Environ. 39 (5), 571–580.

Goedkoop, M., Heijungs, R., Huijbregts, M., De Schryver, A., Struijs, J., Van Zelm, R., 2013. ReCiPe 2008 Report I: Characterisation. Version 1.08, Ministerie van VROM, Den Haag, Nederland.

Greene, L.E., Shaw, B.L., 1990. The steps for successful life cycle cost analysis. IEEE Conf. Aerosp. Electron. 3, 1209–1216.

Guérin-Schneider, L., Tsanga Tabi, M., Roux, P., Catel, C., Biard, Y., 2018. How to better include environmental assessment in public decision-making: Lessons from the use of an LCA-calculator for wastewater systems. J. Clean. Prod. 187, 1057–1068.

Halog, A., Manik, Y., 2011. Advancing integrated systems modelling framework for life cycle sustainability assessment. Sustainability 3 (2), 469–499.

Harkouss, F., Fardoun, F., Biwole, P.H., 2018. Passive design optimization of low energy buildings in different climates. Energy 165, 591–613.

Heijungs, R., Settanni, E., Guinee, J., 2013. Toward a computational structure for life cycle sustainability analysis: uniffying LCA and LCC. Int. J. Life Cycle Assess. 18 (9), 1722–1733.

Hellweg, S., Milà i Canals, L., 2014. Emerging approaches, challenges and opportunities in life cycle assessment. Science 344, 1109–1113.

Hong, T., Kim, J., Lee, M., 2019. A multi-objective optimization model for determining the building design and occupant behaviors based on energy, economic, and environmental performance. Energy 174, 823–834.

Hoogmartens, R., Van Passel, S., Van Acker, K., Dubois, M., 2014. Bridging the gap between LCA, LCC and CBA as sustainability assessment tools. Environ. Impact Assess. Rev. 48, 27–33.

Hunkeler, D., Lichtenvort, K., Rebitzer, G., 2008. Environmental Life Cycle Costing. CRC Press, Boca Raton, FL.

Invidiata, A., Lavagna, M., Ghisi, E., 2018. Selecting design strategies using multi-criteria decision making to improve the sustainability of buildings. Build. Environ. 139, 58–68.

Islam, H., Jollands, M., Setunge, S., 2015. Life cycle assessment and life cycle cost implication of residential buildings-a review. Renew. Sust. Energ. Rev. 42, 129–140.

ISO, 2006a. ISO 14040:2006. Environmental Management—Life Cycle Assessment—Principles and Framework. International Organization for Standardization, Geneva, Switzerland.

ISO, 2006b. ISO 14044:2006. Environmental Management—Life Cycle Assessment—Requirements and Guidelines. International Organization for Standardization, Geneva, Switzerland.

ISO, 2011. ISO 15686-5:2011 Buildings and Constructed Assets—Service Life Planning: Part 5, Life-Cycle Costing. ISO 15686. International Organization for Standardization, Geneva, Switzerland.

Jouini, M., Burte, J., Biard, Y., Benaissa, N., Amara, H., Sinfort, C., 2019. A framework for coupling a participatory approach and life cycle assessment for public decision-making in rural territory management. Sci. Total Environ. 655, 1017–1027.

Kolotzek, C., Helbig, C., Thorenz, A., Reller, A., Tuma, A., 2018. A company-oriented model for the assessment of raw material supply risks, environmental impact and social implications. J. Clean. Prod. 176, 566–580.

Kouloumpis, V., Azapagic, A., 2018. Integrated life cycle sustainability assessment using fuzzy inference: a novel FELICITA model. Sustain. Prod. Consump. 15, 25–34.

Langdon, D., 2006. Literature Review of Life Cycle Costing (LCC) and Life Cycle Assessment (LCA). Draft Literature Review for LCC Methodology Project, Management Consulting, Available at http://www.tmb.org.tr/arastirma_yayinlar/LCC_Literature_Review_Report.pdf. (Accessed 4 February 2019).

Liamsanguan, C., Gheewala, S.H., 2008. LCA: a decision support tool for environmental assessment of MSW management systems. J. Environ. Manag. 87 (1), 132–138.

Mah, C.M., Fujiwara, T., Siong Ho, C., 2018. Life cycle assessment and life cycle costing toward eco-efficiency concrete waste management in Malaysia. J. Clean. Prod. 172, 3415–3427.

Means, P., Guggemons, A., 2015. Framework for life cycle assessment (LCA) based environmental decision making during the conceptual design phase for commercial buildings. Procedia Eng. 118, 802–812.

Miah, J.H., Koh, S.C.L., Stone, D., 2017. A hybridised framework combining integrated methods for environmental life cycle assessment and life cycle costing. J. Clean. Prod. 168, 846–866.

Perera, O., Morton, B., Perfrement, T., 2009. Life Cycle Costing: A Question of Value. International Institute for Sustainable Development.

Perriman, R.J., 1993. A summary of SETAC guidelines for life cycle assessment. J. Clean. Prod. 1, 209–212.

Ramasamy, S.V., Titchener-Hooker, N., Lettieri, P., 2015. Life cycle assessment as a tool to support decision making in the biopharmaceutical industry: considerations and challenges. Food Bioprod. Process. 94, 297–305.

Reidy, R., Davis, M., Coony, R., Gould, S., Mann, C., Sewak, B., 2015. Guidelines For Life Cycle Cost Analysis, Land and Buildings. Stanford University, available at https://lbre.stanford.edu/sites/all/lbre-shared/files/docs_public/LCCA121405.pdf. (Accessed 14 February 2019).

Rocchi, L., Kadzinski, M., Menconi, M.E., Grohmann, D., Miebs, G., Paolotti, L., Boggia, A., 2018. Sustainability evaluation of retrofitting solutions for rural buildings through life cycle approach and multi-criteria analysis. Energy Build. 173, 281–290.

Russo Garrido, S., Beaulieu, L., Telles do Carmo, B.B., 2018. The social value of products: what can it be and can it enrich social life cycle assessment? In: Social LCA People and Places for Partnership 6th SocSem Pre-proceeding, Fruitop Thema, Pescara. ISBN: 978-2-9562141-1-3, pp. 43–46.

Sala, S., Vasta, A., Mancini, L., Dewulf, J., Rosenbaum, E., 2015. Social Life Cycle Assessment—State of the Art and Challenges for Supporting Product Policies. EUR 27624 EN. https://doi.org/10.2788/253715 ISBN 978-92-79-54054-7 (print); ISBN 978-92-79-54055-4 (PDF).

Santos, P., Carvalho Pereira, A., Gervásio, H., Battencourt, A., Mateus, D., 2017. Assessment of health and comfort criteria in a life cycle social context: Application to buildings for higher education. Build. Environ. 123, 625–648.

Schmidt, M., Crawford, R.H., 2017. Developing an integrated framework for assessing the life cycle greenhouse gas emissions and life cycle cost of buildings. Procedia Eng. 196, 988–995.

SETAC, 1993. Guidelines for Life—Cycle Assessment: A 'Code of Practice'. SETAC, Brussels.

Soltanpour, Y., Peri, I., Temri, L., 2018. Discussing features of social measures important in SLCA impact indicators' selection. In: Social LCA People and Places for Partnership 6th SocSem Pre-Proceeding, Fruitop Thema, Pescara. ISBN: 978-2-9562141-1-3, pp. 52–56.

Tillman, A.M., Baumann, H., 2004. The Hitch Hiker's Guide to LCA. Studentlitteratur AB, Lund, Sweden.

Toniolo, S., Mazzi, A., Pieretto, C., Scipioni, A., 2017a. Allocation strategies in comparative life cycle assessment for recycling: considerations from case studies. Resour. Conserv. Recy. 117, 249–261.

Toniolo, S., Mazzi, A., Fedele, A., Aguiari, F., Scipioni, A., 2017b. Life cycle assessment to support the quantification of the environmental impacts of an event. Environ. Impact Assess. Rev. 63, 12–22.

Trevisani Juchen, R., Bonacina de Araujo, J., Chabrawi, A.M.R.O., Ugaya, C.M., 2018. Preliminary evaluation of data collection methods for SLCA studies. In: Social LCA People and Places for Partnership 6th SocSem Pre-proceeding, Fruitop Thema, Pescara. ISBN: 978-2-9562141-1-3, pp. 22–27.

UNEP/SETAC, 2009. Guidelines for Social Life Cycle Assessment of Products. United Nations Environment Programme. ISBN: 978-92-807-3021-0. DTI/1164/PA.

Vasta, A., Sala, S., Dewulf, J., 2015. Application of S-LCA at micro and macro scale: an overview. In: Sala, S., Vasta, A., Mancini, L., Dewulf, J., Rosenbaum, E. (Eds.), Social Life Cycle Assessment—State of the Art and Challenges for Supporting Product Policies. Publications Office of the European Union, Luxemburg, pp. 31–38. European Commission, Joint Research Centre, Institute for Environment and Sustainability.

Woodward, D.G., 1997. Life cycle costing-theory, information acquisition and application. Int. J. Proj. Manag. 15 (6), 335–344.

Xue, X., Cashman, S., Gaglione, A., Mosley, J., Weiss, L., Ma, X.C., Cashdollar, J., Garland, J., 2019. Holistic analysis of urban water systems in the greater Cincinnati region: (1) life cycle assessment and cost implications. Water Res. X. 100015.

Zanghelini, G.M., Cherubini, E., Soares, S.R., 2018. How multi-criteria decision analysis (MCDA) is aiding life cycle assessment (LCA) in results interpretation. J. Clean. Prod. 172, 609–622.

CHAPTER

4

Life cycle sustainability assessment: An ongoing journey

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4.1 Introduction

Considering the established concept of "Sustainability", as proposed by the Brundtland report (United Nations General Assembly, 1987), and the three pillars model developed by Elkinton (1998), several aspects are to be accounted in the evaluation of products and processes. In fact, as "Sustainable development is development that meets the needs of present without compromising the ability of future generations to meet their own needs" (United Nations General Assembly, 1987), such evaluation should be performed over a system the boundaries of which comprise all the element required to avoid the shifting of burdens among generations, over both time and space.

The implicit call for fairness at intergenerational (Klöepffer, 2008) and intragenerational level requires an extension of a proper sustainability assessment to scope horizontal and longitudinal dimension of impacts triggered by anthropic activity. Therefore, the whole range of impacts and effects induced must be assessed, based on the different aspects touched on and the timing in which these effects unravel. For these reasons, on one hand, "intact environment, social justice, and economic prosperity" (Finkbeiner et al., 2010: 3310) should represent the final goal of each application and the yardstick of the assessment, in order to address the horizontal dimension. On the other hand, the application of a life cycle thinking lens is required to explore the longitudinal dimension of the impacts and consider both the direct and indirect impacts triggered throughout the different phases of the product or process life and the time-scale of the impacts, considering the relationship between present needs and future opportunities. The proposed EU Integrated Product Policy stressed how life cycle perspective is intrinsically inherent to the greening of the product development process (Charter, 2001), but its systematic application is required to trigger the transition required at global

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market level and, consequently, it must be necessarily verified through a quantitative approach.

As reported by Guinée et al. (2011), the attention towards the environmental component of the impacts developed by the manufacturing, use, and disposal of goods stepped into diffuse awareness far earlier than the other components. This aspect was, in fact, detailed in terms of energy, resource use, efficiency, and pollution control (Assies, 1992) in the late 1960s and early 1970s, when the first rudimentary environmental assessments were realized. The very first experience reported by Guinée et al. (2011), in particular, developed by the Midwest Research Institute (MRI) for the Coca Cola Company (unpublished) in 1969, aimed at analyzing a set of products through the application of a framework defined as "resource and environmental profile analysis" (REPA), with a life cycle approach. Resource efficiency in the production chain and environmental issues represented the main scope of the assessment, evaluated with a company-oriented lens, but setting a crucial turning point for the sector.

A comprehensive and solid theoretical framework for what would subsequently be defined as "environmental life cycle assessment" (E-LCA) appeared only later, in the 1990s, with a remarkable contribution of both scholars and practitioners. In this sense, as underlined by Guinée et al. (2011), on one hand, several journals kept the pace with the evolution of the research conversation, e.g., Journal of Cleaner Production, Resources, Conservation, and Recycling, International Journal of LCA, Environmental Science & Technology, and the Journal of Industrial Ecology; on the other hand, the Society of Environmental Toxicology and Chemistry (SETAC) played a leading role in the definition of the LCA practice, stating its quantitative nature and the strive towards the standardization that should have been undertaken.

The first reported example of the extension of the scope of LCA to include not only the environmental dimension, but the three pillars model (Elkinton, 1998; Remmen et al., 2007), as suggested by Klöepffer (2008), is related to product line analysis (produktlinienanalyse) by the Oeko-Institut in 1987, but it is only far later that a comprehensive conceptual framework appeared. As highlighted by Zamagni (2012), the very concept of sustainability has undergone a mutation throughout the last 40 years within the research field, experiencing an extraordinary increase in interest, almost unparalleled by other topics.

The implementation of sustainability-oriented approaches in research, development, and manufacturing of product, as well as process design and management, requires the application of a systemic perspective in the decision-making. In particular, as stated by Finkbeiner et al. (2010), the shift towards sustainability implies a new paradigm, based on an "active, international, multicriteria, and stakeholder driven" approach (Finkbeiner et al., 2010: 3310), overcoming the old one, i.e., "reactive, national, single-issue and, government driven environmental protection" (Finkbeiner et al., 2010: 3310).

For this reason, visualizing the ideal process of development of the LCSA concept, as proposed by Finkbeiner et al. (2010), started from the application of the general idea of life cycle thinking, with an increased awareness towards resource scarcity and environmental protection against negative effects triggered during manufacturing, use, and disposal of products; this to be followed by the implementation of a single-issue impact assessment framework, such as carbon footprint and water footprint, followed by integrated life cycle assessment (LCA), mainly focused on environmental impacts. The preeminence attributed to the environmental component has been reasonably justified by the fragility and impossibility to compensate large scale or long-term impacts (Klöpffer 2003). Resource efficiency and eco-efficiency assessment have, then, been developed, opening the era of more comprehensive assessment.

The conceptual approach for Life Cycle Sustainability Assessment (LCSA) has been, in fact, summarized by Klöpffer (2003, 2008) as an addition of environmental LCA (LCA), economic LCA (life cycle costing (LCC)), and social LCA (SLCA), based on the very same inventory of material and energy flows:

$$\mathrm{LCSA} = \mathrm{LCA} + \mathrm{LCC} + \mathrm{SLCA}$$

In order to take a significant step towards the LCSA, overcoming the limits of LCA, the Sixth Framework Program of the European Commission, in 2006, supported CALCAS, i.e., coordination action for innovation in life cycle analysis for sustainability (Klöepffer, 2008; Guinée, et al., 2011). Given the complexity of the theme and the extent of the stakeholder audience, a multidisciplinary lens has been applied on the subject, to provide the governance system with a decision support tool.

In the following years, the body of knowledge has been increasing progressively, in both research conversation and practice standardization. In order to explore the development of LCSA, a literature review has been performed, based on Web of Science database (http://apps.webofknowledge.com; Accessed March 1, 2019). A total amount of 230 papers resulted from the search based on titles of papers published between 1997 and the first quarter of 2019. A brief descriptive analysis of the body of papers identified is reported in the following.

The median of publications per year is six, and this was reached in 2010 (Fig. 4.1). Of the total papers, 89% have been published from 2011 on, thus highlighting how the research field has been developing in the last decade, with peaks in 2016 and 2018, and a remarkable increasing trend in 2019, when nine papers had been already published in the first quarter

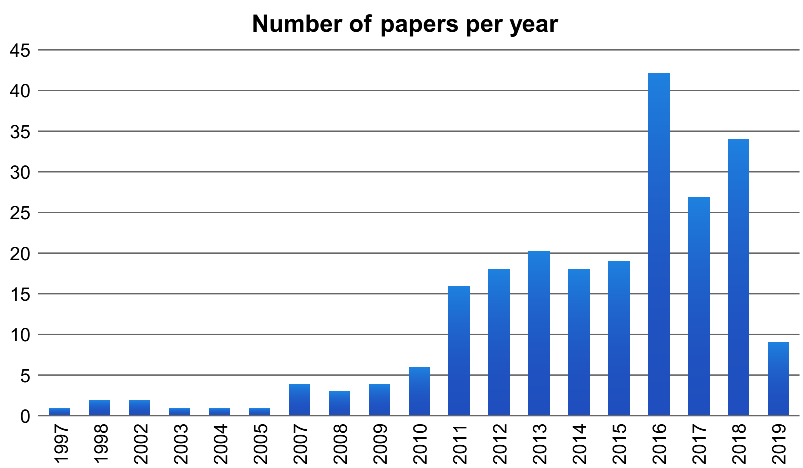


FIG. 4.1 Papers published from 1997 to 2019 (1st of March) by years.

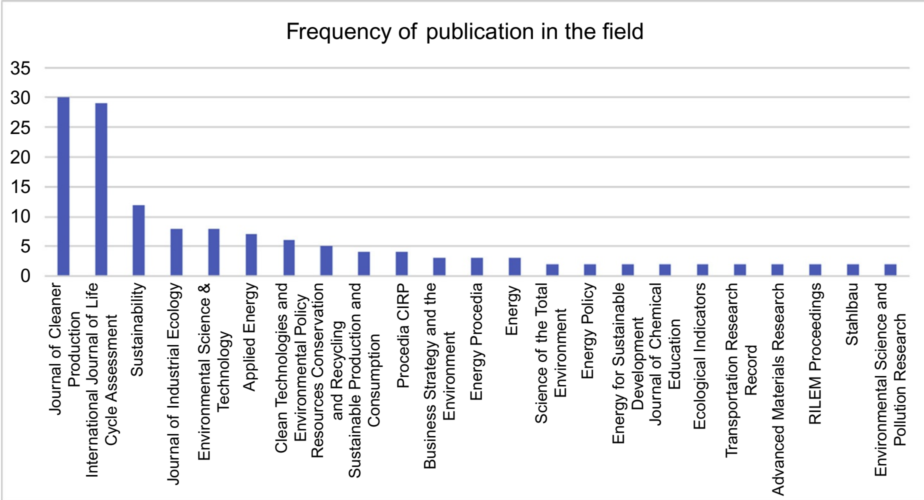


FIG. 4.2 Papers published in the field by journal.

of the year. The mounting attention dedicated to the topic by the research community is testified by the remarkable number of journals proposing related papers. This element is also marked by the distribution of the publications among them (Fig. 4.2).

A total of 109 different journals and conference proceedings have so far accomplished the publication of at least one paper on LCSA. Considering the overall publication frequency, the first seven journals, in terms of frequency of publications, have taken care of the publication of about 50% of the total body of papers available.

As presented in the following chart (Fig. 4.3), the Journal of Cleaner Production and International Journal of Life Cycle Assessment developed the main research conversation, with 15% and 14% of the papers published in the field, respectively. While the former has actively contributed in the last years in particular, e.g., with three of nine overall publications in 2019, the latter seems to have moved forward in the research conversation and its contribution was mainly exploited in 2013 (10 papers). The following years have been characterized by an almost constant, yet limited, number of papers.

The present chapter aims to accompany the reader in the ongoing journey towards a comprehensive and standardized Life Cycle Sustainability Assessment. As a starting point, a brief excursus on the rudimentary experiences published in the field will be offered, together with the progressive refinement of LCSA framework definition, to proceed, then, with an overview of the standardization pathway. With the growing pool of applications and the increasing consistency of literature, a concise anthology of case studies has been collected. As a final note, an overlook of prompts for future development is shared with the reader, offering a perspective over opportunities and space for valuable contribution to the field of research.

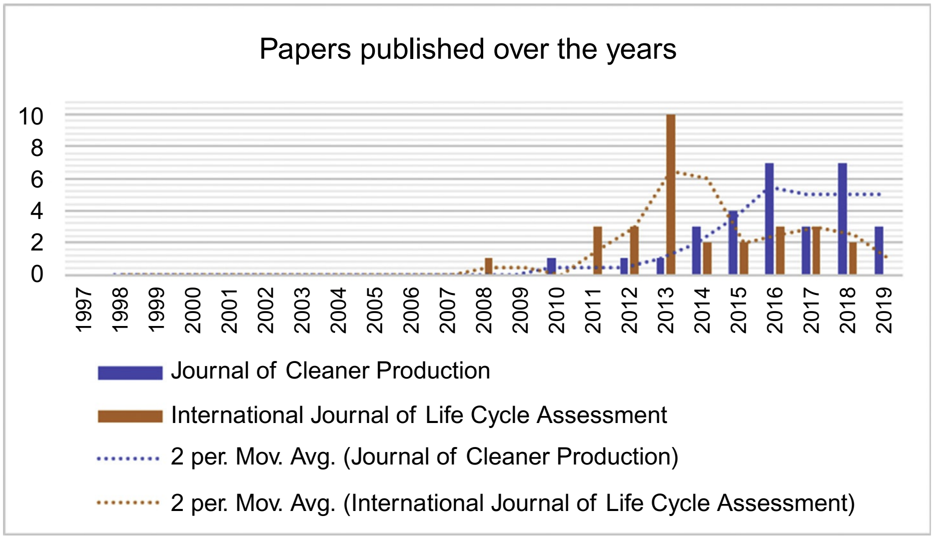


FIG. 4.3 Paper published in the field by the two leading journals (Journal of Cleaner Production and International journal of Life Cycle Assessment).

4.2 Early years: From concept to scheme

Almost in parallel with the first practical application of (environmental) LCA, the late 1990s experienced the preliminary drafting of LCSA concept and its possible application in a comprehensive sustainability framework. As LCA had been defined in 1993 by SETAC as "a process to evaluate the environmental burdens associated with a product or process by identifying and quantifying energy and materials used and wastes release to the environment" (SETAC, 1993), researchers supported a progressive enlargement of the assessment boundaries towards LCSA, starting from the very same sectors where LCA had made the first steps. In fact, an early hint to the pathway towards the conceptualization of sustainability assessment applied in a life cycle perspective was presented by Selmes et al. (1997), as cited by Boron et al. (2017), to the 75th anniversary celebrations of the Institution of Chemical Engineers. The same work (Klöpffer, 2003) responsible for the first theorization of LCSA framework, as presented in the previous section, emphasized the role of chemistry in meeting the goals of sustainable development. Therefore, its preeminence as testing ground for the LCSA application is regarded as an opportunity for operationalizing sustainability in a field where both processes and products are carriers of potential hazard for the environment and products are widespread in the different industrial sectors as well as on the market.

As a first step, the integration of the economic aspects into sustainability evaluation appeared quite early (Eyerer, 1996; Finkbeiner, 2010). This is obviously due to the close connection of such concepts to the overall performance evaluation of a product or process and the business model related. On the other hand, the social dimension of sustainability has only recently been implemented into an operative LCSA scheme, even though its formal inclusion was outlined during the very early stage of development. In fact, as reported by Finkbeiner et al. (2010), already in the last years of the 20th century, the triple bottom-line of sustainability had been translated into several evaluation frameworks. In the early years, however, the research community appeared unable to develop a common framework for the attribution of a relative weighting among the three pillars of sustainability, namely environmental, economic, and social aspects (Klöpffer, 2003).

Following examples provided by Finkbeiner et al. (2010), the three perspectives embodied into LCSA has been translated by into a ternary diagram, namely the life cycle sustainability triangle (LCST). In analogy with the geological characterization of soil textures or chemical mixtures, in fact, any triple-parameters scheme may be evaluated. In this case, as already outlined by Hofstetter et al. (1999), the environmental impacts, environmental, economic, and social impacts can be visualized, based on the relative weights attributed. The representation of a hypothetical weighting set is given in Fig. 4.4 (Finkbeiner et al., 2010). The same scheme is suggested as a valuable tool for scenarios comparison. In particular, the evaluation of the relative weighting is accomplished by following the dominance patterns along the axes, starting from the three corners, where each of the performance areas of evaluation (i.e., environmental, economic, and social) are individually rated 100%. The general rule is, evidently, that the sum of the weighting factors must be equal to 100%.

A second scheme proposed by Finkbeiner et al. (2010) is the life cycle sustainability dashboard (LCSD), as outlined by Hardi and Semple (2000) and detailed by Traverso and Finkbeiner (2009). Fig. 4.5 reports the visualization of the LCSD, as a composition of three different and free-standing evaluations, separately performed over the three relevant aspects of sustainability.

In accordance with Klöpffer (2003), the abovementioned scheme implies that the three pillars of sustainability have to be assessed separately beforehand. Only in light of such results can the comprehensive sustainability assessment be accomplished and, therefore, further

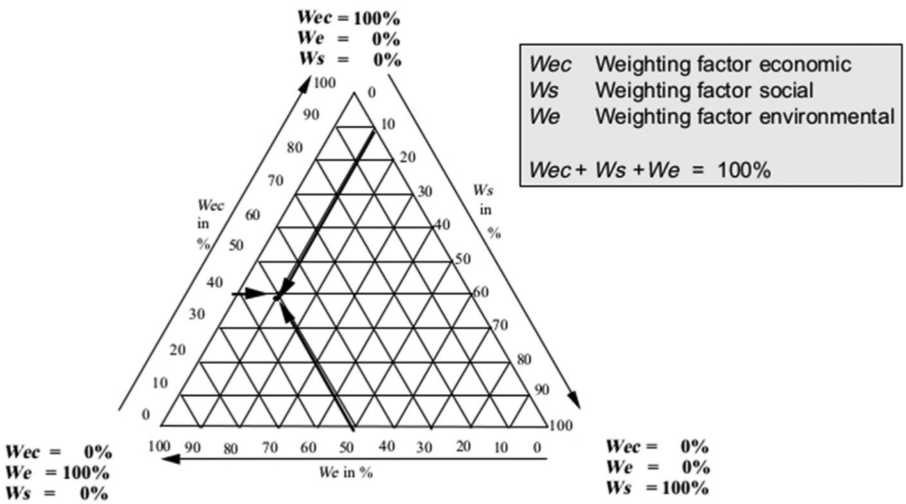


FIG. 4.4 Example of a weighting scheme for LCSA, as presented by Finkbeiner et al. (2010).

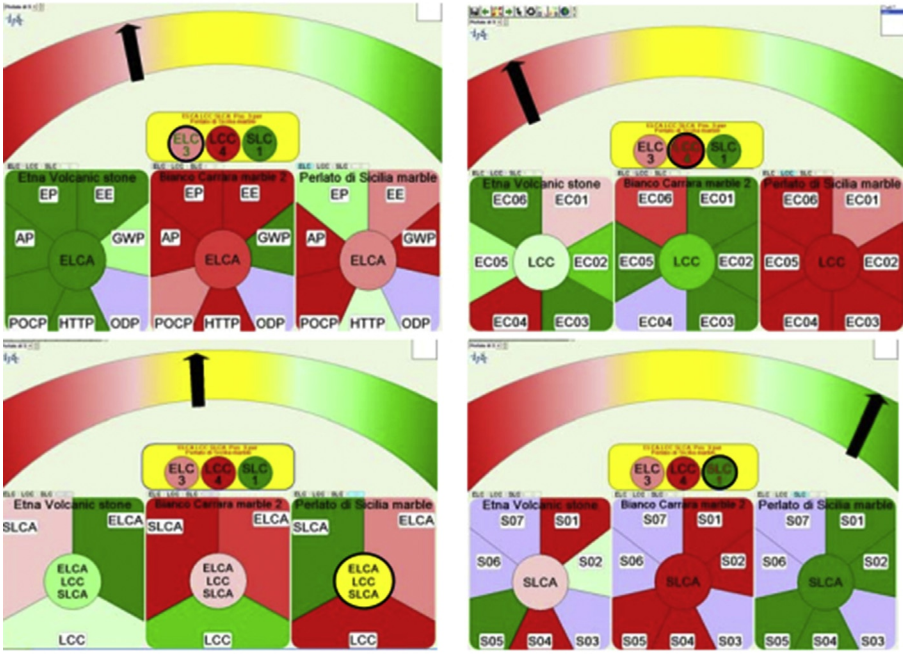


FIG. 4.5 LCSD graphical scheme (Finkbeiner et al., 2010).

systemic evaluations may be integrated. In particular, the modular approach allows the implementation of political and geographical specificity into the assessment, given the place specificity of sustainability-related issues and their possible longitudinal dynamics (Coenen and Truffer, 2012).

The following years have been characterized by the advance of integrated and progressively more complex assessment schemes, such as the one proposed by Halog and Manik (2011) and reported in Fig. 4.6; in which several multilevel approaches are proposed to integrate the traditional LCA-based scheme, such as multicriteria decision analysis (MCDA) or multiobjective decision making (MODM) to incorporate stakeholders inputs and dynamic interconnections, data envelopment analysis (DEA) for eco-efficiency evaluation, analytic hierarchy process, and stakeholders/experts analysis for sustainability criteria definition.

4.3 Pathway to standardization: The role of LCI/SETAC/UNEP in framework definition

As it is largely recognized in LCSA, as in several other topics, a standardization is necessary in order to achieve a common and widely shared description of the principles and framework for formulating, conducting, and reporting LSCA approach and studies. The importance of a standardization of the processes and the methodologies to develop study,

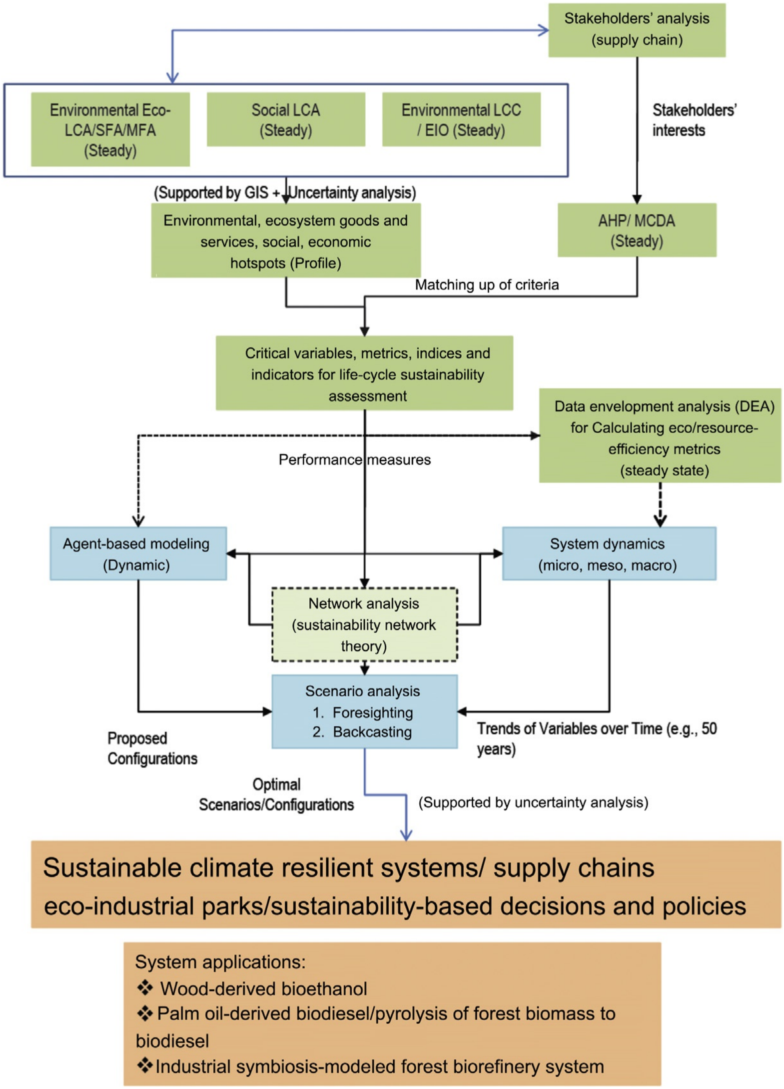


FIG. 4.6 Integrated methodological framework for sustainability assessment, as presented in Halog and Manik (2011).

to apply universally certified measure units, and to account and assess impacts is clear. LCSA can take advantage of consolidated experiences in previous similar tools and approaches.

4.3.1 Standard in life cycle perspective following LCA and LCCA

A life cycle assessment (LCA) study can be performed according to the internationally recognized guidelines, i.e., ILCD Handbook: General guide for Life Cycle Assessment—Detailed guidance (2010), and to specific standards, i.e., ISO 14040 and 14044. The boundary limits for the LCA include all the life cycle phases, from natural resources supply, raw materials processing, components manufacturing, to the end-of-life steps, passing through the maintenance phases and use.

It must be emphasized that LCA has been conceived, developed, and standardized to quantify the potential environmental impacts of goods and services and that it is essentially based on a linear stationary model, founded on technological and environmental relationships. Comparing results of different studies, for instance, is only possible if the assumptions and context of each study are the same. Since its first definitions, the International Standard IS0 14040 series provided principles and framework and methodological requirements for conducting LCA studies (IS0 14040, 1997). Whereas, LCA is already a standardized method (ISO 14044, 2006) and widely used to investigate the potential environmental impacts, LCSA has to be actually implemented and developed following a common recognized asset of regulation.

As is commonly acknowledged (Neugebauer, 2015), LCSA can be considered as the integration, or better, as resulting from the addition of the three dimensions of sustainability perspectives, i.e., economic, environmental, social, and according with several authors it can be presented as the following easy equation:

$$\mathrm{LCSA} = \mathrm{LCA} + \mathrm{LCC} + \mathrm{SLCA}$$

According particularly with Kloepffer (2006), LCA can represent a useful starting point to develop an integrated methodology, combining the three aspects. LCA has shown that quantification is possible and this advantage should be preserved in adding the economic and social aspects and to develop an integrated LCSA.

In this context, for a useful and effective LSCA standardization, the already existing LCA standardization approach can be applied; particularly rooted in some important pillars, usefully synthesized in the main few following definitions.

4.3.1.1 Glossary

First of all, such as in several other contexts, it is important to speak the same language. The glossary of terminology in LCA has been created to provide a common vocabulary for people around the world to use when they talk about LCA data and databases. The glossary uses the International Organization for Standardization (ISO) terminology, as far as it is available, and provides additional explanation (LCI-UNEP, 2011). The same glossary has to be assumed and improved also in LCSA.

4.3.1.2 Interfaces for data exchange

In order to communicate and share data, the database contents should be suitable for exchange via standard interfaces into other LCA software or systems. However, contents need first to be harmonized to avoid misunderstanding, misinterpretations, and unintended inconsistencies.

4.3.1.3 Basic methodological structure

As for any single assessment component (i.e., LCA, LCCA, SLCA), LCSA can also be carried out in four steps in a processual and iterative manner, which are composed of: goal and scope definition, inventory analysis, impact assessment (involving the four steps: classification, characterization, normalization, and analysis of data quality) and interpretation.

4.3.1.4 Functional unit and system boundaries

It is a very important point that different life-cycle based methods for sustainability assessment (Klöepffer, 2008) have to have the consolidated approach in the functional unit definition and use consistent—ideally identical—system boundaries. The primary purpose of a functional unit is to provide a reference to which the inputs and outputs are related, ensuring the comparability of LCSA results and defining the quantification of the identified function of the product/process/services. Also, in an LCSA, the inventory and impact indicators must be related to a common product functional unit, which is the basis of all techniques described. As with the S-LCA (UNEP/SETAC, 2011), it is recommended that the functional unit describes both the technical utility of the product and the product's social utility.

LCSA system boundaries must be defined according to the following definitions, which are well known in any life cycle thinking and life cycle assessment study:

- from cradle to gate, which means to collect data and information from raw material extraction to manufacturing and assembling of the product;
- from cradle to grave, from raw material extraction to their return to the environment as waste or emissions;
- from gate to gate, considering only what is inside the fence of the company, excluding supply and distribution;
- from gate to grave, which includes distribution phase, use, and end of life phases; and
- from cradle to cradle, which means to assume a circular economy perspective, thanks that all outputs (such as emissions, water, and waste) produced at end of life return in input, closing the loop.

It is recommended, whenever feasible, that a combined framework for impact assessment based on the individual S-LCA, LCC and (environmental) LCA frameworks is used (UNEP/SETAC, 2011).

4.3.1.5 Quali-quantitative inventory data

The assumption that primary data are the best option both from a qualitative and quantitative point of view seems to be definitely true also for LCSA. At the same time, complete and effective existing databases, such as Ecoinvent[a] or the European ILCD (Life Cycle Database of the International Life Cycle Data System) providing a common basis for consistent, robust, and quality-assured data and studies, can contribute to the correct completion of all data-collecting catalogs. Inventory analysis involves data collection and calculation procedures to quantify relevant inputs and outputs of a product system. The process of conducting an inventory is iterative. Data must respond to main properties such as precision, completeness, temporal consistency, geographic and technological connection.

If for LCA and LCC a quantitative approach is perfectly recognized as possible and useful, on the contrary, there is a wide debate about the opportunity to utilize quantitative inventory

[a] See Ecoinvent (2003).

data for social assessment, in SLCA, or to opt for qualitative data and indicators. To achieve the most precise assessment and interpretation, a combination of quantitative and qualitative data, indicators, and analysis can be considered at this moment as the best option (Grießhammer et al., 2006). Maybe, quantitative data and indicators are not able to include and afford all social impact effects. At the same time, qualitative results can be turned into semiquantitative outcomes, as suggested by Grießhammer et al. (2005).

4.3.1.6 Impact categories

For LCSA, they should be chosen in accordance with internationally recognized categorizations/standards. For an LCSA study, it is recommended that all impact categories that are relevant across the life cycle of a product are selected. These should follow the perspectives provided by each of the three techniques and consider the stakeholder views when defining the impact categories. Furthermore, by considering all relevant impact categories from a cross-media, multidimensional (social, economic, and environmental), intergenerational, and geographic perspective, potential trade-offs can be identified and assessed (UNEP/SETAC, 2011).

For the LCA impact assessment, an evolution of several methods during the time can be considered as an abundant basin where it is possible find several calculation methods. Starting from Eco-indicator99,[b] performed by the Ministry of Housing, Spatial Planning, and the Environment, the Netherlands (1999), and updated in Recipe in 2009, across EDIP (Denmark, 2003), updated in Impact World+ (2016) and ILCD (2012), and ultimately leading to the EU LC-Impact (2016).

During the last years, also for LCC, several models have been developed for the determination of the economic impact of a product, concerning its whole life cycle (Durairaj et al., 2002). In particular, SETAC specified an LCC methodology (Hunkeler et al., 2008) providing an assessment of the costs of a product across its entire life cycle and published guidelines describing the method and a code of practice (Swarr et al., 2011). LCC aims at enabling options to be more effectively evaluated considering the impact of all costs, assisting in the effective management of processes, and facilitating choice between different alternatives. In terms of LCC impact categories, aggregated cost data provides a direct measure of impacts.

For an S-LCA, from a social perspective, following the UN declaration on economic, social, and cultural rights, the potentially most affected stakeholder groups identified are mainly workers and local community groups. According to the definition of subcategories of the S-LCA guidelines (UNEP/SETAC, 2009), "they aim to assess whether practices concerning wages are in compliance with established standards and if the wage provided is meeting legal requirements, whether it is above, meeting, or below industry average, and whether it can be considered as a living wage." For SLCA, the indicators of human rights, safety, cultural heritage, working conditions, have to be considered, and mainly health and education, according with SDGs UN Agenda 2030 (UN, 2015) where the goal n. 3 and 4—respectively Health and Education—represent tremendously meaningful hotspots.

In that way, an LCSA can also be carried on at subcategory level for each stakeholder group and not at the level of midpoint or impact categories. In terms of social impacts on stakeholder

[b]Eco Indicator 99 – Reports, Ministry of Housing, Spatial Planning and the Environment.

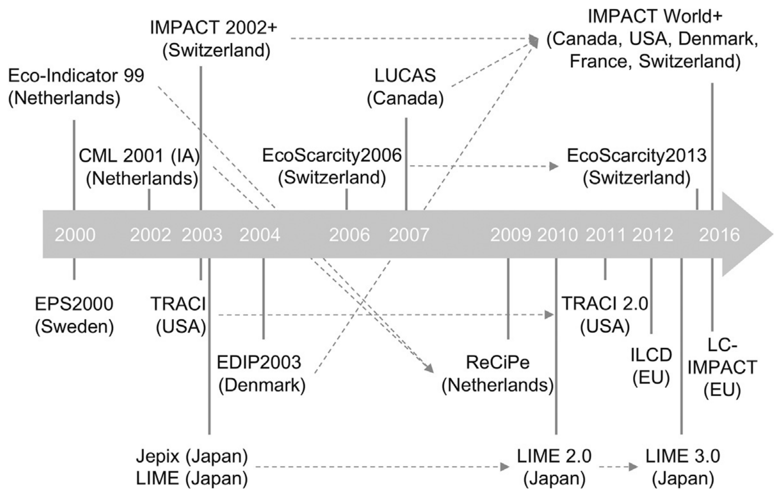


FIG. 4.7 Calculation Method evolution in LCA. *Courtesy of Rosenbaum, R.K., 2014. Selection of impact categories, category indicators and characterization models. In: Curran, M.A. (Ed.), Goal and Scope Definition in Life Cycle Assessment, LCA.*

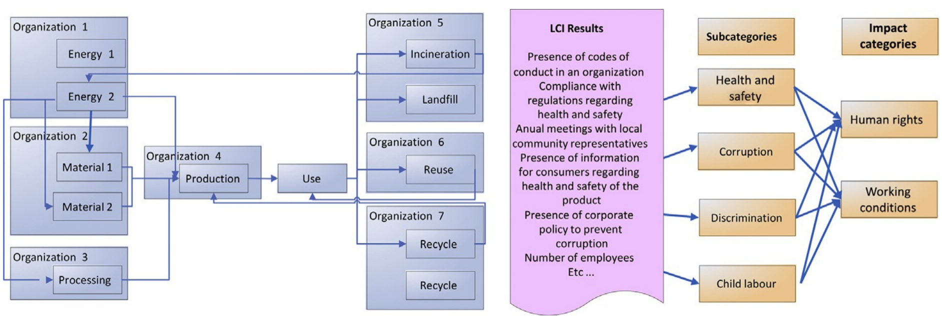


FIG. 4.8 Examples of a social life cycle inventory (S-LCI) and interrelationships to subcategories and impact categories. *Courtesy of UNEP/SETAC, 2011. Towards a Life Cycle Sustainability Assessment: Making Informed Choices on Products, UNEP/SETAC Life Cycle Initiative.*

category "society," several subcategories, such as contribution to national economy and national budget, or employment creation, risks, impact, or conflicts, were also described (Tsurukawa et al., 2011) (Figs. 4.7 and 4.8).

4.3.1.7 Indicators

According to Neugebauer et al. (2015), an indicator can be defined as "something representing the state of a certain aspect or effect used to measure a progress towards a stated goal and it can function as variables, parameters, measures, measurement endpoints or thresholds." However, indicators have been specially defined as a tool to measure a causal effect. For LCSA inventory, as well as for LCA, midpoint and endpoint indicators can be distinguished, describing each step along the cause–effect chain.

In several studies, indicators for environmental impacts are generally more easily measurable and they have been intensively studied and analyzed in the past. According to Maranghi et al. (2016), many indicators have been largely applied in several different studies, mainly thanks to the great standardization work done over the years; at the same time, it is necessary to recognize which indicator can be more significant and crucial for each specific assessment.

4.3.1.8 Interpretation and evaluation of results

The interpretation of results in SLCA, such as in standardized LCA methodology, has to verify completeness and full overlay of all impacts, consistency and sensitivity, relevance of information, and engagement of stakeholders Evaluation can use a wide range of quantitative, semiquantitative, or totally qualitative methods, standardized or specifically performed for a certain product. The actual evaluation of social aspects has to be devoted to finding solutions to put in action.

It is therefore a firm belief that the evaluation of circularity and of whole life cycle thinking strategies should be performed not only from an environmental life cycle perspective, but also including social and economic considerations (Princigallo et al., 2016). To support the decision-making process, environmental life cycle indicator scores and economic criteria can be combined with social assessment together with a multicriteria decision analysis methodology, which allows the weighting of the different scores.

For the LCSA framework improvements, Neugebauer et al. (2015) suggested a new approach, named "tiered approach," to implement LCSA considering an indicator hierarchy and implementing evaluation phase thanks to a whole assessment. For an effective practical implementation of LCSA, the authors defined three levels of analysis, starting with meaningful indicators on level 1 (defined as "sustainability footprint"), then adding additional indicators, such as best practices at a second level, and concluding with a complete set of indicators for a whole sustainable performance comprehensive assessment at the third level. All indicators have to be performed for each level, taking into consideration the main properties they must have, i.e., relevance, robustness of the method, and feasibility (Fig. 4.9).

4.3.2 The life cycle initiative as a tool for LCSA application

The UN Environment Life Cycle Initiative[c] is a public-private, multistakeholder partnership enabling the global use of credible life cycle knowledge by private and public decision makers. It has existed for more than 15 years, since being launched by UN Environment and the Society of Environmental Toxicology and Chemistry (SETAC) in 2002. It is life cycle approach oriented, supporting decisions and policies makers towards the "shared vision of sustainability as a public good" by engaging its multistakeholder partnership (governments, businesses, and scientific and civil society organizations). Life Cycle Initiative is also promoting an "Integration of social aspects into LCA." The system of methods used in LCA was carried out and verified by UNEP-SETAC, in order to understand whether and how the social aspects can be considered together with the environmental ones or combined and connected to develop a social LCA in order to achieve a complete LCSA.

[c]https://www.lifecycleinitiative.org/about/about-lci/.

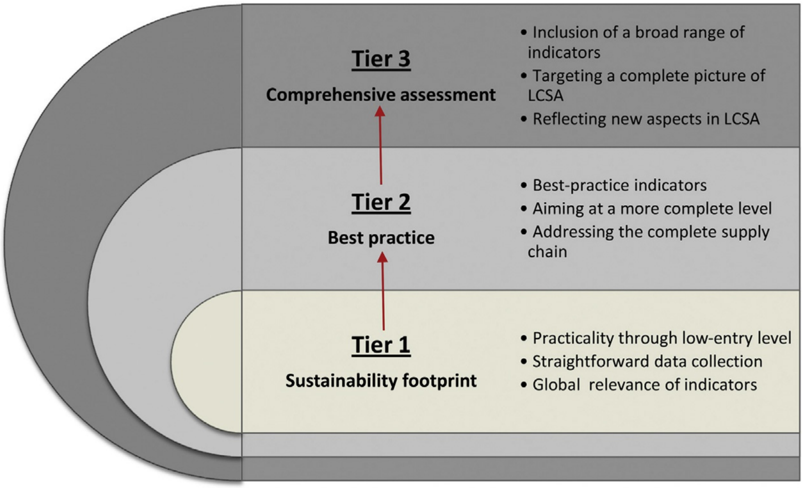


FIG. 4.9 Structure for the Tiered approach (Neugebauer et al., 2015).

A preeminent purpose of the Life Cycle Initiative is to encourage a life cycle thinking approach and knowhow also in the UN 2030 agenda actions in order to achieve sustainable development global goals in a faster and more efficient way. Fig. 4.10 shows the Theory of Change for the Life Cycle Initiative, linking its key deliverables with the expected impact as defined by the initiative vision, which can be considered as a fundamental example for LCSA methodology development.

The Life Cycle Initiative 2017–22 strategy document suggests technical advice to improve the applicability of methodologies for specific applications, to orient scientific research and

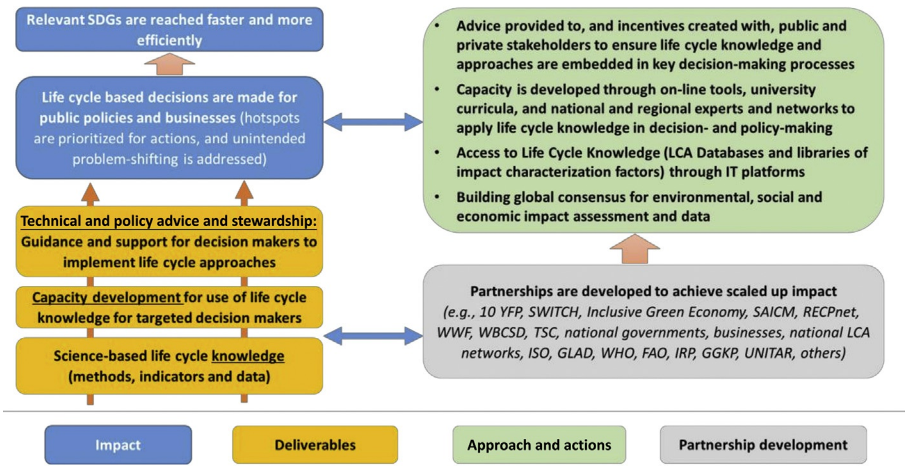


FIG. 4.10 Theory of Change of the Life Cycle Initiative (Life Cycle Initiative 2017–22 Strategy document, UN).

practical implementation by some actions oriented to implement a national hotspots analysis tool, working with certification schemes, and to develop data and methods on life cycle costing supporting sustainable public procurement. The initiative is also working on life cycle capacity development, aiming at generating the necessary skills and capacity for the global application of life cycle approaches and on a life cycle knowledge consensus and platform definition to ensure science-based global consensus building and to promote an access to life cycle knowledge as a public good.

4.3.3 Organization life cycle assessment (OLCA) promoting LCSA

Towards standardization over the long term, the development of a standard set of social indicator life cycle thinking at the organizational level is relevant, meaningful, and feasible using a similar framework to product LCA standards (OLCA). In the UNEP-LCI document *Towards a Life Cycle Sustainability Assessment*, they highlight "the need to provide a methodological framework for LCSAs and the urgency of addressing increasingly complex systems are acknowledged globally." In addition, a particularly interesting perspective is in terms of "organizational LCA," proposed by LCI-UNEP as well (UNEP, 2015).

Some guidelines and standards can contribute to achieve a useful standardization also in LCSA. For instance, the "Organisation Environmental Footprint (OEF) Guide" (European Commission, 2013), ISO/TR 14069 or the "Corporate Value Chain (Scope 3) Accounting and Reporting Standard" (WRI and WBCSD, 2011) can be usefully applied, not only for an OLCA, but also as standardization supporting LCSA.

In this context, once again, the principles of a holistic perspective and whole life cycle are applied, considering all life cycle stages for organization, the complete supply or value chains, from raw material and resources supply, through energy, material production, and manufacturing, to use and end-of-life treatment, and final disposal. Potential trade-offs can be considered and identified, taking into account all environmental, economic and social issues according with an LCSA, with a cross-media and multidimensional perspective. As already suggested from a general point of view, also in the case of OLCA, to support decision-making and management process, environmental life cycle indicator scores, economic and social criteria can be assessed thanks to a multicriteria decision analysis methodology, effectively supporting scores' weighting phase.

Organizational life cycle assessment (O-LCA) has significant potential to help corporations, authorities, institutions, and other organizations improve their environmental performance by providing the necessary, credible information for decision-making. Considerable efforts are underway to build global knowledge and capacity for understanding, developing, and promoting more sustainable products and services. One key effort is to increase the availability of foundational data on energy, materials, land, and water consumption, and on related emissions into water, air, and soil, so that we have comprehensive information on materials and products over their life cycle. This comprehensive information is obtained by the use of LCSA. As the technical basis for the practice of LCSA becomes more standardized and as more decisions are supported by this methodology, the demand for high quality documented, transparent, and independently reviewed data has increased tremendously (UNEP, 2014).

In 2013, the European Commission launched the draft of its OEF Guide, (European Commission, 2013) and 1 year later, ISO/TS 14072 (ISO, 2014) had been developed by the International Organization for Standardization. According to Finkbeiner and König (2013), most of the ISO 14044 (2006b) requirements (27 out of 31) can be usefully transferred from products to organizations. UNEP/SETAC Life Cycle Initiative started the project about O-LCA of definition exploring how to concern organizations in a wide LCA approach, representing an important milestone also in an LCSA development (UNEP, 2015).

4.3.4 Eco-efficiency and environmental product declaration (EPD) as a standardization tool for companies

Eco-efficiency analysis, conceived by the World Business Council for Sustainable Development (WBCSD, 2011), can be defined as "a management philosophy that encourages business to search for environmental improvements that yield parallel economic benefits." It is based on an actual and effective approach, supporting organizations to incorporate environmental and social issues into their procedures, actions, and policies.

Eco-efficiency can be considered as a tool for quantifying the relationship between economic value creation, social aspects, and environmental impacts, throughout the entire life cycle of a product. Thanks to this definition, it the robust relationship and similarity with LCSA seems to be clear. In that way, LCSA standardization can take advantage of the consolidated and well-known eco-efficiency methodology and standardization. ISO 14045 (2012) defined a standard for product systems describing principles, requirements and guidelines for eco-efficiency assessment. The Environmental Product Declaration (EPD) can be considered, in a certainly way, a sort of specific application of LCA. While the overall goal of an EPD is to provide information about the environmental impacts of products, at the same time it can be considered as a standardization tool to quantify the total impacts of a product or a system (Del Borghi, 2013).

Specific standards are available for environmental labels and declarations based on a whole life cycle approach. The International Standards Organization (ISO) classified existing environmental labels into three typologies—types I, II, and III—and specified the preferential principles and procedures for each one of them (ISO 14021, ISO 14024, and ISO 14025). An Environmental Product Declaration (EPD), also referred to as type III environmental declaration, is a standardized (ISO 14025, 2010) and LCA-based tool to communicate the environmental performance of a product (Grahl and Schmincke, 2007). There are a number of requirements for how the LCA should be performed to be used as basis for an EPD. That approach can be usefully extended considering an LCSA, adding cost and social assessment to environmental impact evaluation.

The EPD methodology can represent a guideline, concerning specific definition in terms of production modeling, kind of data, and data collection methods, indicators. With the aim to reach outcomes comparability between products, all requirements should be correctly defined and referred to product category rules (ISO 14025), which are documents providing guidelines for developing an EPD for a specific product category. In that way, EPD becomes a useful tool also to communicate results and details on products' performances.

4.4 LCSA development in two decades of practice: A case study anthology

Life cycle sustainability assessment methodologies have been applied in various jurisdictions. LCSA case studies application fall under two broad areas, namely, product development or performance assessment (Gbededo et al., 2018). A compendium of LCSA case studies are described in the following subsections. The summary of the LCSA case studies is presented in Appendix A.

4.4.1 Demolition processes for end-of-life building

Bozhilova-Kisheva et al. (2012), applied LCSA to assess two different demolition processes on a high-rise end-of-life (EoL) building (17 levels with 6525 m^2 gross floor area) in the Netherlands. The scope of demolition was on the complete demolition of the building. The assessment was performed on a functional unit of 1 m^2 gross floor area. In the demolition process, three different demolition methods were used for different floors of the 17 level building (see Table 4.1). But the top-down and high-reach methods were included in the LCSA because they can replace each other.

While both methods produce recyclable material streams, the quality of the material stream produced by high-reach method depends on the quality of dismantling. Top-down method, however, produce quality waste material stream irrespective of the quality of dismantling.

Due to lack of information for the ELCC and SLCA, the authors excluded background processes from the inventory items. In the ELCA, the inventory included materials from demolition (steel, red brick, concrete, and CDW mix), technical equipment, energy consumption, and waste treatment. The cost categories for the ELCC were selected following UNEP-SETAC *Environmental Life Cycle Costing: A Code of Practice*. The inventory items were costs for labor and equipment, energy, waste disposal, sale of recyclable materials, and 12% overhead. The SLCA indicators used in this study were selected from UNEP-SETAC guidelines for SLCA products, based on expert advice from the demolishing company and the company supervising the demolition operations. Company-specific data are irrelevant in SLCA decision-making because the processes are performed by the same company. However, process-specific indicators (hours of work created, and quantity of secondary resources produced) may produce different values for decision-making.

TABLE 4.1 Demolition methods applied to EoL building.

Demolition method	Area of building applied	Demolition method assessed by LCSA
Top-down method	Top seven floors	Assessed by LCSA
High-reach method	Middle five floors	Assessed by LCSA
Short-reach method	First two floors	Not assessed

For the same amount of materials demolished, the top-down method uses more technical equipment (equipment: bobcat machine, wire crane, and 20 t excavator; fuel: 4.12 L/m^2 GFA), but less fuel consumption compared to the high-reach method (equipment: 30 t excavator and 65 t excavator; fuel: 4.63 L/m^2 GFA).

The costs for waste disposal and sale of recyclable materials were assumed to be the same for both demolition methods because equal quantity and quality of materials are produced. For both demolition methods, the highest cost was attributed to capital cost (hiring of technical equipment). The labor cost for top-down method was three times higher than that for the high-reach method; however, the energy costs were comparable. In terms of social performance, the top-down method created 0.43 h/m^2 GFA of employment whereas the high-reach method created 0.10 h/m^2 GFA. The authors were confronted with methodological challenges including:

- computation of SLCA inventory the function unit;
- identification of connection points between inventories; and
- lack of easy access to data for the costs and social indicators for background processes.

To assess the strength and weaknesses of the LCSA template used in this case study, the authors concluded that the template should be applied to more complex and data intensive processes. In addition, detailed ELCA involves more inventory items (than used in this study), which would be practicable to assess using LCC and SLCA.

4.4.2 Reuse of waste electrical and electronic equipment components

In assessing the reusability of waste mobile phone components from waste electrical and electronic equipment (WEEE) in China, Lu et al. (2014) used LCSA to compare reuse to other end-of-life strategies (such as materials recovery and disposal). The functional units used by the authors were 100 waste mobile phones produced around the year 2010. The life cycle impact was measured using Eco-indicator 99 (EI99). The Eco-indicator 99 is an endpoint approach (Dreyer et al., 2003), which assesses environmental impact on human health, ecosystems, and natural resources (Lu et al., 2014) whilst integrating LCA uncertainties (Pushkar, 2013). In assessing the economic costs, Lu et al. (2014) considered only the costs incurred by stakeholders in the end-of-life stages. The social impacts of both recycling strategies on direct and indirect workers were determined following the *Guidelines for Social Life Cycle Assessment Products* and using mid-point evaluation indicators such as employment, housing and education. The results of their LCA and LCC assessment revealed that reuse is environmentally and economically friendlier than materials recovery but the SLCA does not show clear social benefit of reuse. In terms of new raw materials consumption, both strategies contributed positively to the environment. They found that the formal recycling sector creates less employment but offers higher wages and social guarantee as well as better health conditions than the informal sector. Many factors (such as time range, physical situation, speed of technology innovation, etc.) were identified to affect reusability of end-of-life WEEE, which should be considered in practice. The authors are of the view that LCSA could be used by waste recycling practitioners to select suitable and sustainable end-of-life strategies but recommends improvement in the integration of LCA, LCC, and SLCA.

4.4.3 Transportation fuels

LCSA methodology was assessed and tested on four different transportation fuels using multicriteria decision analysis (MCDA) by Ekener et al. (2018). The transportation fuels were petrol from Nigerian crude oil, petrol from Russian crude oil, ethanol from Brazilian sugar cane, and ethanol from United States corn.

LCA was conducted separately from "well to tank" and "tank to wheel" (mixed fuel) to ensure that environmental impacts from production to end-use of the product are considered separately and subsequently combined and aggregated to obtain the final LCA results. The SLCA data were obtained from a previous study conducted by Ekener-Petersen et al. (2014) to assess potential social impacts of various bio and fossil fuels. The data were supplemented by the introduction of "job creation" as a positive social impact from literature. Fig. 4.11 illustrates the LCSA approach used by Ekener et al. (2018). LCC was based on the direct cost borne by the producer but excluded fees and taxes associated with the product. The data for LCC were based on secondary data from Luo et al. (2009) (for petrol and sugarcane ethanol) and Pimentel and Patzek (2005) (for corn ethanol).

The multiattribute value theory (MAVT), which has been used in LCA and environmental management applications (Ferretti et al., 2014; Stefanopoulos et al., 2014; Apperl et al., 2015; Rahimi and Weidner, 2004), was applied to combine the results of LCA, SLCA, and LCC, and construct a sustainability index for the different transportation fuels. The sustainability indices were prioritized for three different stakeholder profiles (egalitarian, hierarchist, and individualist) used in other studies (Bachmann, 2013; Hacatoglu et al., 2015; De Schryver et al., 2013). Different stakeholder profiles prioritize the sustainability perspectives differently. The corresponding priority (in descending order of priority in brackets) of the different

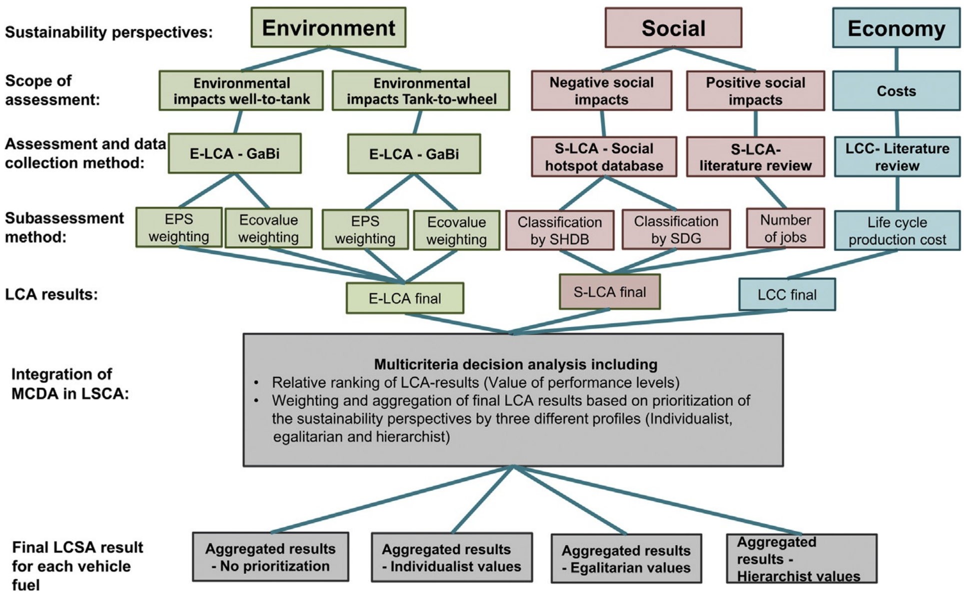


FIG. 4.11 Approach for LCSA of transportation fuels (Ekener et al., 2018).

stakeholder profiles are: egalitarian (social, environmental, economic), hierarchist (environmental, economic, social) and individualist (economic, environmental, social).

The LCA results show that the total environmental impact (from lower to higher impact) of the transportation fuels considering "well-to-tank" are in the order, ethanol-sugarcane < petrol < ethanol-corn. The reason for the high environmental impact of biofuel from corn is attributed to the large land requirement and the perceived consumption of fossil fuel in the production chain in the United States. But when considering "tank-to-wheel," the biofuels had lower environmental impacts compared to fossil fuels. The petrol from Nigerian oil and biofuel from Brazilian sugarcane showed more severe negative social impacts than petrol from Russian oil and biofuel from US corn. The positive social impacts were in the order: fossil fuels (lower positive impacts), biofuels from US corn (medium positive impacts), and biofuels from Brazilian sugarcane (high positive impacts). The total LCC was highest (0.0203 €/MJ) for ethanol from US corn but lowest (0.0111 €/MJ) for ethanol from Brazilian sugarcane. The total LCC for the petrol fell in between the biofuels (0.0132 €/MJ for Nigeria and 0.0126 €/MJ for Russia). The results were confirmed by the values of performance levels of the different transportation fuels for the different sustainability dimensions, as presented in Table 4.2.

The results of the relative sustainability ranking of the transportation fuels differed among the different stakeholder profiles (Fig. 4.11). This implies that different stakeholders will have different transportation fuel preferences. According to the authors, the dataset in the Ecoinvent database used for their study are about two decades old (2000–18) casting doubt on the current applicability of the sustainability performance results for the transportation fuels assessed.

4.4.4 Reinforced concrete buildings in seismic regions

Gencturk et al. (2016) developed an LCSA framework and used it to assess the impact of earthquake actions on lifetime structural performance of reinforced concrete buildings. The case study was applied to a reinforced concrete building (four-story three-bay RC moment resisting frame) located in San Francisco, California. The system boundary for the assessment covered the structural components of the entire building and the functional unit was from cradle to grave but excluded operation, maintenance, and nonseismic repair not directly related to

TABLE 4.2 Performance levels for different transportation fuels (Ekener et al., 2018).

	LCA		SLCA			
			Negative impacts			
Fuel	Ecovalue	EPS	SHDB	SDG	Jobs created	LCC
Petrol-Nigerian oil	0.693	0.558	0.152	0.113	0	0.652
Petrol-Russian oil	0.478	0.225	0	0	0	0.859
Ethanol-Brazilian sugarcane	1	1	0.728	0.699	1	1
Ethanol-US corn	0	0	1	1	0.333	0

EPS, Environmental Priority Strategies; *SDG*, Sustainable Development Goal; *SHDB*, Social Hotspot Database.

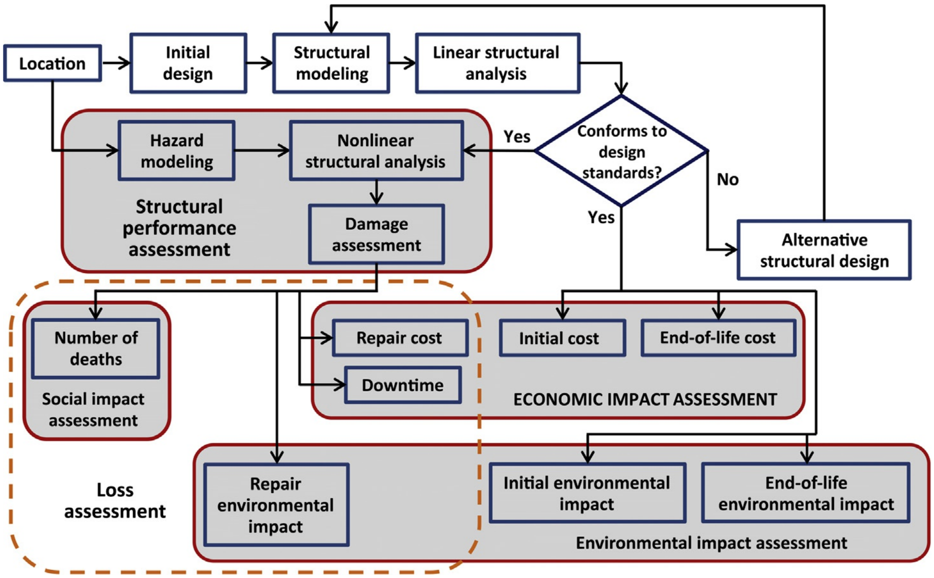


FIG. 4.12 Approach for LCSA of reinforced concrete buildings in seismic regions in San Francisco, California (Gencturk et al., 2016).

structural performance. In their framework, environmental impact was measured based on environmental emissions and waste generation. Similarly, a number of indicators were used to assess the economic impacts from both direct costs (materials, construction, operation, and end-of-life), indirect costs (downtime, loss/interruption of business, job loss, price increase, and supply disruption), and negative social impacts (deaths, injuries, stress, displacement, etc.). The proposed LCSA framework used by the authors is presented in Fig. 4.12.

The performance-based earthquake engineering (PBEE) methodology developed by Pacific Earthquake Engineering Research (PEER) Center was adopted to related seismic structural damage to losses. Following the PBEE framework, the loss assessment step involved LCCA, LCEIA, and LCSIA. Detailed explanation of the PEER PBEE framework has been presented by many authors (Porter, 2003; Deierlein et al., 2003; Moehle and Deierlein, 2004; Günay and Mosalam, 2013). The PBEE framework has extensively been used by many researchers, for example, to assess the seismic risks of new and existing buildings (Yang, 2013), community seismic resilience (Burton et al., 2016), and structures in fire (Lange et al., 2014).

The developed sustainability assessment framework was successfully applied to the RC building with the following findings:

- The greatest economic and environmental impacts occurred at the construction stage and material production stage, respectively. However, use and end-of-life phases contributed less environmental and economic costs.
- A robust design could considerably reduce downtime and the number of collapse and partial collapse cases as well as reducing the cost (10 times) and environmental impact (2 times) of the use phase.

The authors conclude that a more resilient design can significantly reduce the environmental, economic, and social impacts in the use phase of the structure. Their proposed framework could be used to compare alternative designs for informed decision making. According to the authors, their proposed framework, even though successfully applied to the case study, has the following limitations:

- Anticipated differences in reliability of data sources due to the use of several databases.
- Several assumptions made in predicting the extent of damage corresponding to each damage state.
- Exclusion of nonstructural components and contents of the buildings from the assessment.
- Aging-induced impact factors were not considered.

4.4.5 Dimethyl sulfoxide solvent recovery from hazardous wastewater

An ethylene-vinyl alcohol copolymer (EVOH) based adsorbent developed by S-Metalltech 98 Ltd., Szentendre, Hungary is used to remove arsenic (As(III) and As(V)) from water. The process of manufacturing the EVOH produces hazardous wastewater containing 20 wt % dimethyl sulfoxide (DMSO). The current manufacturing process (linear, open technology) involves incineration of the wastewater including the DMSO. To minimize environmental impact, a closed technology, which involves the recovery of DMSO using distillation and incineration of about 2% wastewater has been developed. Zajáros et al. (2018), applied LCSA on a functional unit of 1 m^3 adsorbent to compare the sustainability of two scenarios:

- 98% DMSO recovery (DMSO_R).
- 98% DMSO recovery and usage as renewable resources (DMSO_R + PV).

The LCA results for the two scenarios were compared with the original data for DMSO from databases. Their assessment was based on ISO 14040 standard using SimaPro 7.2 demo, GaBi 4 programs, and Ecoinvent database.

The results of the LCSA (Fig. 4.13), reveals that the DMSO_R + PV reduces environmental, social, and economic impacts. Specifically:

- the recovery and reuse (50% renewable energy) reduces the amount of water usage in manufacturing process and the resultant hazardous wastewater production by 27% and 98%, respectively; and
- recovery by distillation produces DMSO of at least 95% purity.

The authors noted that LCSA can be used to identify sustainability of future development.

4.4.6 Fly ash substitutions for cement in concrete structures

Wang et al. (2017) used an LCSA model to assess the sustainability of fly ash (FA) concrete (C50) structures with FA substitutions (0%, 20%, 30%, 40%, and 50%) for cement. The case study was applied on a 9×20 m prestressed concrete simply supported girder bridge. The system boundary and function unit were respectively "cradle to gate" and 1 m^3 FA concrete. Life cycle impact data for concrete (such as cement and aggregates) were obtained from

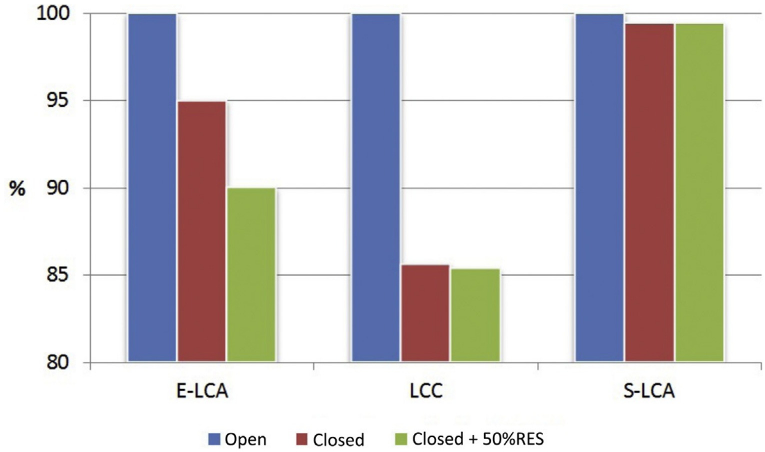


FIG. 4.13 LCSA of different technologies for recovery of DMSO (Zajáros et al., 2018).

European Life Cycle Database (ELCD). The authors used energy for transportation and preparation of concrete (petrol, oil, diesel, and electricity), data were derived from other studies (Yang et al., 2002; Yang, 2003). Environmental impact assessment method used was Eco-indicator 99 (EI99). In addition to the sustainability dimensions, the authors used Monte Carlo simulation approach to calculate reliability of individual members in the structure whiles the probabilistic network evaluation technique (PNET) was used to calculate the reliability of the whole structure.

From their case study, the best addition of FA for social impact is 40%. From the environmental dimension, the strength of concrete decreased with increasing amounts of FA. In terms of economic impact, the optimal substitution of FA ranges from 20% to 40%. The reliability of the upper, lower, and whole system fluctuated with FA substitutions, but the highest initial reliabilities were obtained for 30% FA substitutes. The reliability index ($\beta_T = 4.3449$) for the whole structure without maintenance for 30% FA replacement corresponded to a service life of a little over 50 years. When all the three sustainability dimensions were combined, 30% substitution of cement with FA significantly reduced environmental, economic, and social impacts (Fig. 4.14).

4.4.7 Electricity generation systems

LCSA has been applied to assess the sustainability of one or more electricity systems in many countries such as United Kingdom, Greece, Portugal, and the United Kingdom.

4.4.7.1 Electricity generations options in the United Kingdom

Stamford and Azapagic (2012) assessed electricity generation options for the United Kingdom using LCSA from "cradle-to-grave." The electricity options are coal (pulverized), gas

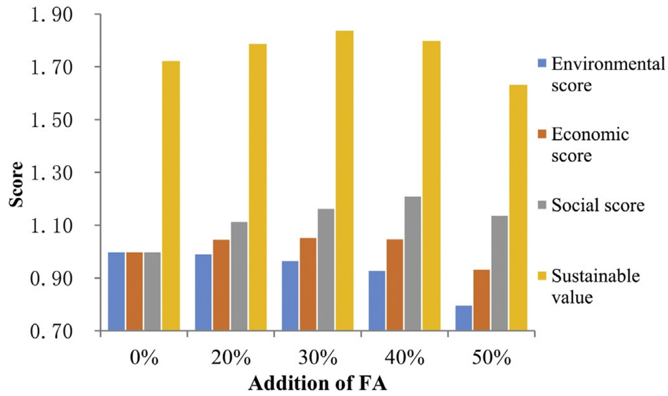


FIG. 4.14 Sustainability scores for different substitutes of cement for FA in concrete (Wang et al., 2017).

(CCGT), nuclear (pressurized water reactor), offshore wind, and photovoltaics. Forty-three sustainability indicators developed by Stamford and Azapagic (2011) from direct stakeholder (industry, government, academia, and NGOs) engagement and literature review were used. Environmental impacts (LCA) were calculated using GaBi v4.4 software and Ecoinvent v2.2 database. All data were revised to reflect UK conditions.

They report from their assessment, coal (pulverized) power, even though second cheapest option, had the worst environmental performance. CCGT was found to be the cheapest option but with the highest cost variability, high fossil fuel depletion, and lowest employment. Nuclear power performed better in eight environmental indicators but had the second lowest life cycle employment, highest health impact from radiation, and highest number of fatalities in a single incident. Wind, like nuclear power, was the best in terms of environmental impacts, second highest in employment provision, increased energy security, but worse in freshwater and terrestrial eco-toxicity as well as nonfossil resource depletion. Offshore PV performed badly in terms of environmental and economic dimensions, but provides the highest employment.

The authors, therefore, concluded that no single electricity option in United Kingdom is most sustainable and that there is the need for trade-offs and compromises.

4.4.7.2 Grid-connected photovoltaic systems in Northeast England

Similarly, Li et al. (2018) proposed and applied a comprehensive LCSA model on three types of grid-connected solar photovoltaic (monocrystalline silicon, s-Si; polycrystalline silicon, p-Si; and cadmium telluride, CdTe thin film) electricity generation cells in northeast England. In this case study, a 4 kWp residential roof-mounted grid-connected system is used with functional unit of per unit of electricity produced. Both top-down and bottom-up approaches were used to select 13 sustainability indicators through relevant stakeholder engagement and literature review. GaBi professional v6.1 15 software and Ecoinvent 3.1 integrated database were used for assessment of environmental impacts. Table 4.3 shows the sustainability rankings of the three solar PV systems.

TABLE 4.3 Sustainability ranking of solar PV systems in Northeast England (Li et al., 2018).

		Type of solar photovoltaic systems		
		Silicon		Thin film
Sustainability dimensions	Indicators	s-Si	p-Si	CdTe
Techno-economic	Availability factor	1	1	1
	Capacity factor	2	1	3
	Levelized cost	2	1	3
	Payback period	2	1	3
	Profitability	1	2	3
	Subtotal	8	6	13
Environmental	Circularity	1	1	2
	Energy payback period	1	1	2
	Global warming potential	1	1	2
	Acidification potential	2	2	1
	Eutrophication potential	1	1	2
	Ozone layer depletion potential	1	1	2
	Subtotal	7	7	11
Social	Bill reduction rate	1	2	3
	Employment provision	1	1	1
	Subtotal	2	3	4
Grand total		17	16	28

Polycrystalline silicon (p-Si) systems were found to be the most sustainable option followed by monocrystalline silicon (s-Si) systems. Cadmium telluride (CdTe) thin film systems were the worst performing in all three sustainability dimensions.

4.4.7.3 *Electricity generation systems in Greece*

In Greece, Roinioti and Koroneos (2019) applied LCSA methodology to assess and compare the sustainability of seven electricity generation systems (conventional lignite-fired power plants, combined-cycle gas turbine (CCGT) plants, large hydropower plants (reservoirs), wind power stations, photovoltaics, small hydropower plants, and biomass/biogas power plants). Natural gas and fossil fuel are both imported. CCGT is an indigenously produced fuel. The analysis was performed on a functional unit of 1 kWh of electricity. The system boundaries were from “cradle to gate” because operability indicators were not analyzed. The sustainability indicators used were six for LCA, three for LCC, and six for SLCA. LCA was conducted for the year 2015 (due to completeness of available data) using Gemis version 4.9.5 software and its database. Background LCA data and technical parameters (e.g.,

TABLE 4.4 Ranking of electricity technologies with different preferences in Greece (Roinioti and Koroneos, 2019).

		Priority given to:		
Electricity technology	**Equal weights**	**Environmental aspect**	**Economic aspect**	**Social aspect**
Lignite	7	7	6	6
CCGT	6	6	3	7
Large hydro	3	3	5	5
Small hydro	2	2	2	3
Wind	1	1	1	2
PV	4	4	7	1
Biomass/biogas	5	5	4	4

efficiency operation time, lifetime, and average plant capacity size) were adjusted to reflect the country's conditions. Data on the cost of electricity generation was obtained from industry studies and the rate of discount was assumed to be 10%. Integration of the sustainability pillars to support the evaluation and selection of alternatives was performed with multiattribute value theory (MAVT) using the multicriteria decision support software, Web-HIPRE V1.22. All the sustainability pillars were given equal weighting (0.33) because stakeholder preferences were not considered. The electricity generation technologies were ranked based on their sustainability score. The technology with the highest score was considered more sustainable. Similarly, different weightings (five times more important, 0.714) were assigned to each of the sustainability pillars. The ranking of the technologies with different preferences is shown in Table 4.4.

From the assessment, wind energy was found to be more sustainable followed by small hydropower plants for equal weights and when priority was given to the environmental and economic criteria. In terms of social aspects as preference, photovoltaics were more sustainable followed by wind and small hydropower. Fossil fuel options (CCGT and lignite plants) were the least preferred, even though CCGT ranked third when preference was given to the economic aspects. The authors recommended an increase of the share of energy from renewable sources (wind, hydropower, and photovoltaics) whilst reducing fossil fuel electricity options (CCGT and lignite plants).

4.4.7.4 Electricity generation systems in Portugal

Similarly, LCSA methodology was used by Kabayo et al. (2019) to assess and compare six main electricity generation systems (coal, natural gas, small hydro, large hydro, wind, and ground mounted photovoltaic (PV)) operating in Portugal from 2012 to 2016. It was assumed that all coal fuel was sourced from Columbia, while natural gas originated from Algeria (45% via pipeline) and Nigeria (55% by shipping). The choice of 5-year (2012–16) period for assessment was to account for the variation in generation and related environmental, economic, and social impacts from year to year. The assessment was performed from "cradle-to-grave" for a functional unit of 1 MWh of electricity generated. Five environmental issues comprising of 11 indicators, and 4 socioeconomic issues comprising of 5 indicators were assessed. In assessing the environmental impacts (LCA), the SimaPro 8.0 software was used. Background data were obtained from Ecoinvent v3.0 database. They collected foreground data from sources directly related to systems operating in Portugal as of 2016. It was assumed that the system characteristics and conditions were constant throughout the plant lifetime. Environmental impacts were assessed using ReCiPe impact assessment method implementing a hierarchist (where priority is given to the environmental dimension) midpoint approach. For toxicity-related indicators and freshwater scarcity footprint, Usetox 1.04 and AWARE methods were used, respectively. The overall sustainability performance was obtained by ranking each generation system based on un-weighted color gradient scale for each indicator. The results of the overall sustainability performance are shown in Fig. 4.15.

Their study revealed that renewable systems were more sustainable than fossil fuel-based systems. In particular, small hydro systems were found to be most sustainable whilst coal systems were the least sustainable.

Indicator	Unit/MWh	CO	NG	HL	HS	WD	PV
MD	kg Fe eq.	3.1	1.1	2.2	2.0	**18.6**	13.9
FD	kg oil eq.	**238.1**	154.8	1.3	0.9	4.4	13.4
GW	kg CO_2 eq.	**965**	444	14	4	16	50
OD	kg CFC-11 eq.	5E-06	**6E-05**	5E-07	3E-07	2E-06	9E-06
TA	kg SO_2 eq.	**2.62**	0.31	0.03	0.02	0.11	0.33
FWEut	kg PO_3^{4-}eq.	**4E-01**	2E-03	1E-03	1E-03	2E-02	3E-02
AqAc	kg SO_2 eq.	**3.14**	0.38	0.03	0.02	0.12	0.36
FWEc	CTUe	**0.50**	0.03	0.03	0.01	0.11	**0.54**
FWSF	world.m^3	1.5	12.1	**615.6**	0.6	1.3	23.9
HTcar	CTUh	**6E-09**	6E-09	5E-10	4E-10	2E-09	3E-09
HTnon	CTUh	1E-09	7E-11	4E-11	2E-11	2E-10	**2E-09**
DE	Pers.years	**8.7E-05**	1.0E-04	1.6E-04	4.7E-04	2.0E-04	9.9E-04
TE	Pers.years	6.4E-04	3.8E-04	**1.8E-04**	5.2E-04	2.8E-04	1.2E-03
DFF	Relative %	**100%**	67.1%	0.5%	0.4%	1.8%	5.6%
CF	%	77.4%	21.7%	21.9%	30.8%	29.0%	**20.7%**
LCOE	USD	87.5	**129.7**	113.2	92.5	63.6	76.9

← Most favourable Most unfavourable →

CO: Coal; NG: Natural Gas; HL: Large Hydro; HS: Small Hydro; WD: Wind; PV: Photovoltaic; CTU: Comparative Toxic Units (e: ecosystems; h: humans).

FIG. 4.15 Overall sustainability performance of different electricity generation systems in Portugal (Kabayo et al., 2019).

4.5 Outlook: Perspective and opportunities

As sustainability is widely recognized as the real challenge of our generation and life cycle thinking, in this sense, as the proper theoretical framework for sustainability application, its assessment must be regarded as a strategic decision-support element for planning at industrial, operational, and policy level (Ramos, 2019). For this reason, the identification of the forefront of both theory development and practical implications could represent the first step towards the LCSA of tomorrow, which, as underlined by Pope et al. (2014), should move from the *ex post* assessment framework to a preeminent role into *ex ante* toolbox for eco-design of products, processes, and systems.

Several challenges have been identified in literature (Zamagni, 2012; Guin'ee, 2016) and more are posed by everyday practice, such as a deeper integration of the three pillars into LCSA and harmonization among the existing models, the implementation of a multimethod approach to address uncertainties, and the broadening of impacts and scope of the LCSA, in terms of both temporal and spatial dimensions and dynamics. As conclusion for the present chapter, a brief excursus of solutions offered in literature to the abovementioned issues is provided in the following.

4.5.1 Integration and harmonization

Several authors (Zamagni, 2012; Guin'ee, 2016; Gloria et al., 2017; Kua, 2017) spread out the call for a deeper integration among different aspects of LCSA, namely environmental, social, and economic, with particular regard to their mutual relationship and reciprocal effects. As they are currently addressed separately, following the scheme proposed by Klöpffer (2003), in terms of both independent inventories and analyses, even when developed under the same premises, rules, and scopes, they are unable to deliver an overall assessment. Thus, the direct application of LCSA as sum of ELCA + SLCA + LCC is actually failing in providing a result going beyond the sum of impacts of its different constituents (Lee and Kirkpatrik, 2001; Zamagni, 2012). In a context of difficult data collection, where SLCA still results under development and it could be affected by higher uncertainties, compared to ELCA and LCC, the conceptual framework appears inadequate to fully depict the interrelationships and interdependencies of the three pillars' assessments (Zamagni et al., 2013).

In this sense, the effort towards standardization and harmonization of different existing tools would support the construction of a common pathway for researchers, in analogy with what is already proposed and accomplished for ELCA, supporting the identification and prioritization of common goals and methods. Zamagni (2012), following the same foundation stone of the approach, which calls directly to humankind and generational equity as yardstick for sustainability (Bruntland, 1987), attempted a holistic perspective in posing the human at the very center of LCSA, and proposing "well-being adjusted life years" as a unique LCSA indicator. Within the body of literature, new approaches emerged during the last years, such as life cycle sustainability unified analysis (LiCSUA, proposed by Kua (2017), incorporating key features of LCSA framework (Klöpffer and Renner, 2007), and the life cycle sustainability analysis framework proposed under CALCAS. Corona et al. (2017), on the other hand, accepted the three-pillar model of LCSA, applying the same structure as ELCA to SLCA, suggesting, at the same time, new classification and characterization models, following United Nations Environment Program/Society for Environmental Toxicology and Chemistry (UNEP/SETAC) guidelines on S-LCA. Nevertheless, this remains an open issue, which should be perceived as a starting point for every future development of LCSA.

4.5.2 Implementation of a multimethod approach

Following the call for integration of different aspects of LCSA and in order to fill the gap between modeling and reality, i.e., reducing the uncertainties inherent to the modeling activity, a multimethod approach is widely regarded as the main opportunity available (Halog and Manik, 2011; Gloria et al., 2017; Ren, 2018a,b). From the beginning of LCSA application, the intrinsic trans-disciplinary nature of the subject was recognized as requiring an integration of methods and models (Guinée et al., 2011), to address specific sustainability issues. This raised the issue of selection, sharing, and availability of these models and of the proper matching between models and sustainability questions (Gloria et al., 2017).

Within the framework of a growing circular economy, LCSA should be also able to adapt and include tools able to model industrial symbiosis, circular material flow analyses, and resource scarcity. Many analytical approaches have been recently explored as opportunities to improve the traditional LCSA. An overview is provided in Appendix B and some significant examples are reported here in the following.

Plevin (2016), for instance, as presented by Gloria et al. (2017), explored the opportunity offered by Global Change Assessment Model (an integrated assessment model) to improve the comprehensiveness and robustness of Climate-LCA of biofuels. In particular, following the approach set by consequential LCA, market interactions, cascade consequences at global scale, and evolution in socio-economic indicators, such as population and GDP, and technical knowledge are addressed. Wu et al. (2017), on the other hand, integrated an agent-based modeling approach to the life cycle inventory analysis in order to address temporal and spatial variation of indicators into the case study of green building development. Moreover, an agent-based modeling has been applied to address micro-level interactions and heterogeneity displayed at individual level (Gloria et al., 2017).

The same Kua (2017), abovementioned, proposes an integrated and unified analysis, considering "soft" indicators, such as vulnerability, resilience, and stakeholders' risk aversion through approaching "cross-links indicators, inter- and intradimensional consequences, rebound effects, and potential 'transitioning' of these indicators into a single framework" (Gloria et al., 2017: 1451). He et al. (2019), addressing the subject of sustainability assessment of products, set accuracy as the first space for improvement of LCSA and they proposed an indicators' analysis approach to mitigate the impact of uncertainties over the final results; the set of indicators belonging to five conceptual areas, namely energy, environment, resource, technology, and economy (Fig. 4.16).

Ren (2018a,b), still focused on uncertainties management, proposed a comprehensive life cycle sustainability prioritization framework for ranking the energy systems. As presented in Fig. 4.17, a multistage approach was applied. In particular, a first step involved a fuzzy two-stage logarithmic goal programming method, used to determine the weights of the criteria for

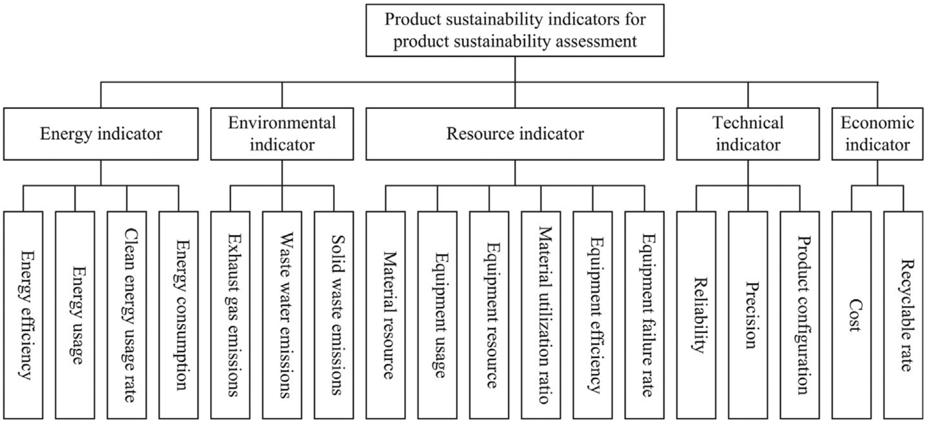
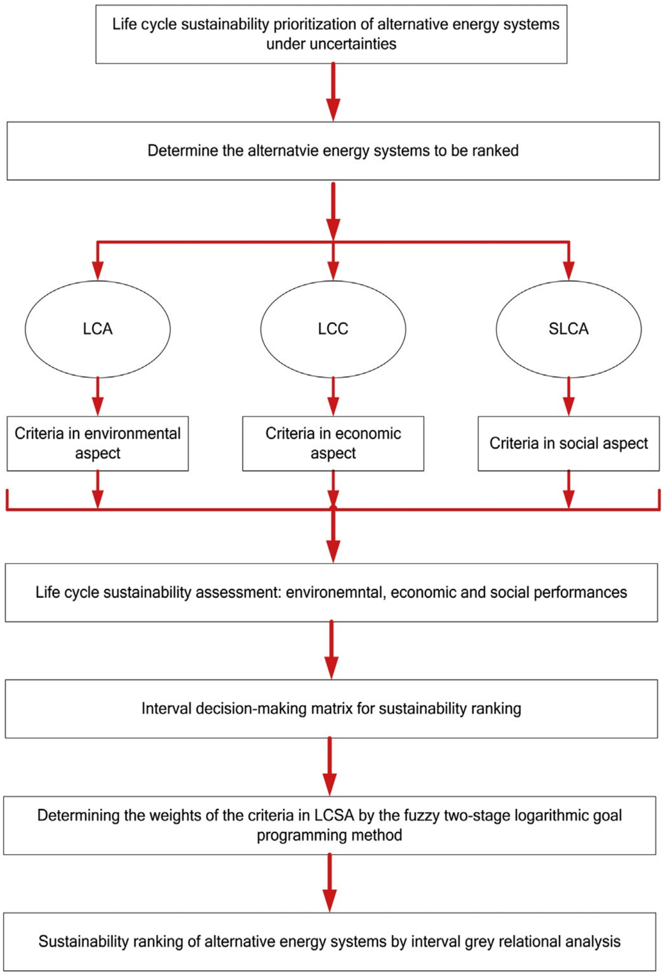


FIG. 4.16 Product sustainability indicators (He et al., 2019).

FIG. 4.17 Flowchart describing the application of life cycle sustainability assessment for prioritization (Ren et al., 2015; Ren, 2018a,b).



sustainability assessment. An interval gray relational analysis method was then applied for the prioritization activity. The results obtained from a system analysis of alternatives, giving rise to a prioritized, multicriteria assessment, is regarded as a robust and reliable decision-support tool, highlighting the vocation of LCSA to be applied at the design stage.

4.5.3 Broadening of impacts and scope

In a context of growing application and implications of the LCSA framework, the broadening of impacts and scope addressed by the tool (or set of tools) and how to harmonize a deeper and wider assessment are widely regarded as open challenges (Schaubroeck and Rugani, 2017). Guin'ee (2016), for instance, suggests that LCSA should not be limited to the product or organization's level, but should be used also for analysis encompassing entire systems and economies.

In addition to this, due to the extreme complexity of systems underpinned even to the simplest LCSA, dynamicity of models and results and their adaptability are key elements for the overall reliability of the assessment. As both natural and anthropic systems are evolutionary by definition, a static LCSA may only return an instant picture of something that is either changing before the eyes of the analyst or not mimetically representative of the phenomenon as a whole. For these reasons, a dual-dimensional trajectory must be followed, as suggested by Wu et al. (2017), following the development of the indicators on scales both temporal, i.e., on different interval of time, and spatial, i.e., in terms of spatially distributed simulation (Fig. 4.18).

Under these premises, how to define the new boundaries for the LCSA and how to account for direct and indirect interactions among the new level of analysis (e.g., technological, economic, and political) remain still open issues (Gloria et al., 2017).

4.5.4 Final remarks

Concluding this overview of the ongoing pathway of LCSA, two further elements for discussion are worth mentioning, related to the scale of application of LCSA, namely at global and immanent level, i.e., the overall context of sustainability, and on the local and contingent dimension, i.e., the diffuse application of LCSA.

With regard to the context of sustainability, the relationship between LCSA, as decision-support tool, and sustainable development goals (UN SDGs), as a widely recognized framework for decision-makers, has been evaluated by a few authors (Gloria et al., 2017; Ramos, 2019). As LCSA comprises also socio-economical assessment into an integrated tool, in fact, it actually responds directly to the call for a supportive and trans-disciplinary approach proposed by UN SDGs. Thus, promoting a pro-active attitude towards the planning of activities, operations, processes, programs and policies, based on quantitative and comprehensive assessment of impacts. As stated by Zamagni et al. (2013), LCSA, in this sense, as many predictive-modeling activities, is inherently permeated with self-denying prophecies ("e.g., in predicting undesired consequences, which will be combated before they have the chance to develop," Zamagni et al., 2013: 1637), which may actually represent a double edged weapon for the diffusion of LCSA at the decision-making level.

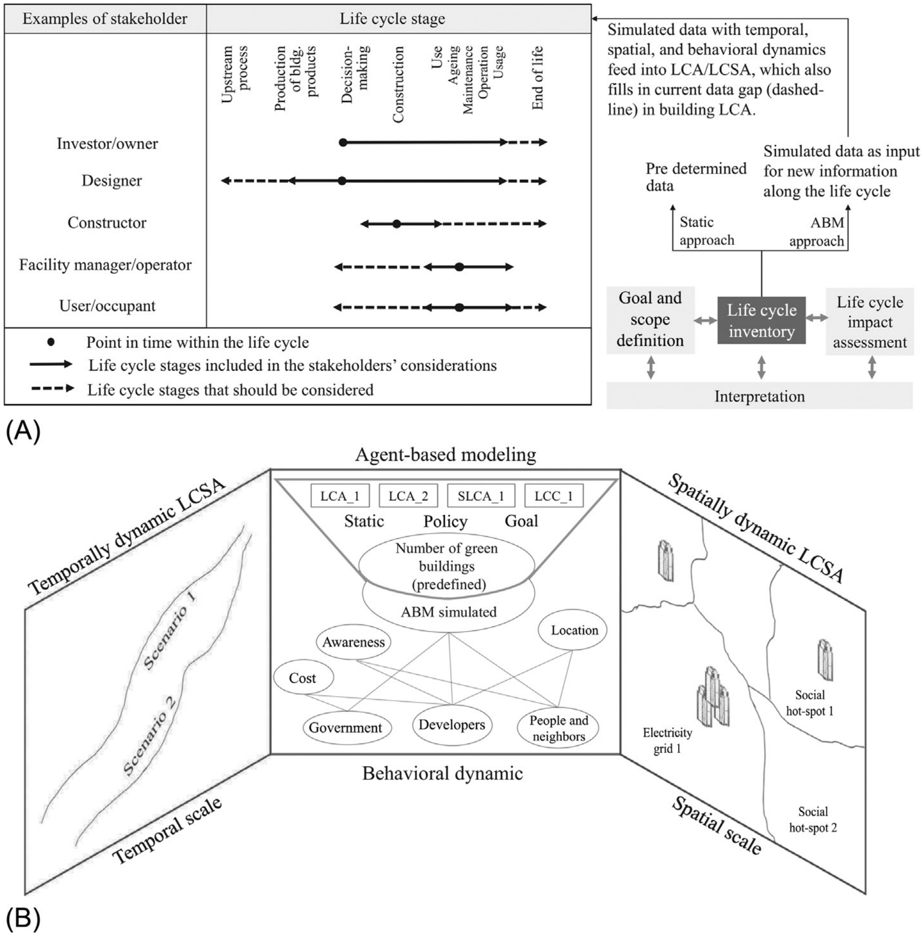


FIG. 4.18 Outline of the integration of an agent-based model into LCA framework for the case study of LCSA of buildings. In particular: (A) depicts two general LCI approaches towards data; (B) compares the static modeling with the agent-based modeling (ABM), demonstrating the temporal, spatial, and behavioral dynamics (Wu et al., 2017). *(A) Left part of the subfigure is adapted from ISO 21931-1:2010 Fig. B.2 (ISO 2010, 19).*

On the other hand, in order to promote a wider application of the LCSA framework and, consequently, a broader diffusion of sustainability-related considerations in the design of products and processes, a set of simplified tools would certainly be required. A set of tools possibly fitting with this call could include indicators and scenario analysis. As highlighted by Bell and Morse (2018), the use of indicators and indices, which is typically aimed at simplifying systems and conveying complex information to the public, could be curbed to this scope, but it would require a detailed analysis of the specificity of sectors and applications of LCSA and it would probably not respond to "one size fits for all." The same approach could be applied through scenario analysis, where processes are simplified and alternative solutions characterized at a level useful to improve the understanding of the decision-makers and speed-up the adaptation of the proposal to the development, for example, of a project, and yet complex enough as to model the process itself adequately, without losing the reliability of results proposed (Spangenberg, 2018).

References

Apperl, B., Pulido-Velazquez, M., Andreu Álvarez, J., Karjalainen, T.K., 2015. Contribution of the multi-attribute value theory to conflict resolution in groundwater management application to the Mancha Oriental groundwater system, Spain. Hydrol. Earth Syst. Sci. 19, 1325–1337.

Assies, J.A., 1992. Introduction paper to SETAC-Europe workshop on environmental life cycle analysis of products. In: Life-Cycle Assessment, Proceedings of a SETAC-Europe Workshop on Environmental Life Cycle Assessment of Products, December 2–3 1991, Leiden; SETAC-Europe: Brussels, Belgium.

Bachmann, T.M., 2013. Towards life cycle sustainability assessment: drawing on the NEEDS project's total cost and multi-criteria decision analysis ranking methods. Int. J. Life Cycle Assess. 18, 1698–1709.

Bell, S., Morse, S., 2018. Sustainability indicators past and present: what next? Sustainability 2018 (10), 1688. https://doi.org/10.3390/su10051688.

Boron, S., Murray, K.R., Thomson, G.B., 2017. Sustainability education: towards total sustainability management teaching. In: Leal, F.W., Brandli, L., Castro, P., Newman, J. (Eds.), Handbook of Theory and Practice of Sustainable Development in Higher Education. World Sustainability Series, Springer, Cham.

Bozhilova-Kisheva, K.P., Hu, M., Van Roekel, E., Olsen, S.I., 2012. An integrated life cycle inventory for demolition processes in the context of life cycle sustainability assessment. In: International Symposium on Life Cycle Assessment and Construction. IFSTTAR.

Bruntland, G.H., 1987. Our Common Future. Oxford University Press, Oxford.

Burton, H.V., Deierlein, G., Lallemant, D., Lin, T., 2016. Framework for incorporating probabilistic building performance in the assessment of community seismic resilience. J. Struct. Eng. 142, C4015007.

Charter, M., 2001. Integrated product policy (IPP) and eco-product development (EPD). In: Proceedings Second International Symposium on Environmentally Conscious Design and Inverse Manufacturing, Tokyo, Japan, pp. 672–677.

Coenen, L., Truffer, B., 2012. Places and spaces of sustainability transitions: geographical contributions to an emerging research and policy field. Eur. Plan. Stud. 20(3).

Corona, B., Bozhilova-Kisheva, K.P., Olsen, S., Miguel, G.S., 2017. Social life cycle assessment of a concentrated solar power plant in Spain: a methodological proposal. J. Ind. Ecol. 21 (6), 1566–1577. https://doi.org/10.1111/jiec.12541.

De Schryver, A.M., Humbert, S., Huijbregts, M.A., 2013. The influence of value choices in life cycle impact assessment of stressors causing human health damage. Int. J. Life Cycle Assess. 18, 698–706.

Deierlein, G., Krawinkler, H., Cornell, C., 2003. A framework for performance-based earthquake engineering. In: Pacific Conference on Earthquake Engineering, 2003 Christchurch, New Zealand.

Del Borghi, A., 2013. LCA and communication: environmental product declaration. Int. J. Life Cycle Assess. 18 (2), 293–295. https://link.springer.com/article/10.1007/s11367-012-0513-9.

Dreyer, L.C., Niemann, A.L., Hauschild, M.Z., 2003. Comparison of three different LCIA methods: EDIP97, CML2001 and Eco-indicator 99. Int. J. Life Cycle Assess. 8, 191–200.

Durairaj, S.K., Ong, S.K., Nee, A.Y.C., Tan, R.B.H., 2002. Evaluation of life cycle cost analysis methodologies. Corp. Environ. Strat. 9(1). http://scholarbank.nus.edu.sg/handle/1063568231.

Ecoinvent, 2003. https://www.ecoinvent.org/.

Ekener, E., Hansson, J., Larsson, A., Peck, P., 2018. Developing life cycle sustainability assessment methodology by applying values-based sustainability weighting-tested on biomass based and fossil transportation fuels. J. Clean. Prod. 181, 337–351.

Ekener-Petersen, E., Höglund, J., Finnveden, G., 2014. Screening potential social impacts of fossil fuels and biofuels for vehicles. Energy Policy 73, 416–426.

Elkinton, J., 1998. Cannibals with forks: the triple bottom line of 21st century business. Gabriola Island: New Society Publishers, Autumn, Environ. Qual. Manag, 37–51. https://doi.org/10.1002/tqem.3310080106.

European Commission, 2013. Organisation Environmental Footprint Guide. European Commission-Joint Research Centre-Institute for Environment and Sustainability. http://eur-lex.europa.eu/JOHtml. do?uri=OJ:L:2013:124:SOM:EN:HTML.

Eyerer, P., 1996. Ganzheitliche Bilanzierung—Werkzeug zum Planen und Wirtschaften in Kreisläufen. Springer, Heidelberg, Switzerland.

Ferretti, V., Bottero, M., Mondini, G., 2014. Decision making and cultural heritage: an application of the multi-attribute value theory for the reuse of historical buildings. J. Cult. Herit. 15, 644–655.

Finkbeiner, M., König, P., 2013. Carbon footprint and life cycle assessment of organizations. J. Environ. Acc. Manage. 1, 55–63.

Finkbeiner, M., Schau, E.M., Lehmann, A., Traverso, M., 2010. Towards Life Cycle Sustainability Assessment, pp. 3309–3322. https://doi.org/10.3390/su2103309.

Gbededo, M.A., Liyanage, K., Garza-Reyes, J.A., 2018. Towards a Life Cycle Sustainability Analysis: a systematic review of approaches to sustainable manufacturing. J. Clean. Prod. 184, 1002–1015.

Gencturk, B., Hossain, K., Lahourpour, S., 2016. Life cycle sustainability assessment of RC buildings in seismic regions. Eng. Struct. 110, 347–362.

Gloria, T., Guinee, J., Wei Kua, H., Singh, B., Lifset, R., 2017. Charting the future of life cycle sustainability assessment. J. Ind. Ecol. 21 (6), 1449–1453.

Grahl, B., Schmincke, E., 2007. The part of LCA in ISO type III environmental declarations. Int. J. Life Cycle Assess 12 (1), 38–45.

Grießhammer, R., Manhart, A., Dreyer, L.C., FlysjÖ, A., 2005. Developing a Set of Indicators for a Social LCA. November.

Grießhammer, R., Benoît, C., Dreyer, L.C., Flysjö, A., Manhart, A., Mazijn, B., Méthot, A.L., Wedema, B., 2006. Feasibility Study: Integration of Social Aspects into LCA.

Guin'ee, J.B., 2016. Life cycle sustainability assessment: what is it and what are its challenges? In: Clift, R., Druckman, A. (Eds.), Taking Stock of Industrial Ecology. Springer Open, London, pp. 45–68.

Guinée, J.B., Heijungs, R., Huppes, G., Zamagni, A., Masoni, P., Buonimici, R., Ekvall, T., Rydberg, T., 2011. Life cycle assessment: past, present and future. Environ. Sci. Technol. 45 (1), 90–96.

Günay, S., Mosalam, K.M., 2013. PEER performance-based earthquake engineering methodology, revisited. J. Earthq. Eng. 17, 829–858.

Hacatoglu, K., Dincer, I., Rosen, M., 2015. A new model to assess the environmental impact and sustainability of energy systems. J. Clean. Prod. 103, 211–218.

Halog, A., Manik, Y., 2011. Advancing integrated systems modeling framework for life cycle sustainability assessment. Sustainability 3, 469–499. https://doi.org/10.3390/su3020469.

Hardi, P., Semple, P., 2000. The dashboard of sustainability—from a metaphor to an operational set of indices. In: Proceedings of the Fifth International Conference on Social Science Methodology, Cologne, Germany, 3–6 October.

He, B., Luo, T., Huang, S., 2019. Product sustainability assessment for product life cycle. J. Clean. Prod. 206, 238–250.

Hofstetter, P., Braunschweig, A., Mettier, T., Müller-Wenk, R., Tietje, O., 1999. The mixing triangle:correlation and graphical decision support for LCA-based comparisons. J. Ind. Ecol. 3, 97–115.

Hunkeler, D., KlÖpffer, W., Pesonen, H., Ciroth, A., Swarr, T., Pagan, B., Itsubo, N., 2008. Life cycle costing. In: SETAC World Congress, Sydney, August 6.

ILCD Handbook, 2010. General Guide for Life Cycle Assessment—Provisions and Action Steps. European Commission—Joint Research Centre.

ILCD Handbook, 2012. Towards More Sustainable Production and Consumption for a Resource-Efficient-Europe. Joint Research Centre.

Impact World+, 2016. http://www.impactworldplus.org/en/publications.php.

ISO, 2006b. ISO 14044: Environmental Management—Life Cycle Assessment—Requirements and Guidelines. International Organization for Standardization, Geneva.

ISO, 2012. ISO 14045—Environmental Management—Eco- Efficiency Assessment of Product Systems—Principles, Requirements and Guidelines. International Organization for Standardization, Geneve.

ISO, 2014. ISO/TS 14072: Environmental Management—Life Cycle Assessment—Requirements and Guidelines for Organizational Life Cycle Assessment. International Organization for Standardization, Geneve.

Kabayo, J., Marques, P., Garcia, R., Freire, F., 2019. Life-cycle sustainability assessment of key electricity generation systems in Portugal. Energy 176, 131–142.

Klöepffer, W., 2006. The Role of SETAC in the Development of LCA. Int. J. Life Cycle Assess. 11 (Special issue 1), 116–122.
Klöepffer, W., 2008. Life cycle sustainability assessment of products (with Comments by Helias A. Udo de Haes, p. 95). Int. J. Life Cycle Assess. 13 (2), 89–95.
Klöpffer, W., 2003. Life-cycle based methods for sustainable product development. Int. J. Life Cycle Assess. 8 (3), 157–159.
Klöpffer, W., Renner, I., 2007. Lebenszyklusbasierte Nach-haltigkeitsbewertung von Produkten. TaTuP Zeitschrift des ITASzur Technikfolgenabsch ätzung 16 (3), 32–38.
Kua, H.W., 2017. On life cycle sustainability unified analysis. J. Ind. Ecol. https://doi.org/10.1111/jiec.12665.
Lange, D., Devaney, S., Usmani, A., 2014. An application of the PEER performance based earthquake engineering framework to structures in fire. Eng. Struct. 66, 100–115.
LC-Impact, 2016. EU, https://lc-impact.eu/doc/LC-Impact_report_SEPT2016_20160927.pdf.
Lee, N., Kirkpatrik, C., 2001. Methodologies for sustainability impact assessments of proposals for new trade agreements. J. Environ. Assess Policy Manage. 3 (1), 395–412.
Li, T., Roskilly, A.P., Wang, Y., 2018. Life cycle sustainability assessment of grid-connected photovoltaic power generation: a case study of Northeast England. Appl. Energy 227, 465–479.
Lu, B., Li, B., Wang, L., Yang, J., Liu, J., Wang, X.V., 2014. Reusability based on life cycle sustainability assessment: case study on WEEE. Procedia CIRP 15, 473–478.
Luo, L., Van Der Voet, E., Huppes, G., 2009. Life cycle assessment and life cycle costing of bioethanol from sugarcane in Brazil. Renew. Sustain. Energy Rev. 13, 1613–1619.
Maranghi, S., Parisi, M.L., Basosi, R. 2016. Potentialities of LCA for Urban Systems Sustainability Assessment X Convegno dell'Associazione Rete Italiana LCA 2016 Life Cycle Thinking, sostenibilità ed. economia circolare Ravenna, June 23–24.
Moehle, J., Deierlein, G.G., 2004. A framework methodology for performance-based earthquake engineering. In: 13th World Conference on Earthquake Engineering, August 1–6, Vancouver, B.C., Canada.
Neugebauer, S., Martinez-Blanco, J., Scheumann, R., Finkbeiner, M., 2015. Enhancing the practical implementation of life cycle sustainability assessment e proposal of a Tiered approach. J. Clean. Prod. 102, 165–176.
Pimentel, D., Patzek, T.W., 2005. Ethanol production using corn, switchgrass, and wood; biodiesel production using soybean and sunflower. Nat. Resour. Res. 14, 65–76.
Plevin, R., 2016. Assessing the climate effects of biofuels using integrated assessment models: part I, methodological considerations. J. Indust. Ecol. https://doi.org/10.1111/jiec.12507.
Porter, K.A., 2003. An overview of PEER's performance-based earthquake engineering methodology. In: Proceedings of Ninth International Conference on Applications of Statistics and Probability in Civil Engineering (ICASP9) July 6–9, San Francisco.
Princigallo, A., Visini, D., Bonoli, A., Olsen, S.I., Niero, M., 2016. Comparative environmental and economic assessment of production, use and recycling of aluminium cans: Bologna vs Copenhagen. In: ATTI X Convegno dell'Associazione Rete Italiana LCA. 2016, pp. 352–359.
Pushkar, S., 2013. Using eco-indicator 99 to evaluate building technologies under life cycle assessment uncertainties. J. Architect. Eng. 20. 04013010.
Rahimi, M., Weidner, M., 2004. Decision analysis utilizing data from multiple life-cycle impact assessment methods: part II: model development. J. Indust. Ecol. 8, 119–141.
Ramos, T.B., 2019. Sustainability assessment: exploring the frontiers and paradigms of indicator approaches. Sustainability 11, 824. https://doi.org/10.3390/su11030824.
Remmen, A., Jensen, A.A., Frydendal, J., 2007. The triple bottom line—the business case of sustainability. In: Life Cycle Management. A Business Guide to Sustainability, pp. 10–11. UNEP/SETAC Life Cycle Initiative. ISBN 978-92-807-2772-2. UNEP 2007.
Ren, J., 2018a. Life cycle aggregated sustainability index for the prioritization of industrial systems under data uncertainties. Comput. Chem. Eng. 113, 253–263.
Ren, J., 2018b. Multi-criteria decision making for the prioritization of energy systems under uncertainties after life cycle sustainability assessment. Sustain. Prod. Const. 16, 45–57.
Ren, J., Manzardo, A., Mazzi, A., Zuliano, F., Scipioni, A., 2015. Prioritization of bioethanol production pathways in China based on life cycle sustainability assessment and multicriteria decision-making. Int. J. Life Cycle Asses. 20 (6), 842–853.
Roinioti, A., Koroneos, C., 2019. Integrated life cycle sustainability assessment of the Greek interconnected electricity system. Sustain. Energy Technol. Assess. 32, 29–46.

Schaubroeck, T., Rugani, B., 2017. A revision of what life cycle sustainability assessment should entail: towards modeling the net impact on human well-being. J. Indust. Ecol. https://doi.org/10.1111/jiec.12653.

Selmes, D.G., Boron, S., Murray, K., 1997. Industry, Life Cycle Assessment and Sustainability, pp. 153–156. Paper presented at IChemE Jubilee research event.

Society of Environmental Toxicology and Chemistry (SETAC), 1993. Guidelines for Life – cycle Assessment: A "Code of Practice". In: SETAC workshop in Sesimbra, Portugal 31 March – 3 April, Brussels: SETAC.

Spangenberg, J.H., 2018. Behind the scenarios: world view, ideologies, philosophies. an analysis of hidden determinants and acceptance obstacles illustrated by the Alarm scenarios. Sustainability 10, 2556. https://doi.org/10.3390/su10072556.

Stamford, L., Azapagic, A., 2011. Sustainability indicators for the assessment of nuclear power. Energy 36, 6037–6057.

Stamford, L., Azapagic, A., 2012. Life cycle sustainability assessment of electricity options for the UK. Int. J. Energy Res. 36, 1263–1290.

Stefanopoulos, K., Yang, H., Gemitzi, A., Tsagarakis, K.P., 2014. Application of the multi-attribute value theory for engaging stakeholders in groundwater protection in the Vosvozis catchment in Greece. Sci. Total Environ. 470–471, 26–33.

Swarr, T., Hunkeler, D., KlÖpffer, W., Pesonen, H.-L., Ciroth, A., Brent, A.C., Pagan, R., 2011. Environmental Life Cycle Costing: A Code of Practice. SETAC Press, Pensacola, FL.

Traverso, M., Finkbeiner, M., 2009. Life cycle sustainability dashboard. In: Proceedings of the 4th International Conference on Life Cycle Management, Cape Town, South Africa, 6–9 September.

Tsurukawa, N., Prakash, S., Manhart, A., 2011. Social Impacts of Artisanal Cobalt Mining in Katanga. Democratic Republic of Congo. Öko-Institut e.V, Freiburg, Germany.

UN Agenda 2030, 2015. https://www.un.org/sustainabledevelopment/development-agenda/.

UNEP, 2011. Global Guidance Principles For Life Cycle Assessment Databases. A Basis for Greener Processes and Products. United Nations Environment Programme and Society for Environmental Toxicology and Chemistry, Paris, France.

UNEP, 2015. Guidance on Organizational Life Cycle Assessment. Life-Cycle Initiative. United Nations Environment Programme and Society for Environmental Toxicology and Chemistry, Paris, France.

UNEP/SETAC, 2009. Guidelines for social life cycle assessment of products. United Nations Environment Programme, Paris.

UNEP/SETAC, 2011. Towards a Life Cycle Sustainability Assessment: Making Informed Choices on Products. UNEP/SETAC Life Cycle Initiative, p. 2011.

United Nations General Assembly, 1987. Report of the World Commission on Environment and Development: Document A/RES/42/187. In: 96th Plenary Meeting, 1–374.

Wang, J., Wang, Y., Suna, Y., Tingley, D.D., Zhang, Y., 2017. Life cycle sustainability assessment of fly ash concrete structures. Renew. Sustain. Energy Rev. 80, 1162–1174.

WRI and WBCSD, 2011. Corporate Value Chain (Scope 3) Accounting and Reporting Standard—Supplement to the GHG Protocol Corporate Accounting and Reporting Standard. World Resources Institute and World Business Council for Sustainable Development.

Wu, S., Li, X., Apul, D., Breeze, V., Tang, Y., Fan, Y., Chen, J., 2017. Agent-based modeling of temporal and spatial dynamics in life cycle sustainability assessment. J. Indust. Ecol. https://doi.org/10.1111/jiec.12666.

Yang, X.M., 2003. Environmental impact assessment theory and quantitative methods of architectural engineering during planning and design stage. Tsinghua University, Beijing.

Yang, T.Y., 2013. Assessing seismic risks for new and existing buildings using performance-based earthquake engineering (PBEE) methodology. In: Tesfamariam, S., Goda, K. (Eds.), Woodhead Publishing Series in Civil and Structural Engineering: Handbook of Seismic Risk Analysis and Management of Civil Infrastructure Systems. Woodhead Publishing.

Yang, J.X., Xu, C., Wang, R.S., 2002. Life cycle assessment and its application on products. China Meteorological Press, Beijing.

ZajÁros, A., Szita, K., Matolcsy, K., HorvÁth, D., 2018. Life cycle sustainability assessment of DMSO solvent recovery from hazardous waste water. Periodica Polytechnica Chem. Eng. 62, 305–309.

Zamagni, A., 2012. Life cycle sustainability assessment. Int. J. Life Cycle Assess. 17, 373–376. https://doi.org/10.1007/s11367-012-0389-8.

Zamagni, A., Peesonen, H.-N., Swarr, T., 2013. From LCA to life cycle sustainability assessment: concept, practice and future directions. Int. J. Life Cycle Assess. 18, 1637–1641. https://doi.org/10.1007/s11367-013-0648-3.

Further reading

Anand, A., Wani, M.F., 2010. Product life-cycle modeling and evaluation at the conceptual design stage: a digraph and matrix approach. ASME J. Mech. Des. 132 (9). 091010.

EDIP, 2003. http://www.gabi-software.com/support/gabi/gabi-lcia-documentation/edip-2003/.

EPD® System, 2014. https://www.environdec.com.

EU, JRC, 2012. Organisation Environmental Footprint (OEF) Guide, Ispra, Italy.

Finkbeiner, M., Saur, K., 1999. Ganzheitliche Bewertung in der Praxis. In Ökologische Bewertung von Produkten, Betrieben und Branchen; Symposium Bundesministerium für Umwelt. Jugend und Familie Österreich, Vienna, Austria.

Ghadimi, P., Azadnia, A.H., Yusof, N.M., Saman, M.Z.M., 2012. A weighted fuzzy approach for product sustainability assessment: a case study in automotive industry. J. Clean. Prod. 33, 10–21.

He, B., Huang, S., Wang, J., 2015. Product low-carbon design using dynamic programming algorithm. Int. J. Precis. Eng. Manuf. G.T. 2 (1), 37–42.

He, C., Zhang, Q., Ren, J., Li, Z., 2017. Combined cooling heating and power systems: sustainability assessment under uncertainties. Energy 139, 755–766.

Hemdi, A.R., Mat Saman, M.Z., Sharif, S., 2013. Sustainability evaluation using fuzzy inference methods. Int. J. Sustain. Energy 32 (3), 169–185.

ISO, 2006a. IS0 14040, First edition, 1997-06-15. Life Cycle Assessment. International Organization for standardization, Geneva.

ISO, 2010. ISO 14025—Environmental Labels and Declarations. International Organization for Standardization, Geneve.

ISO, 2013. ISO/TR 14069:2013—Greenhouse Gases—Quantification and Reporting of Greenhouse Gas Emissions for Organizations. International Organization for Standardization, Geneve.

Maxim, A., 2014. Sustainability assessment of electricity generation technologies using weighted multi-criteria decision analysis. Energy Policy 65, 284–297.

Pope, J., Annandale, D., Morrison-Saunders, A., 2004. Conceptualizing sustainability assessment. Environ. Impact Assess. Rev. 24, 595–616.

ReCiPE, 2008. A Life Cycle Impact Assessment Method Which Comprises Harmonized Category Indicators at the Midpoint and the Endpoint Level. Lieden University Press. www.leidenuniv.nl/cml/ssp/publications/recipe_characterisation.pdf.

Ren, J., Liang, H., 2017a. Multi-criteria group decision-making based sustainability measurement of wastewater treatment processes. Environ. Impact Assess. Rev. 65, 91–99.

Ren, J., Liang, H., 2017b. Measuring the sustainability of marine fuels: a fuzzy group multi-criteria decision making approach. Transp. Res. D 54, 12–29.

Ren, J., LÜtzen, M., 2017. Selection of sustainable alternative energy source for shipping: multi-criteria decision making under incomplete information. Renew. Sustain. Energy Rev. 74, 1003–1019.

Ren, J., Toniolo, S., 2018. Life cycle sustainability decision-support framework for ranking of hydrogen production pathways under uncertainties: an interval multi-criteria decision making approach. J. Clean. Prod. 175, 222–236.

Rosenbaum, R.K., 2014. Selection of impact categories, category indicators and characterization models. In: Curran, M.A. (Ed.), Goal and Scope Definition in Life Cycle Assessment, LCA.

Santoyo-Castelazo, E., Azapagic, A., 2014. Sustainability assessment of energy systems: integrating environmental, economic and social aspects. J. Clean. Prod. 80, 119–138.

Škobalj, P., Kijevčanin, M., Afgan, N., Jovanović, M., Turanjanin, V., Vučićević, B., 2017. Multi-criteria sustainability analysis of thermal power plant Kolubara-A Unit 2. Energy 125, 837–847.

UN Environment, 2016. The life Cycle Initiative. https://www.lifecycleinitiative.org/about/about-lci/.

CHAPTER

5

Development and applicability of life cycle impact assessment methodologies

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5.1 Introduction

Environmental pollution has an increasing effect on our daily life; as a result, people focus more and more on methods to assess environmental protection properties of industry processes or industrial production. Over the years, life cycle assessment (LCA) has drawn a lot of attention from many experts and scholars, by which the influence of industry processes could be quantified clearly. Nowadays, the LCA method, along with other life cycle ideas, is used to evaluate the environmental protection property of an industrial product, a craft process, or an activity (I.E. Agency, 2018; Intergovernmental Panel On Climate Change, 2014).

In the late 1960s, the life cycle assessment (LCA) concept emerged in the United States. In 1969, the Midwest Institute of the US tracked the processing procedure of bottles of Coca-Cola, including glass bottles and plastic bottles, from resource to final disposal. The resource and environmental profile analysis (REPA) method was used and laid the foundation for LCA (Hunt et al., 1996). In the late 1980s, with the increasingly serious regional and global environmental problems, global environmental awareness had been increasing gradually. The public started to focus on the results of LCA. A number of works of LCA promoted the rapid development of the LCA theory. In 1990, the first international seminar about LCA was held by the Society of Environmental Toxicology and Chemistry (SETAC), in which the concept of life cycle assessment was brought forward for the first time. In 1993, *Life Cycle*

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Assessment outline—a practical guide was published, marking the formal start of research into LCA methodology.

There are four correlative states making up the LCA, which are: (1) goal and scope definition; (2) inventory analysis; (3) impact assessment; and (4) interpretation, as shown in Fig. 5.1 (ISO, 2006a). The purposes of LCA can be: (1) comparison of alternative products, processes, or services; (2) comparison of alternative life cycles for a certain product or service; or (3) identification of parts of the life cycle where the greatest improvements can be made (Roy et al., 2009). In 1997, the International Standardization Organization (ISO) enacted *Environmental management—life cycle assessment—principles and framework* (ISO, 2006a) on the basis of SETAC and incorporated LCA into this standard series, putting forward the basic principles and frameworks of LCA by the international standard form (ISO, 2006a; Czyrnek-Delêtre et al., 2017).

The LCA is an assessment tool used to evaluate life cycle impact and resource utilization of a product, a process, or an activity (Khang et al., 2017). It has been used in many domains since its birth, such as energy, chemistry, food, agriculture, building materials, and so on (Lundie and Peters, 2005; Thomassen et al., 2008; Renó et al., 2011; Nemecek et al., 2011; Restianti and Gheewala, 2012; Valderrama et al., 2012; Garrett and Ronde, 2013). Software came into being to improve the efficiency of an LCA, because of its strong systematic nature, wide-ranging aspects, and a great deal of work. At present, the most popular pieces of LCA software worldwide are SimaPro and Gabi. SimaPro was invented by PRé Consultants in Netherlands and Gabi was invented by PE International in Germany (PRé-Consultants, 2014; Pe-International, 2014; Van Genderen et al., 2016). Different forms of LCA software integrate a large number of universal databases and environmental impact assessment models. The complicated process of modeling LCA and environmental impact analysis has been simplified, which can make LCA practitioners concentrate on researching core data and improve their work efficiency.

Sustainability science is a solution-oriented discipline, whose core scientific question is how to evaluate and improve sustainability reasonably (Robert et al., 2005; Mihelcic et al., 2003). Life cycle thought can support sustainability assessment. As there are many environmental policies in Europe, sustainable consumption and production implementation plans and European resource efficiency plans are supported by life cycle thought (Sala et al., 2013). Andersson et al. (1998) tested the feasibility of incorporating the sustainable principle into each stage of an LCA, which was the first attempt to apply LCA to sustainability

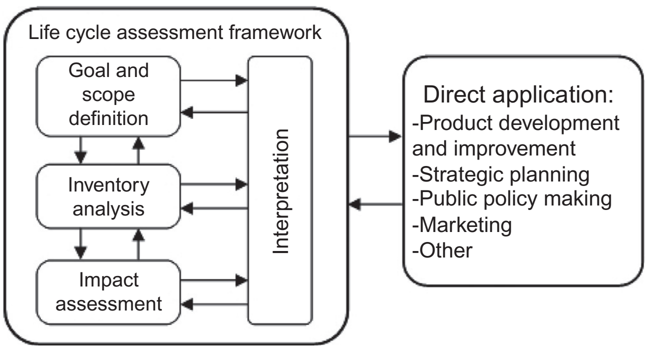


FIG. 5.1 Stages of an LCA (Czyrnek-Delêtre et al., 2017).

assessment. Upham (2000) applied sustainable principles to impact analysis of an LCA, promoting the development of a more general theory. Hunkeler and Rebitzer (2005) pointed out in *The Future of Life Cycle Assessment*, published in *The International Journal of Life Cycle Assessment*, that the perspective of an LCA should be extended to economy and society. Klöpffer et al. (2008) (Weidema, 2006) put forward the technical framework of the LCSA, including environmental, economic, and social aspects, stating that the framework of the LCSA should contain the life cycle assessment (LCA), the life cycle cost (LCC) and the social life cycle assessment (S-LCA), which can be expressed as:

$$\mathrm{LCSA} = \mathrm{LCA} + \mathrm{LCC} + \mathrm{SLCA}$$

The technical framework above makes LCA theory grow from focusing only on energy and environment analysis to more comprehensive sustainable assessment, including not only economy, but also social impact aspects. At present, environmental LCA, LCC, and S-LCA constitute the basis of LCSA (Ciroth and Franze, 2011), but they have different maturity. The environmental LCA, traditional ISO LCA, has improved gradually, becoming almost mature. The ecologically based LCA, Eco-LCA, mainly considers the impact of ecosystem products and services (such as water, mineral substance, carbon sequestration, etc.) on economic activity. The analyses on material flow and energy flow are common methods for LCA to analyze matter, materials, and energy (Zhang et al., 2010). The economic LCA was introduced by the concept of the life cycle cost theory and was self-fulfilling. For example, Desmond (2002) analyzed the life cycle cost of cars by using LCC method. The LCC method takes currency flows, like matter and energy flows, into consideration and structures cost according to life cycle stages and stakeholder, providing key economic indicators of evaluation system (Swarr et al., 2011).

Compared to LCC, S-LCA is still in its infancy. van Schooten et al. (2003) (Becker and Vanclay, 2003) proposed that evaluation would be impacted by society. The United Nations Environment Program (UNEP) suggested integrating the social criterion into traditional LCA method. Labuschagne et al. (Brent and Labuschagne, 2006) showed a sustainability method to evaluate projects and technologies used in processing industry, called social impact indicators. Grießhammer et al. (2006) first issued feasibility analysis of S-LCA. Geibler et al. (2006) (Brent and Labuschagne, 2006) advised that S-LCA should consist of eight indicators: health and safety, work conditions and quality, employment impact, education and training, knowledge management, innovation potential, customer acceptance and social benefits, and social dialogue. Norris (2006) showed life cycle attributes assessment (LCAA), the main idea of which involves putting a social responsibility certification system into an environmental life cycle assessment inventory and using an environmental inventory structure to evaluate social factor. Andrews et al. (2009) made social life cycle impact assessment (LCIA) of products of Canadian greenhouse tomato, using LCAA method, which showed that the social life cycle impact assessment can help greenhouse tomato production enterprises to fulfil their social responsibility. Benoît et al. (2010) issued an S-LCA guide and pointed out that social impact assessment should make analysis from five major stakeholders: workers/employees, community, society (at the national and global levels), consumers, and value chain actors. The S-LCA guide posed questions in life cycle social impact assessment and gave solutions of these questions at the same time, becoming the main reference for evaluation of social impacts. However, additional studies are still needed in methodological and practical aspects.

Zamagni et al. (2013) thought that, compared to E-LCA, S-LCA faces two difficulties, in data and indicators, respectively. Furthermore, the technical framework of LCSA still needs to be further understood and applied in three respects: environment, economy, and society.

5.2 The environmental assessment—LCA

The LCA is an analysis tool used to compute and evaluate the environmental impact and resource utilization of a product, a process, or an activity in the whole life cycle, including raw material mining, raw material transportation, products production, products transportation, products use, products maintenance, recycling, and final treatment (Rebitzer et al., 2004; Granovskii et al., 2006; de Haes et al., 1999, 2002). The research of an LCA focuses on the whole life cycle of a product and other objects to evaluate the environment and resource aspects, so its level is expressed from cradle to grave (Cabeza et al., 2014; Rafaschieri et al., 1999; Margaret et al., 1996; Pehnt, 2006).

5.2.1 Technical framework

5.2.1.1 Goal and scope definition

This phase is the first step, and is perhaps the most significant link of an LCA. In this phase, the expected product of the study, system boundaries, functional unit (FU), and assumptions are defined (Yue et al., 2013; Guinée, 2002). The system boundary is often expressed by the system balance diagram, and also contains processes supporting the life cycle of the product, process, or activity. The functional unit (FU) standardizes the inventory data and its definition depends on the affect type of environment and study itself. The FU is the basis of most products during the study. LCA is a continuous adjustment process. The breadth and depth of investigation depends on its target. The range of study also depends on the object of study and the potential application fields of study result (Curran, 1996). With the increasing number of information, the understanding of the system might be changed. The original target and the range of the research should also be adjusted.

5.2.1.2 Inventory analysis

In this phase, the data of resource, energy consumption, and contaminant discharged into the environment, in each stage of an LCA, have been collected and processed. Therefore, this phase takes the most time of the whole LCA, when data collection may be more time consuming (Hendrickson et al., 2010; Crawford, 2008). The time spent in this section depends on the quality of the data collected. If a good database is available, as well as customers and suppliers willing to offer help, the task of inventory analysis should be easier.

The inventory of an LCA is a summary of all inputs and outputs related to the system, based on the functional input and output data sheets. It is also the basis of the third phase impact assessment (ISO, 2006b). The correctness of the results depends directly on the quality of the inventory. The work of inventory analysis is an iterative process, throughout each stage of an LCA. With the process of collecting and disposal of data, the understanding of the research system could be deepened continuously. New demands on collection scope and

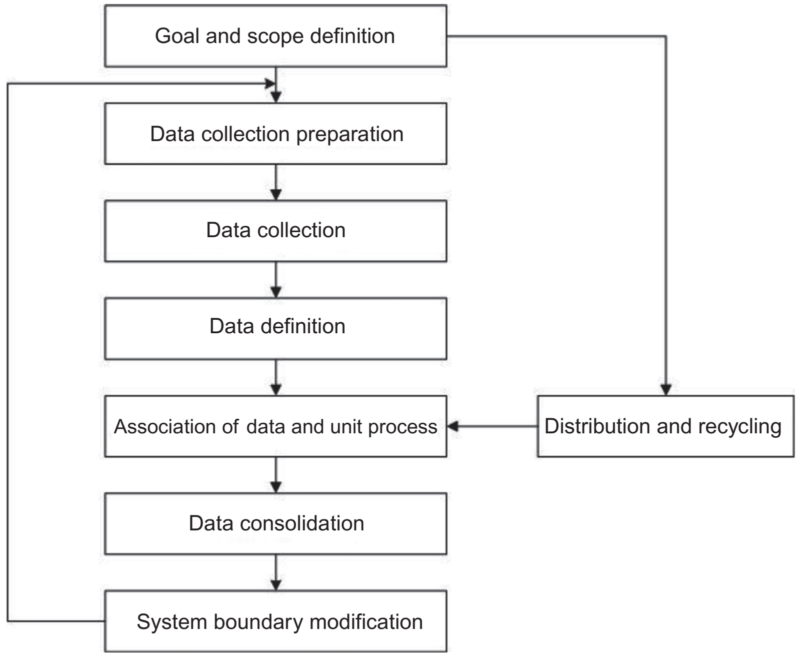


FIG. 5.2 Diagram of life cycle inventory analysis (Roy et al., 2009).

quality of data may come out and the data collection program should be modified in line with the research target. Fig. 5.2 shows a diagram of life cycle inventory analysis (Roy et al., 2009).

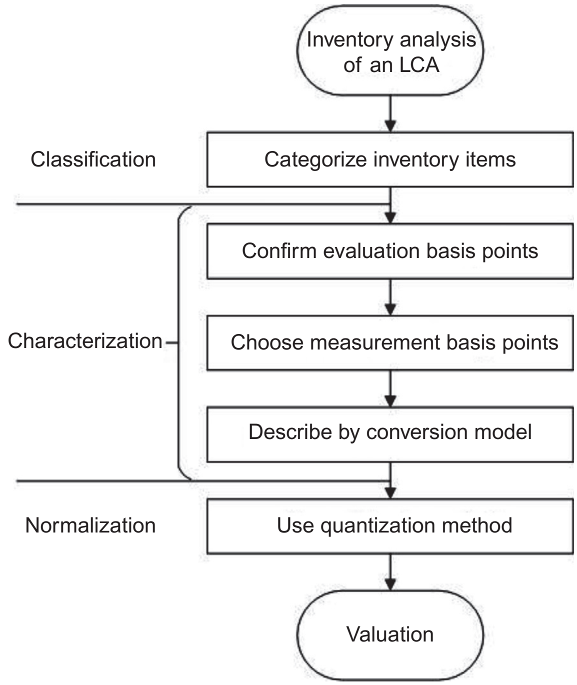
The Eco-Indicator 99 is a powerful and useful tool; it is a weighting method used for product design and also in most LCA studies. It can make an LCA more understandable and transform the LCA results into user-friendly units: the Eco-indicators (Ministry of Housing, 2000).

5.2.1.3 *Impact assessment*

The impact assessment is the work center of an LCA. According to the data provided by the inventory analysis about consumption and emissions, the environmental impact of product will be evaluated (Finnveden et al., 2009). The impact of environment is also defined when matter and energy are exchanged among the research and environment. However, compared to other phases, the maturity of impact assessment is still not enough, and more work is needed. Fig. 5.3 shows the execution steps of impact assessment (Liu and Ma, 2009). Several elements make up the impact assessment, including classification, characterization, normalization, and valuation.

After clarifying the environmental impact categories that the study concerns, the inventory data will be distributed into different environmental impact categories. There are different environmental categories in different impact assessment methodologies. For example, SETAC classifies the environmental impact into three types, which are the impact on ecosystems, human health, and resource consumption. Guinée (2002) divide environment problems into the exhaustion problem, the pollution problem, and the perturbation problem. Different

FIG. 5.3 Executions steps of life cycle assessment impacts assessment.



subclasses are also separated from the broad categories of environmental impacts, when the broad category of the impact on ecosystems consist of global warming, acidification, photochemical smog, ozone depletion, water eutrophication, and so on (Du and Karoumi, 2014). Different categories of environmental emission could make the same kind of environmental impact. For example, the inducing factors of haze include PM2.5, SO_2, NO_X, and so on, but SO_2 could also cause acidification.

The environmental impact potential of the environmental emission factor is quantified by characterization, which is the basis of related knowledge about physics, chemistry, biology, and toxicology. The computational models of characterization consist of load model, equivalent model, inherent chemical behavior model, overall exposure-effect model, and site exposure-effect model. The equivalent model is the most commonly used model. In this model, each impact type will be offered a specified matter as reference substance, called the characteristic factor. The impact potential to one impact type of the other matter will be measured by the characteristic factor. For example, the ability to cause the greenhouse effect of greenhouse gases will be expressed as CO_2 equivalents, where the ability to cause the greenhouse effect of exhausting 1 kg CH_4 is equivalent to the ability of exhausting 25 kg CO_2.

The final step is normalization. According to the contribution, different types of environmental impact are weighted to evaluate integrated environmental impact (ISO, 2006c). There are many environmental impact types to be investigated. The normalization is a necessary step, which helps decision-makers make an overall consideration about every aspect of environmental impact, avoiding decision mistakes caused by lack of knowledge.

5.2.1.4 *Interpretation*

Interpretation is the last phase of an LCA, which is made up of summary and discussion about inventory analysis and impact assessment. The weakness of a product or a craft process will be identified, and relevant suggestions will be offered (ISO, 2006b).

5.2.2 Research progress

The research related to the environmental management standards ISO 14040 and ISO 14044 made by the International Standardization Organization (ISO) have the most impact on LCA, which reflects the research consensus of LCA all over the word. At present, the main research directions of LCA include inventory analysis methods and impact assessment methods. The theory of the inventory analysis method, which places emphasis on normalizing the data acquisition, is reaching maturity. The research of the impact assessment impact consists of an assessment index system, impact assessment characterization model, the normalization of the assessment results, and so on. It is reflected in a rising series of environmental damage types, building mathematical models of life loss, and measuring and determining the toxicity of the pollutants to human health and ecosystems (Hertwich et al., 1999). Up to now, the methodology and reference system of the LCIA is still developing forward continuously. There is no uniform standard accepted generally. There are different kinds of methods proposed, internationally, to evaluate the impact, such as eco-scarcity method (Hanssen, 1999), environmental priority solution (EPS) method (Steen et al., 2019), eco-indicator method (Spriensma, 1999), and environmental design industrial product (EDIP) method (Wenzel et al., 2000). The characteristics and weight approaches used by the environmental impact assessment methods above are listed in Table 5.1 (Hanssen, 1999; Finnveden, 1997).

TABLE 5.1 Summary of some LCIA methodologies that include weighting approaches.

Assessment method		**Eco-scarcity**	**EPS**	**Eco-indicator**	**EDIP**	**LIME**
Major version		1991 1997	1992 1997 2000	1995 1999	1997 2003	2003
The range of application		Switzerland	World	Europe or Netherlands	Denmark	Japan
The weighting selection of the characterization factors	Average value				√	
	Specific scope		√	√		√
	Goal-distance method	√			√	
	Expert decision			√		
	Monetization method		√			√

At present, the research is combined with material flow analysis (MFA) and the LCA is carried out internationally, which is one of the important directions in which to extend the application range of LCA. For example, the environmental science research center at Leiden University in the Netherlands calculated the exhaust potential value of ore resources and built many metallic environmental impact characterization models. It offers important reference standards and basic information for the evaluation of original resources, energy consumption, and environmental impact of industrial production (van Westenenk et al., 2019). Swedish scholars combined SFA with LCA, and recorded the emission data of the polyvinyl chloride (PVC) material chain and all the others related to PVC in Sweden. They have also turned the data into an environmental subject score of LCA, which provides clear direction to the life cycle assessment of PVC (Tukker et al., 1997).

5.2.3 Application situation

An LCA method can evaluate the environmental aspects in the decision-making process by studying the whole life cycle of products, industry, and even the industrial chain. The evaluation could be strategic, or be a specific operation, which makes the industrial internal behavior more in accord with the principle of sustainability.

5.2.3.1 Used in industry and enterprise sectors

The major applications of an LCA in industry and enterprise sectors consist of:

(1) the life cycle assessment of the production process and the integration production progress; and
(2) the life cycle assessment combined with the industrial long-term planning and the logistics analysis in strategy formulation.

The primary territories can be summed up as:

(1) the identification and diagnosis of the product system;
(2) the evaluation and comparison of the life cycle assessment of a product;
(3) the evaluation of the effect of product improvement;
(4) the ecological design of products and development of new products;
(5) the process design and the recycling management; and
(6) the audit of cleaner production (Houillon and Jolliet, 2005; Perugini et al., 2004; Yang et al., 2004; Lopes et al., 2003; Xiang et al., 2003).

5.2.3.2 Used in government administration and international organizations

Life cycle assessment can solve the problems of the rational allocation of resources and environment in all life cycle stages (production, use, recycle, and disposal), from the microcosmic aspect. The interaction and impacts among the socioeconomic system and the natural ecological laws system provide the basis for government administration to develop the environmental policy of region and industry macroscopically (Carlsson Reich, 2005; Funazaki et al., 2003; Park et al., 2003; Chevalier et al., 2003; Shiels et al., 2002). The main content aims to:

(1) formulate environmental policy and international management system, coordinate the regional or global environmental problems, improve and protect environment by standard ways, and satisfy the needs of sustainable development of economy;
(2) establish environmental product standard and implement ecolabelling plan;
(3) formulate corresponding tax, credit, investment, and environmental protection policy, and accelerate the development of the industry of waste recovery and recycling;
(4) optimize energy, transportation, and waste management solutions of government, minimize the environmental load and economic cost of the economic system; and
(5) provide the public with information about the related products and raw materials, improve ecological consumption benchmarks, and advocate green and sustainable consumption.

5.2.4 Limitation

Though the technical framework and analysis features of an LCA have also been widely accepted and understood, there is no consensus on the specific operation. Therefore, LCA faces many complications, as exhibited in the definition of system boundary, the definition of the impact categories, the choice of the impact evaluation models, and so on. As far as evaluation methods are concerned, some defects remain in current impact evaluation models and integration methods.

5.2.4.1 Objective problems

It is almost impossible for life cycle assessment to avoid the influence of subjective factors, which is determined by the understanding of the LCA method, the knowledge of the system evaluated, the knowledge background, and the value judgment of the executors of an LCA (Huijbregts, 1998; Owens, 1996). For example, in one respect, the choice of the system boundary depends on the cognition degree of the object of the study and the predetermined research target. In another aspect, when quantificationally evaluating an impact type, several kinds of models can be used to determine the effect. Different models can offer different results, some of which can be measured in orders of magnitude. Thus, artificial selections of impact assessment models make the evaluation results subjective.

5.2.4.2 Limitations of information and data

For LCA, the limitations of information and data are one of the major hindrances (Ciroth, 2004; Ross et al., 2002). This hindrance appears in two aspects: lack of information which can be used as influence category basis, and lack of information and data from each stage of the LCA. It is usually hard to obtain the related data and the data gained is of low quality. The typical production process or the mean level are adopted as the substitutes. The data may be estimated by empirical formulas (or experience judgment); this means that the data may be inaccurate and the error may be bigger, which could be misleading. It is also not easy to solve the problems above. In order to solve the problems, the mining industry, the raw material production industry, and the product manufacturing industry should work together, effectively.

5.2.4.3 Time and geographical constraints

Space and time should be integrated in environmental effect aspect in LCA, whether the original data or the evaluation result both have limitations of time and space. Additionally, the LCA comes from the European perspective. Therefore, the impact categories included in the prototype of the LCA reflect Europe's environmental problems. Though the integration of space and time can compare the different systems according to the consistent boundary definition, some significant environmental impact of the systems beyond Europe are covered.

5.3 The economic assessment—LCC

Life cycle cost (LCC) is the sum of the costs throughout the whole life cycle of a product. Theoretically, an LCC covers the entire life cycle of a product or an engineering project. The life cycle cost assessment is an economic evaluation of a product or an engineering project across its lifetime, which helps decision makers to choose the best investment plan, on the basis of the least cost (Woodward, 1997; Khan et al., 2010). An LCC can be expressed as follows (Andrae et al., 2004):

$$\mathrm{LCC} = IC + OC + DC$$

where IC is initial investment cost, OC is operating cost, and DC is discarding cost.

The whole life process of a system or an equipment (planning and design, acquisition and installation, operation and maintenance, renewal and reform, and even scrap and recycle) will be taken into account, synthetically, which makes minimum the life cycle cost. After building the model of an LCC, the profound influence and insignificant influence can be gained by sensitivity analysis to provide a reference for later decisions.

In an LCC, the traditional mode, aiming at minimum acquisition expense, has been abandoned, which is a major breakthrough of cost decision. The new mode is not like the old way of only thinking about short-term benefits or particular cost of equipment. The key point of an LCC is to estimate the overall cost. For an example, the control strategy with the target for the minimum annual burden can't show the minimum overall cost of the equipment or the system. The essence of an LCC is to ask decision-makers to take the whole situation into account and plan accordingly, and mainly consider the long-term benefit. The significance of an LCC can be shown as follows:

(1) according with the strategy of sustainable development;
(2) avoiding unnecessary loss caused by blind selection, and making decisions more scientific and effective; and
(3) allocating resources efficiently.

5.3.1 Classification

According to different classification criteria, there are three methods to classify LCC, which are content dependence, time dependence, and cost dependence (De Benedetto and Klemeš, 2009; You et al., 2012).

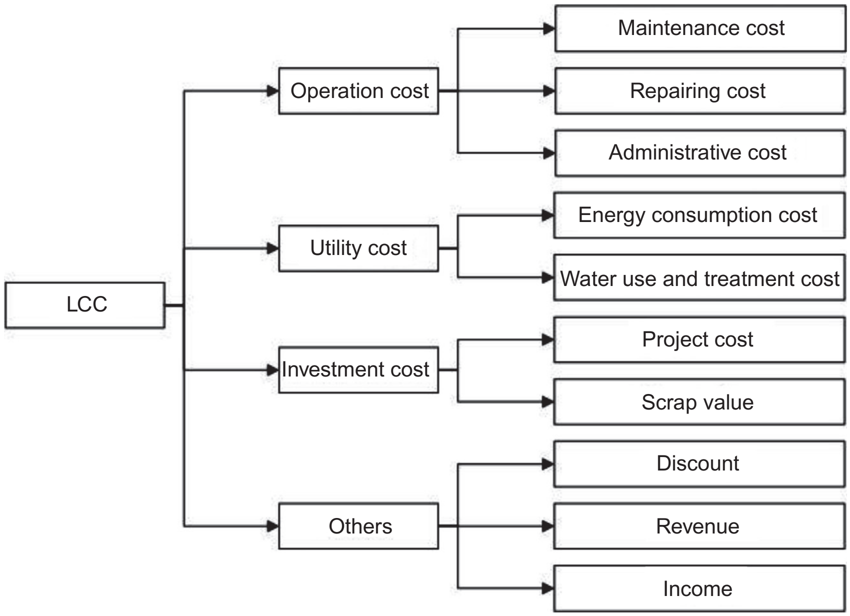


FIG. 5.4 Content dependence classification.

5.3.1.1 *Content dependence*

On the basis of content, the LCC could be divided into four categories, which are operation cost, utility cost, investment cost, and others, as shown in Fig. 5.4.

5.3.1.2 *Time dependence*

Two categories are divided from the LCC, initial cost and future cost, on the basis of time. The initial cost is the total cost before the equipment is put into use, while the future cost is the total cost of the equipment from being put into use to being scrapped. The future cost mainly consists of nonrecurring cost and repetitive cost. The nonrecurring cost (nonannual cost) is the sum of necessary nonrecurring expense to keep the equipment in good condition when the equipment starts running. The repetitive cost (annual cost) is the accumulated cost regularly devoted to make the equipment run smoothly, including maintenance cost, operating cost, administrative cost, and repair cost (Fig. 5.5).

5.3.1.3 *Cost dependence*

In consideration of cost dependence, an LCC could be divided into the three categories of operation and maintenance cost, alternative cost, and construction cost. Some subclasses are also included in this classification, so that cost function can be defined. Each cost can be expressed by a tree diagram (different equipment consists of different costs) for the convenience of observing and cost analysis.

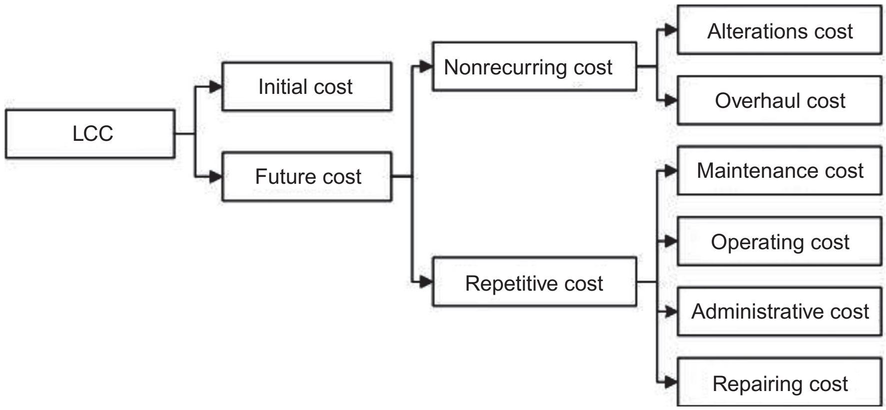


FIG. 5.5 Time dependence classification.

5.3.2 Analyze content

Compared to a traditional economic evaluation method, which only considers one expense, LCC takes the whole life cycle cost as the standard. The core of an LCC is to obtain expenses of different equipment or projects and estimate the total cost of the LCC by a corresponding estimation method. The content of an LCC consists of several aspects, as discussed below (Politano and Frohlich, 2006).

5.3.2.1 Cost breakdown

In order to improve computational accuracy, the LCC should be classified from top to bottom to gain the minimum cost unit until it can be evaluated. The minimum cost unit is a tree diagram that is neither repeated nor omitted, which includes all related cost units.

5.3.2.2 Cost estimate

The cost estimate of an LCC is aimed at providing better choice among different products and equipment. The most effective method is to quantify them as unit cost for comparison. The process of estimation is also before the expense, called cost modeling. The problems between different algorithms can be solved by constructing cost estimation methods for each expense.

5.3.2.3 Cost conversion

Some differences exist in the fixed number of years of cost estimation, the service life of equipment and system. The money also has different value in different years. It is thus necessary to standardize the parameters above for the convenience of calculation and comparison. There are three common methods: the net present value method, the equivalent annual method, and the final value method.

5.3.2.4 Sensitivity analysis

All cost units have many elements affecting the result, with different kinds of units. In order to confirm whether an element is important, the influence of elements to cost units should be analyzed quantitatively. The quantitative indices will also be gained, and the decision basis for life cycle cost is provided.

5.3.2.5 *Trade-off analysis*

The trade-off analysis has two main demands: that the choice will be included in adequate alternatives, and that balance and comparison are made between the life cycle costs. It turns out that the trade-off analysis is very effective for reducing the life cycle cost, especially for maintenance and support costs in the middle and later periods. The early costs for security and safety of equipment not only ensure smooth operation, but can also reduce the huge expense of maintenance.

5.3.3 Estimating method

According to different stages of information at hand and cost estimation, an LCC estimating method can typically be divided into three methods, which are parameter estimating method, analogy estimating method, and engineering estimating method.

5.3.3.1 *Parameter estimating method*

Parameter estimating method (parameter analysis method) is aimed at studying the inner system of cost estimation on the basis of parameters and variables. Estimating the life cycle cost is to choose the physical and performance parameters with the greatest and most obvious influence, building on the original data of similar equipment. At the same time, the relationship between cost and parameters is expressed by the mathematical model of regression analysis. In different application areas, this method has different application modes.

Showing the advantages of convenience and speed, the parameter estimating method inputs performance parameters into the system and gains the relationship between the estimation equipment and the cost (or the characteristic quantity and the cost). And the difference of changing the design scheme will also be evaluated objectively. In sample points gained by the parameter estimating method, several problems (such as error, delay, alteration, and so on) exist in the system, directly manifested by sample points. Compared to the analogy estimating method, the parameter estimating method can improve stability of the estimated value of the new cost, because of fewer variables and less outside impact.

The parameter estimating method still has some disadvantages: reliable and effective cost estimation must have huge historical costs and data stored into the database to make the sample useable again in cost estimation. After updating the system, the risk of the extrapolated database will be improved. There is a huge difference between the updated system and the quondam system; so, the system with big update range or technical differences will be refused.

5.3.3.2 *Analogy estimating method*

In the analogy estimating method, the life cycle cost will be estimated from the data of the known equipment first. Then, the equipment and data information will be analyzed and compared, and the coefficient will be determined by the related parameters, which makes the process of estimation complete. The above method is usually used in scheme reform, scheme planning, and argument of the equipment implementation. There are several impact factors of the analogy estimating method in the estimating process, which are condition, time, and level of similarity and dissimilarity of the similar products. The mathematical model can be expressed as:

$$C = C' n_1 n_2 n_3$$

where C is the cost of an LCC, C' is the similar cost of an LCC, and n_i ($i = 1, 2, 3$) are similar coefficients, respectively.

The analogy estimating method could be divided into two types. One of them is to estimate roughly the cost of the similar equipment or the equipment to be built. The other is to estimate roughly the costs among the similar cost equipment and the equipment to be built. In the final stage of an LCC, the analogy estimating method will be widely used and shown full expression in similar application of data.

5.3.3.3 *Engineering estimating method*

In the engineering estimating method, the cost of each unit will be calculated first. Then the sum of the costs above will be determined, which is the overall life cycle cost. Before the process, the detailed parameters of system and the cost of equipment maintenance are needed simply. The usage frequency is usually high in late problem. The mathematical model can be shown as flows:

$$C = C_1 + C_2 + C_3$$

where C is the life cycle cost and C_i ($i = 1, 2, 3$) are unit costs in different times, respectively.

5.4 The social assessment—S-LCA

5.4.1 Research status

The social life cycle assessment is a social impact evaluation tool, which is aimed at evaluating the social and sociometric impact of a product. It is the only way to evaluate the social impact from the angle of the life cycle (Lehmann et al., 2013). The impact includes positive and negative influence in the whole life cycle of the product (Benoît et al., 2009). The whole life cycle consists of mining, processing, manufacture, transport, use, recycling, maintenance, and disposal. The S-LCA is another type of LCA, placing emphasis on the social aspect (Klöpffer and Ciroth, 2011).

Compared to other social impact assessment methods, the biggest difference is the study of the whole life cycle of a product. The directly positive or negative impacts on stakeholders are included in the social impacts of an S-LCA. When scaling up, the indirect impact should also be taken into consideration. These impacts are related to the behavior of enterprise, the sociometric process, and the accumulation of social capital.

By analyzing the pertinent literatures, the research statuses of the S-LCA field are as follows.

5.4.1.1 Guidance document

In 2009, the United Nations Environment Program (UNEP) and Society for Environmental Toxicology and Chemistry (SETAC) published *Guidelines for Society Life Assessment of Products* (hereinafter referred to as *Guidelines*). *Guidelines* (Benoît et al., 2009) takes ISO14040: 2006 *Environmental management-life cycle assessment-Principles and frameworks* and ISO 14044: 2006 *Environmental management-life cycle assessment-Requirements and guidelines* as the skeleton. The four phases of goal and scope definition, inventory analysis, impact assessment, and interpretation have been put forward. Each has been stipulated and described in detail. After that, case studies have full references. In the inventory analysis phase, *Guidelines* provides two social impact categories. One of them departs from stakeholders and the other goes form the impact types, which provides the foundation for S-LCA to development database and design software.

In 2010, the UNEP/SETAC came up with an S-LCA methodology manual, which expounded in detail the impact categories based on stakeholders (Benoît-Norris et al., 2011). The stakeholders consist of workers, local community, society, consumer, and value chain-participant. The impact categories include 31 subclassifications, including fair pay, technological development, fair competition, and so on. The manual gave the exact definition of the each subclassification, listed the international conventions and agreements related to the subclassification, and put forward the common goals of all mankind and suggested instructions.

5.4.1.2 Theoretical research

In the years before and after 2009, when *Guidelines* was published, scholars meticulously discussed the theoretical framework of the S-LCA. The standardized indicators missed in the methodology have been investigated. It also proposed the adjustment method for the classification method of stakeholders. All the above have obtained some achievements.

Benoît-Norris et al. (2011) summarized the development of the S-LCA methodology put forward by UNEP/SETAC and suggested that the methodology manual still needed to be perfected with the development of the case study. Meanwhile, he pointed out that the methodology manual only provides the inventory indicators of each social classification and the approaches of data collection, but the method to uniformly quantize the data of the research result had not been explained, which plays a role for standardizing models in LCA. Therefore, the author suggested that the standardization models need further investigation.

Dreyer (2009) summarized the development of the S-LCA methodology and put forward a set of frameworks of the S-LCA, combined with the previous work. He also summarized the methods of quantitative evaluation and set up four models of impact catalogue. The methods in the paper have been used in case studies. Mathe (2014) discussed the classification of stakeholders. He thought that the classification of stakeholders should not be limited to the five categories proposed in *Guidelines*, which should be supplemented and adjusted.

5.4.1.3 Case study

In the case study aspect, many scholars use different select and quantitative methods of indicators. By evaluating the social life cycle assessment of different products, kinds of conclusions, good for decisions, have been gained. Some scholars have compared the social impact of different products. The pros and cons of two products in the social impact aspect have been investigated, and the best product has been chosen.

In 2009, Blom and Solmar evaluated the behavior of ethyl alcohol, biodiesel, and marsh gas in their own life cycle on the basis of the S-LCA technical framework put forward in *Guidelines*, from the human rights, the working conditions, health, safety, the culture heritage, the government management, and the sociometric reverberation. The method combined with the qualitative and quantitative methods has been used to score three kinds of fuels. The result shows that the social impact of marsh gas is the best and that ethyl alcohol is the worst, which proves that the biofuels have superiority in social impact.

In 2014, Hosseinijou et al. evaluated the social life cycle impact of rolled steel and cement used in northern Iran, on the basis of the classification of stakeholders raised in *Guidelines*. The paper simplified the inventory by analyzing material flow and expert interviews, and the analytic hierarchy process has been used to conduct quantitative evaluation. The result shows that rolled steel is better than cement, viewed from the angle of the S-LCA. According to the study result, the author also put forward some proposals for the two industries.

Hunkeler (2006) took labor time as the intermediate variable and quantificationally turned the environmental cost of products into the social endpoint impact, according to the labor time used to pay for the social life (residence, medical treatment, education, and so on). This characterization method has been used to evaluate the social life assessment of two different detergents.

Some scholars have built the social impact assessment models of products on the basis of the S-LCA methods. By using and studying these cases, the key factors of the social impact of product have been found out. In 2013, Maink et al. carried out the work of the S-LCA about the palm biofuel produced in Jambi province, Indonesia. According to the research result,exploitative labor relations are the most important factor for the sustainable development of palm biofuel. Local communities and laborers bear the major social cost of the industry development. Aparcana and Salhofer (2013), in 2014, suggested the use of the S-LCA method to evaluate the social impact assessment of the garbage collection and recycling system in low income countries. The three-level evaluation index system of the garbage collection and recycling system in low income countries has been built on the basis of the S-LCA method, and the social impact has been evaluated by 26 semiquantitative indexes.

Feschet et al. (2013) set up the quantization conversion approach of the changes in health status and the economic benefit in the social impact, in accordance with the Preston curve in economics (the relation between the life span and the GDP, raised by Preston). They also used this approach to study Cameroon's banana industry. In 2015, Dong and Thomas put forward the stakeholders and the production phase from cradle to grave based on *Guidelines*. They chose three stakeholders and the phase from cradle to construction completed; the weight was determined by expert decision. Finally, the building social impact evaluation model was built, which was used to evaluate the social life cycle assessment of an engineering project in Hong Kong.

In 2013, Ekener and Finnveden used the S-LCA theory to identify the potential social spots of a notebook computer. The potential social impact in this case has also been studied. Baumann et al. (2013) used the empirical S-LCA method to compare the damage, which was caused by the supplemental restraint system in its life cycle, and salvation. The main evaluation index was the disability adjusted life year. The result showed that the efficiency of SRS of saving life and preventing damage may be different in different life cycle stages.

The S-LCA was used as a management tool by Gabriella Arcese et al. (2013) in the tourist trade in 2013. The potential adverse social impact was evaluated. The questionnaire was set up on the basis of *Guidelines*. After analyzing the findings, the important factors of the social impact of the tourist trade have been defined. In 2013, Gervásio and Da Silva studied the user cost of the social life cycle impact factors of the expressway bridge, separately. The user cost was divided into the vehicle operating cost, the travel delay cost, and the accident cost. The quantitative calculation models of three costs were provided.

5.4.1.4 Deficiency and prospect

Since *Guidelines* has been issued by UNEP/SETAC in 2009, the S-LCA has great progress and development. But it is still in the primary level of development (Jørgensen, 2013). According to the publications above, there are some deficiencies of the S-LCA, being listed as:

(a) Data collection difficulties Some social impact assessment report and statistical yearbooks of the related department can offer information, but the database is not enough. Data collection is one of the most serious problems of the S-LCA (Benoît et al., 2009). At present, the most urgently needed S-LCA databases are modeling data and social impact data, which leave researchers facing lots of difficulties.

(b) The absence of characterization models. Compared to the LCA, the S-LCA has a big disadvantage, which is a lack of characterization models. For example, LCA can use the characterization models to quantitatively unify the environmental impact as the human injury eigenvalue (Guinée et al., 2011), whereas S-LCA does not have such characterization models.

(c) Can't define the impact of the functional units. As indicated above, the result of S-LCA can't be quantified uniformly, due to the lack of characterization models. So, the impact of each functional unit can't be confirmed, which makes the range of application of the result very limited (Hosseinijou et al., 2014).

(d) The absence of software. With the support of enough data, the software can make the analysis of social spots and the simplification of inventory more reliable and simpler (Lehmann et al., 2013; Benoît et al., 2009).

5.4.2 Social impact and social impact assessment

5.4.2.1 Social impact

In the method of an S-LCA, the social impact will be defined as the impact of social relation combined with a physical activity (production, consumption, and disposal) and actions taken by stakeholders (Benoît et al., 2009). The social impact is usually considered to be complicated, which is the result of the system network and comes with different perspectives. In addition, the social impact also has feedback effect on product systems, and the effect may cause changes of social impact itself. Because of complexity and subjectivity of the social impact, it is unsuited to unilaterally describe the state of product systems in an S-LCA. So, a set of indicators is established from the stakeholders.

5.4.2.2 Social impact assessment

Social impact assessment is a technical means to analyze and evaluate the impact and result of policies, projects, events, activities, and so on in social aspect. Social impact assessment is a specific social science research method applied to policies or projects, which is aimed at understanding the situation, reasons, and results of social life. Scientific knowledge and methods will be used to analyze the social changes, impacts, and results caused by policies or projects, and useful knowledge or policies will be offered to reduce negatives and achieve effective management (Benoît et al., 2009).

5.4.2.3 Social life cycle assessment

Social life cycle assessment (S-LCA) is an evaluation tool used to evaluate potential positive or negative effects of a product in its whole life cycle in social aspect, including the process of raw material mining, production, distribution, application, reuse, maintenance, recycling, and final disposal. The S-LCA can not only be used alone but also combined with E-LCA, supplementing the S-LCA of a product (Benoît et al., 2009). The E-LCA also has the same technical framework as LCA, as shown in Fig. 5.1, meaning that the S-LCA is the expansion of the LCA in social aspect.

The generic and fixed-point data will be used to evaluate the social life cycle (supply chain, including service stage and disposal stage) assessment of a product. The difference between an S-LCA and other social impact assessment tools is the object (products and service) and its range (the whole life cycle). In an S-LCA, the social contents of evaluated products have positive or negative effect on stakeholders, directly. The effect may be connected with the impact of enterprise behavior, socio-economic process, and social capital, and the indirect impact of stakeholders will also be taken into consideration.

An S-LCA doesn't have an aim of judging whether a product should be produced or not; nor can an S-LCA give a report to stakeholders by itself, but record utility of a product. Theoretically, an S-LCA has multiple uses, even to evaluate things that may obviously endanger society (such as weapons). An S-LCA will provide social information for decision-makers, encourage the exchange of ideas about production and consumption in all aspects of society, improve work efficiency and finally aim at benefits of stakeholders (Benoît et al., 2009).

5.4.3 Technical framework

5.4.3.1 Goal and scope definition

The goal definition is the first step of an S-LCA, which should describe the intended use and ambition of the S-LCA; and then specific research can be defined to achieve the goal under the constraint conditions.

The second step is to confirm the scope of the research. As a part of the scope definition, the productions and functional units should be defined. On the basis of this information, the product system can be modeled by using the process data or in/output data. In the stage of scope definition, the depth of research should also be defined, and it is necessary to decide which units need general data, or need special data collection.

The stage of goal and scope definition includes: expatiation of the object of study (including goals, product functions, product utility, functional units, and so on); the definition activity variables for use and unit procedures contained; preparation for collecting data and detailed description about which data need to be collected and to which impact category the data belongs; and the confirmation of all stakeholders involved in life cycle of products and the types of comments needed.

The final aim of an S-LCA is to accelerate the development of social conditions and social economic performance of products in the whole life cycle for stakeholders. Another potential purpose (promoting the improvement of social economic conditions, encouraging stakeholders and decision-makers, having conversations with government officials) of using S-LCA is also very important, and it also should encourage stakeholders to participate in the process of goal and scope definition.

5.4.3.2 *Impact assessment*

The social life cycle impact assessment (S-LCIA) is the third phase of an S-LCA. The purpose of the S-LCIA is to integrate the inventory data into sub classification and classification, and to help to understand the magnitude and significance of the data collected from the stage of inventory analysis by using extraneous information (Benoît et al., 2009).

The phases of the S-LCIA consist of three mandatory steps, put forward by ISO 14044 (2006) and aim at LCIA. The inventory data can be traced by the related social relationship and the social economic impact can be defined. The three steps are: choose impact classification, characterization methods, and characterization models (classification); associate the inventory data with the specific subclassification and impact classification of S-LCIA; and confirm the result of computational subclassification indexes (characterization) (ISO, 2006c).

(1) Choice of impact classification, subclassification, and characterization models. The choice (impact classification, subclassification, and characterization models) should be kept consistent with the objective and scope of the research. The impact classification is the logical collection of the S-LCA results, related to the social interests of the stakeholders and the decision-makers. In LCA, two kinds of impact classification have been defined, which are end-point form and mid-point form. The end-point form seeks for environmental damage on behalf of the field of protection (such as biology, natural environment, or human health); the aim of the mid-point form is the environmental problems among the inventory and the field of protection (Anon, 1993). The impact assessment shows the causal chain flowing from the inventory to the mid-point indicator, and evaluates the final result by extending the causal model.

Similar to an LCA, there are two social impact categories put forward in an S-LCA. The first category integrates the results of the subclassifications into the interest subjects corresponding to stakeholders (like government); the other category presents the modulization of the results of the subclassification, and also has the causal relationship according to the criterion's definition (like health and safety).

(2) Classification. This step distributes the inventory results to the specific stakeholder categories or the impact categories.

(3) Characterization. The computation of the categories results is included in this section. The ISO 14044 describes the process. The computation of the indicator results (characterization) includes transformation with LCI results and general units, and the results after transforming has also be integrated in the same impact categories (Guinée et al., 2011). The characteristic factors are used in this kind of transformation. The computational result can be expressed by a numerical indicator.

In S-LCIA, the characteristic models under the social and economic influencing mechanisms can't always be operated in mathematics. It may be a logical integration step, and aggregates words or inventory information into a single concept, and can also merge the quantitative social and sociometric inventory data into one category. The characteristic model can also be more complex, which includes the use of additional information (like performance-related points). According to international conventions or the best practice, the performance-related points may be set threshold value. It also needs to be transparent and to be recorded.

There is one important difference between LCA and S-LCA. In LCA, the characteristic model is the product of the inventory data and the characteristic factors are defined on the basis of environmental science; but when evaluating society (qualitatively or quantificationally), a points system, on the basis of the performance-related points, is needed to help evaluate meaning of the inventory data, which is an estimate about impact. Contrary to LCA, the grading and weighting steps of S-LCA may not proceed in characterization step, and the attention is needed that the model and standard of defining characteristic factors, which must be defined and transparent well, in S-LCIA stage. The same goes for the grading and weighting system.

5.4.3.3 *Inventory analysis*

The inventory analysis is a phase to process to collect data, build models and gain social life cycle inventory in a S-LCA. In this phase, the related data will be collected to prioritize data, evaluate hot spots, evaluate fixed-point, and assess impact (characterization). In this phase, the data needs to be verified and the system boundary should be confirmed. Then, the data will be related to functional units and integrated according to different situations.

On the basis of the research target and the definition of scope, the inventory analysis can be started initially. The specific steps can be shown as follows (Benoît et al., 2009):

(1) data collection (filter, prioritize and evaluate hot spot);
(2) data collection preparation;
(3) collect the key data;
(4) characterize;
(5) data validation;
(6) association of data and unit process;
(7) extract the boundary system; and
(8) data consolidation.

The most time-consuming stage is to collect the specific data, which will be used to verify how the organization connects with the production in social and economic aspects. Under ideal conditions, the fixed-point analysis can be finished by accessing the organization, which supplies meaningful input in product process unit. However, though the supply chain is limited, the cost of data collection will be too high and it will also take too much time, which will be impractical. So, it has important meaning for an S-LCA to prioritize the data and predict the significance in the whole process of production.

Random sampling method will be used to reduce the quantity of survey respondents, but the risk of serious problems may be ignored. So, a kind of system with cost-effectiveness, (including hotspots assessment, desktop screening, and limited focus) becomes a feasible option. The data priorities play an important role in a S-LCA. The second step of data collection usually includes the general analysis of social problem of the area, which is the maximum input source in life cycle of a product. Finally, people can get more accurate evaluation combined the regional information and the industry-level data.

5.4.3.4 Interpretation

The interpretation is a process of evaluating results whose purpose is to draw a conclusion. In order to keep consistent with the target and scope of the research, this stage has some specific purposes: analyze results, obtain conclusions, explain the limitation of the research, offer suggestions, and give a report. There are three main steps defined in ISO 14044 (ISO, 2006c): the first step, recognition of the key problems; the second step, the evaluation of the research (include consideration of globality and consistency); the third step, conclusion, suggestion, and report.

An integrated S-LCA evaluation should also include the content of these four main steps:

(1) Recognition of the key problems. The important points of S-LCA are the key social survey results and choices of research methods, which include recognition of core concerns and limitations and assumptions in study; for example, the focus of general evaluation may be a social hot spot, an important inexpectant impact, which is beneficial to society, or a human rights violation discovery in a unit process. In addition, the choice of system boundary and its level of details (each process from the general to the fixed point, then to the product system) are particularly important.

(2) The evaluation of the research. A series of quantitative, half-quantitative, and qualitative methods will be used to evaluate. Some key requirements in the evaluation process include the critical review, the files of the process of evaluation, the action steps ensuring transparency, and the verifiability. The evaluation of the research has two basic requirements (Benoît et al., 2009), which are integrality and consistency.

The integrality evaluation aims at evaluating whether all of the associated important problems have been studied and all necessary data have been collected, or not. It includes the evaluation of the indicators used to gain conclusion and the data gaps.

The consistency evaluation aims to verify the model of the goal and the scope on the basis of initial definition, and verify the correctness of the choice of methodology.

(3) Conclusion, suggestion, and report. The conclusion and the suggestion should be presented in accordance with the goal and the scope of the research. It should put forward a preliminary conclusion and verify whether the conclusion is consistent with the demand of the research or not. If not, the initial steps need to be changed, and, if consistent, the report of the conclusion could start. The report should be absolutely transparent, which means that all the assumptions, the fundamental principles, and the choices should be explained. The suggestion is a way to express the choice of action and the conclusion can be put forward by different ways, according to intended audience and ability to support the conclusion. In order to analyze easily, the conclusion can be expressed by the following forms (Benoît et al., 2009):

a. high level risks/hot spots/influences in each stages of the life cycle, positively or negatively;
b. the most likely hot spots/influences in the life cycle; and
c. the recognition of the hot spots/influences related to stakeholders.

(4) Participation of stakeholders. It is very important of the participation of stakeholders in S-LCA reports, especially in a specific case.

5.4.4 The similarities and differences between S-LCA and LCA

According to Section 5.4.2, the development of S-LCA is based on the evaluation method of LCA. So there are many similarities between S-LCA and LCA. However, some traditional methods of LCA can't apply to S-LCA, because of the complexity and diversity of social impact. So, some changes of theory and method must be taken, according to the actual situation. The similarities and differences between S-LCA and LCA will be shown in this section. The similarities between the two are as follows:

(1) Use the same ISO technical framework (goal and scope definition, inventory analysis, and impact assessment), though several steps of S-LCA have some differences.
(2) Heavy demand for data.
(3) Proceed in an iterative manner.
(4) Need peer review when preparing for communicating with public or comparing.
(5) Provide useful information for decision-makers.
(6) Purpose is not to determine whether a product should be produced or not.
(7) Have the same effect to evaluate a hot topic.
(8) Give no expression of the impact of functional units when using semiquantitative or qualitative data.

Although S-LCA and LCA have many similarities (the same technical framework), there are still some differences among them. The most significant difference is that the focus points of them are different. The focus of an LCA is environmental impact assessment, while the focus of an S-LCA is on the social economic aspect. Where the LCA places emphasis on collecting the physical quantitative data of products and others (production/use/abandonment related), the S-LCA will collect organizational additional information in supply chain. The differences between LCA and S-LCA are shown in Table 5.2.

5.5 The sustainability assessment—LCSA

5.5.1 Theory and practice

According to UNEP/SETAC, the life cycle sustainability assessment (LCSA) will evaluate the environmental, economic, and social impacts of a product (or a craft, or an activity) in its whole life cycle. The evaluation results will be used in the decision-making process. Klöepffer integrated three life cycle assessment methods into a unified framework, which becomes the

TABLE 5.2 Differences between LCA and S-LCA.

Phases	Characteristics
Goal and scope definition	The description of product use is needed in both the two methods, but the S-LCA requires participators to take use stage and function into consideration
	Connect with stakeholders is encouraged by LCA in preliminary studies, while the S-LCA encourages to connect exterior stakeholders and providing impact input
	In the S-LCA, when the subtype is not included in studying, the reasons are needed. However, it is not needed in the LCA
	In the S-LCA, the classification of subtypes is on the basis of the types of stakeholders and impact, while in the LCA, it is only on the basis of the types of impact
	It is more geographically specific in the S-LCA than it in the LCA
Inventory analysis	It is more frequent to collect and apply activity variable data in S-LCA
	The subjective data is more suitable for S-LCA, but sometimes it may cause lots of uncertainty
	There are several differences between quantitative, qualitative, and half-quantitative data
	The sources of data are different
	It has many steps and methods to collect data
Impact assessment	The characteristic model of the S-LCA is different
	The use of performance parameter points of S-LCA is special
	The S-LCA may both have positive and negative influence, but the LCA has rarely positive influence
Interpretation	The significant problems are different
	The additional information of mutual constraint relation of stakeholders in the S-LCA

LCSA thought (Kloepffer, 2008). The expression of the LCSA raised by Finkbeiner et al. can be expressed as (Traverso et al., 2012):

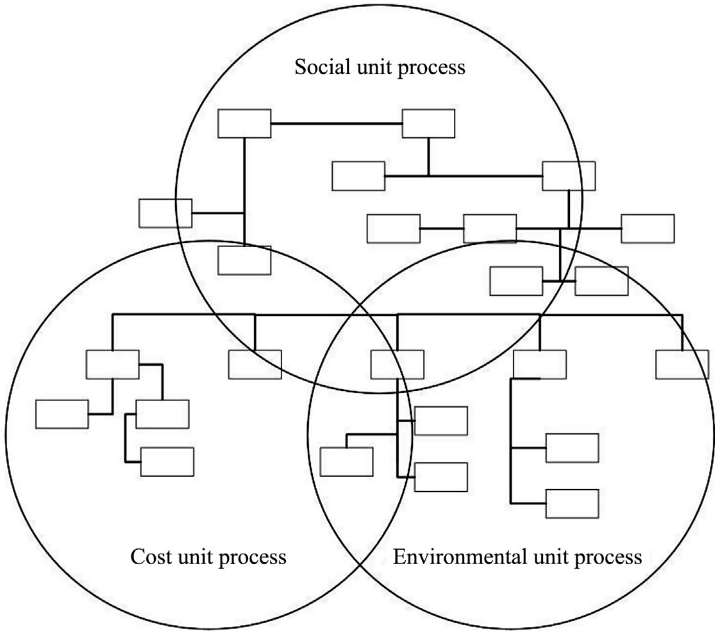
$$\mathrm{LCSA} = \mathrm{E} - \mathrm{LCA} + \mathrm{LCC} + \mathrm{S} - \mathrm{LCA}$$

where the E-LCA is environmental life cycle assessment (this can also be called LCA directly), the LCC is the life cycle cost, and the S-LCA is the social life cycle assessment.

The concept of sustainability assessment of a product can be described by three methods entirely. Fig. 5.6 shows the system boundary of an LCSA (Traverso et al., 2012).

Because of the differences between the LCA, the LCC, and the S-LCA in system boundary, data inventory, and evaluation indicator aspects, the research of the LCSA faces many difficulties (Kloepffer, 2008; Traverso et al., 2012). Therefore, the theory and models still have no definite standard, and the application of the LCSA is also not common. In the past, the LCSA has been expressed by the integration of three or two life cycle assessment methods. The research of two LCA methods integration is more usual (Swarr et al., 2011; Hoogmartens et al., 2014; Chiesa et al., 2016; Mistry et al., 2016; Norris, 2001; Lindahl et al., 2014), while the research of three methods integration is less (Neugebauer et al., 2015; Hake et al., 2017;

FIG. 5.6 LCSA system boundary.



Klöpffer, 2003). A lot of discussion about the LCSA is still underway (Luu and Halog, 2016a; Martínez-Blanco et al., 2014; Zamagni et al., 2013; Vahdat Aboueshagh et al., 2014).

There are several typical applications of LCSA. Luu and Halog (2016b) compared the life cycle sustainability of biomass (i.e., rice hull) power and coal power. The result shows that the biomass power is much better than the coal power in the sustainability of environment and cost, but the indicator of the human health of the social impact aspect is a little behind. Foolmaun and Ramjeawon (2013) compared the life cycle sustainability of four kinds of methods to deal with PET container bottles. Huang and Mauerhofer (2016) evaluated the life cycle sustainability of the ground source heat pump. Schau et al. (2012) evaluated the life cycle sustainability of product remanufacturing. Atilgan and Azapagic (2016) carried out the life cycle sustainability for Turkey's electricity and raised a series of sustainability indicators.

In recent years, for example, many scholars have used the LCSA theory into the remanufacturing field. Warsen et al. (2011) compared the environmental impacts of the manufacturing and remanufacturing process of manual transmissions, which returned the result that the improvement of the environment of remanufacturing the transmissions profits from dramatically decreasing the consumption of material and resource. Charles et al. (2010) used LCSA methods to evaluate the remanufacturing and the manufacturing process of telecommunication equipment in its whole life cycle. The result proved that the remanufacturing process is better than the manufacturing process in environmental aspects. Liu et al. (2014) studied the environmental impact of remanufacturing diesel engines on the basis of the LCA. Shi et al. (2015) studied the environmental impact of remanufacturing liquefied natural gas engines, which has been compared with the remanufacturing of diesel engines. Peng et al. (2016) studied the environmental emission of several kinds of remanufacturing cleaning technologies, on the basis of the LCA theory, and the levels of the environmental emission of these technologies have been compared. Wilson et al. (2014) used the LCA theory to study the validity and the potential range of application of laser cladding technology. Additionally, Fatimah and Biswas (2016) evaluated the sustainability of computer products. Amaya et al. (2010) evaluated the environmental impact of truck fuel injectors.

5.5.2 The existing problems

Scholars have conducted a great deal of research on the LCSA theory. Though abundant achievements have been gained, it is still necessary to continue studying. At present, the research on LCSA focuses on the traditional LCA (E-LCA) and LCC methods, in theory or application aspects. The LCA reflects the impacts on ecological environment, human health, resources, and energy consumption respects, while the LCC offers the approach to integrate the cost of economy and environment into the life cycle framework. However, a product or a system will influence not only the ecological environment but also the social environment. With the deepening and development of research, many scholars gradually realized that it is necessary to development the life cycle tools to solve the social problems.

UNET/SETAC found the key problems in the practice of the LCA. There are many difficulties when implementing the life cycle assessment in developing countries. The professional knowledge and data are in short supply, and the LCA cannot participate directly in solving the problems, such as eradicating poverty, getting a job, fair treatment, and other social problems. When the LCA, LCC, and S-LCA are separated, the practicability of the LCA will be limited and it is impossible to provide complete information to the decision-makers from the view of sustainable development. So, it is significant to carry out the S-LCA in life and production.

5.6 Conclusion

This chapter introduced the origin, classification, framework, and application of life cycle sustainable assessment (LCSA), and made a detailed introduction to LCSA, including LCA, LCC, and S-LCA as the three main aspects.

LCA (E-LCA) is an environmental evaluation method, aiming at analyzing the impact of product, system, and activities on the environment. LCA is the earliest application field of life cycle theory. From early evaluation of Coca-Cola bottles to the evaluation of all industries at the present stages, the LCA is reaching maturity and plays an important role in environmental evaluation.

LCC is the concrete application in the economic field, which abandons the traditional method that aims at minimizing the acquisition expenses. The LCC is a breakthrough for making cost decisions. The stakeholders are required to take the whole situation into account and plan accordingly from macroscopic points of view. The long-term benefits are taken into consideration.

LCA and LCC are not enough to achieve the whole sustainable evaluation. In order to make evaluation complete, S-LCA comes up. The evaluation target of an S-LCA is the impact of the social relation combined with material activities and the activities taken by stakeholders, which is the further development of the LCSA.

References

Amaya, J., Zwolinski, P., Brissaud, D., 2010. Environmental benefits of parts remanufacturing: the truck injector case. In: 17th CIRP International Conference on Life Cycle Engineering, Hefei, ANHUI, China.

Andersson, K., Eide, M.H., Lundqvist, U., Mattsson, B., 1998. The feasibility of including sustainability in LCA for product development. J. Clean. Prod. 6, 289–298.

Andrae, A., Moller, P., Anderson, J., Liu, J., 2004. Uncertainty estimation by Monte Carlo simulation applied to life cycle inventory of cordless phones and microscale metallization processes. IEEE Trans. Electr. Pack. Manuf. 27, 233–245.

Andrews, E., Lesage, P., Benoît, C., Parent, J., Norris, G., Revéret, J., 2009. Life cycle attribute assessment. J. Ind. Ecol. 13, 565–578.

Anon, 1993. Guidelines for Life-Cycle Assessment: A Code of Practice, Society of Environmental Toxicology and Chemistry.

Aparcana, S., Salhofer, S., 2013. Development of a social impact assessment methodology for recycling systems in low-income countries. Int. J. Life Cycle Assess. 18, 1106–1115.

Arcese, G., Lucchetti, C.M., Merli, R., 2013. Social life cycle assessment as a management tool: methodology for application in tourism. Sustainability.

Atilgan, B., Azapagic, A., 2016. An integrated life cycle sustainability assessment of electricity generation in Turkey. Energ. Policy 93, 168–186.

Baumann, H., Arvidsson, R., Tong, H., Wang, Y., 2013. Does the production of an airbag injure more people than the airbag saves in traffic? J. Ind. Ecol. 17, 517–527.

Becker, H.A., Vanclay, F., 2003. The International Handbook of Social Impact Assessment: Conceptual and Methodological Advances. Edward Elgar Publishing, Incorporated, pp. 74–91.

Benoit, C., Dreyer, L., Grießhammer, R., 2006. Feasibility study: integration of social aspects into LCA. Öko-Institut, Freiburg.

Benoît, C., Mazijn, B., Andrews, E.S., N.E.P. United, N.E.P.S. United, 2009. Guidelines for Social Life Cycle Assessment of Products: Social and Socio-economic LCA Guidelines Complementing Environmental LCA and Life Cycle Costing, Contributing to the Full Assessment of Goods and Services Within the Context of Sustainable Development. United Nations Environment Programme.

Benoît, C., Norris, G.A., Valdivia, S., Ciroth, A., Moberg, A., Bos, U., Prakash, S., Ugaya, C., Beck, T., 2010. The guidelines for social life cycle assessment of products: just in time!. Int. J. Life Cycle Assess. 15, 156–163.

Benoît-Norris, C., Vickery-Niederman, G., Valdivia, S., Franze, J., Traverso, M., Ciroth, A., Mazijn, B., 2011. Introducing the UNEP/SETAC methodological sheets for subcategories of social LCA. Int. J. Life Cycle Assess. 16, 682–690.

Blom, M., Solmar, C., 2009. How to Socially Assess Biofuels: A Case Study of the UNEP/SETAC Code of Practice for Social-Economical LCA.

Brent, A., Labuschagne, C., 2006. Social indicators for sustainable project and technology life cycle management in the process industry (13 pp + 4). Int. J. Life Cycle Assess. 11, 3–15.

Cabeza, L.F., Rincón, L., Vilariño, V., Pérez, G., Castell, A., 2014. Life cycle assessment (LCA) and life cycle energy analysis (LCEA) of buildings and the building sector: a review. Renew. Sustain. Energy Rev. 29, 394–416.

Carlsson Reich, M., 2005. Economic assessment of municipal waste management systems—case studies using a combination of life cycle assessment (LCA) and life cycle costing (LCC). J. Clean. Prod. 13, 253–263.

Charles, L.G., Kuester, E., Renee, M., Thomas, A.O., Donald, O., William, J.S., 2010. Lifecycle assessment of the environmental benefits of remanufactured telecommunications product within a green supply chain. In: Proceedings of the 2010 IEEE International Symposium on Sustainable Systems and Technology, pp. 1–6.

Chevalier, J., Rousseaux, P., Benoit, V., Benadda, B., 2003. Environmental assessment of flue gas cleaning processes of municipal solid waste incinerators by means of the life cycle assessment approach. Chem. Eng. Sci. 58, 2053–2064.

Chiesa, M., Monteleone, B., Venuta, M.L., Maffeis, G., Greco, S., Cherubini, A., Schmidl, C., Finco, A., Gerosa, G., Denti, A.B., 2016. Integrated study through LCA, ELCC analysis and air quality modelling related to the adoption of high efficiency small scale pellet boilers. Biomass Bioenergy 90, 262–272.

Ciroth, A., 2004. Uncertainty in life cycle assessment. Int. J. Life Cycle Assess. 9, 141–142.

Ciroth, A., Franze, J., 2011. LCA of an Ecolabeled Notebook: Consideration of Social and Environmental Impacts Along the Entire Life Cycle. GreenDeltaTC GmbH.

Crawford, R.H., 2008. Validation of a hybrid life-cycle inventory analysis method. J. Environ. Manage. 88, 496–506.

Curran, M.A., 1996. Environmental life-cycle assessment. Int. J. Life Cycle Assess. 1, 179.

Czyrnek-Delêtre, M.M., Smyth, B.M., Murphy, J.D., 2017. Beyond carbon and energy: the challenge in setting guidelines for life cycle assessment of biofuel systems. Renew. Energ. 105, 436–448.

De Benedetto, L., Klemeš, J., 2009. The environmental performance strategy map: an integrated LCA approach to support the strategic decision-making process. J. Clean. Prod. 17, 900–906.

de Haes, H.A.U., Jolliet, O., Finnveden, G., Hauschild, M., Krewitt, W., Müller-Wenk, R., 1999. Best available practice regarding impact categories and category indicators in life cycle impact assessment. Int. J. Life Cycle Assess. 4, 66.

De Haes, U., Finnveden, G., Goedkoop, M., Hauschild, M., Hertwich, E., Hofstetter, P., Jolliet, O., Klöpffer, W., Krewitt, W., Lindeijer, E., Müller-Wenk, R., Olsen, S., Pennington, D.W., Potting, J., Steen, B., 2002. Life-cycle impact assessment: striving towards best practice.

Desmond, B., 2002. Life Cycle Costing (LCC) im Automobilbau. Verlag Dr. Kovac, Hamburg, Germany.

Dong, Y.H., Ng, S.T., 2015. A social life cycle assessment model for building construction in Hong Kong. Int. J. Life Cycle Assess. 20, 1166–1180.

Dreyer, L.C., 2009. Inclusion of Social Aspects in Life Cycle Assessment of Products. DTU Management, Kongens Lyngby.

Du, G., Karoumi, R., 2014. Life cycle assessment framework for railway bridges: literature survey and critical issues. Struct. Infrastruct. E 10, 277–294.

Ekener-Petersen, E., Finnveden, G., 2013. Potential hotspots identified by social LCA—part 1: a case study of a laptop computer. Int. J. Life Cycle Assess. 18, 127–143.

Fatimah, Y.A., Biswas, W.K., 2016. Sustainability assessment of remanufactured computers. Procedia CIRP 40, 150–155.

Feschet, P., Macombe, C., Garrabé, M., Loeillet, D., Saez, A.R., Benhmad, F., 2013. Social impact assessment in LCA using the Preston pathway. Int. J. Life Cycle Assess. 18, 490–503.

Finnveden, G., 1997. Valuation methods within LCA—where are the values? Int. J. Life Cycle Assess. 2, 163.

Finnveden, G., Hauschild, M.Z., Ekvall, T., Guinée, J., Heijungs, R., Hellweg, S., Koehler, A., Pennington, D., Suh, S., 2009. Recent developments in life cycle assessment. J. Environ. Manage. 91, 1–21.

Foolmaun, R.K., Ramjeawon, T., 2013. Life cycle sustainability assessments (LCSA) of four disposal scenarios for used polyethylene terephthalate (PET) bottles in Mauritius. Environ. Dev. Sustain. 15, 783–806.

Funazaki, A., Taneda, K., Tahara, K., Inaba, A., 2003. Automobile life cycle assessment issues at end-of-life and recycling. JSAE Rev. 24, 386.

Garrett, P., Ronde, K., 2013. Life cycle assessment of wind power: comprehensive results from a state-of-the-art approach. Int J Life Cycle Assess. 18, 37–48.

Geibler, J.V., Walbaum, H., Liedke, C., 2006. In Development of Sustainable Bioprocesses: Modelling and Assessment [M]. John Wiley & Sons, Ltd, Chichester, UK, pp. 82–113.

Gervásio, H., Da Silva, L.S., 2013. Life-cycle social analysis of motorway bridges. Struct Infrast. Eng. 9, 1019–1039.

Granovskii, M., Dincer, I., Rosen, M.A., 2006. Life cycle assessment of hydrogen fuel cell and gasoline vehicles. Int. J. Hydrogen Energ. 31, 337–352.

Guinée, J.B., 2002. Handbook on life cycle assessment: operational guide to the ISO standards. Springer, Netherlands.

Guinée, J.B., Heijungs, R., Huppes, G., Zamagni, A., Masoni, P., Buonamici, R., Ekvall, T., Rydberg, T., 2011. Life cycle assessment: past, present, and future. Environ. Sci. Technol. 45, 90–96.

Hake, J., Koj, J.C., Kuckshinrichs, W., Schlör, H., Schreiber, A., Wulf, C., Zapp, P., Ketelaer, T., 2017. Towards a life cycle sustainability assessment of alkaline water electrolysis. Energy Procedia 105, 3403–3410.

Hanssen, O.J., 1999. Status of life cycle assessment (LCA) activities in the Nordic Region. Int. J. Life Cycle Assess. 4, 315.

Hendrickson, C.T., Lave, L.B., Matthews, H.S., 2010. Environmental Life Cycle Assessment of Goods and Services: An Input-Output Approach. Taylor & Francis.

Hertwich, E.G., McKone, T.E., Pease, W.S., 1999. Parameter uncertainty and variability in evaluative fate and exposure models. Risk Anal. 19, 1193–1204.
Hoogmartens, R., Van Passel, S., Van Acker, K., Dubois, M., 2014. Bridging the gap between LCA, LCC and CBA as sustainability assessment tools. Environ. Impact. Asses. 48, 27–33.
Hosseinijou, S.A., Mansour, S., Shirazi, M.A., 2014. Social life cycle assessment for material selection: a case study of building materials. Int. J. Life Cycle Assess. 19, 620–645.
Houillon, G., Jolliet, O., 2005. Life cycle assessment of processes for the treatment of wastewater urban sludge: energy and global warming analysis. J. Clean. Prod. 13, 287–299.
Huang, B., Mauerhofer, V., 2016. Life cycle sustainability assessment of ground source heat pump in Shanghai, China. J. Clean. Prod. 119, 207–214.
Huijbregts, M., 1998. Application of Uncertainty and Variability in LCA. Part I: A General Framework for the Analysis of Uncertainty and Variability in Life Cycle Assessment.
Hunkeler, D., 2006. Societal LCA methodology and case study (12 pp). Int. J. Life Cycle Assess. 11, 371–382.
Hunkeler, D., Rebitzer, G., 2005. The future of life cycle assessment. Int. J. Life Cycle Assess. 10, 305–308.
Hunt, R.G., Franklin, W.E., Hunt, R.G., 1996. LCA—how it came about. Int. J. Life Cycle Assess. 1, 4–7.
I.E. Agency, 2018. World Energy Outlook 2018.
Intergovernmental Panel On Climate Change, 2014. Climate Change 2013—The Physical Science Basis: Working Group I Contribution to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge.
ISO, 2006a. Environmental Management-Life Cycle Assessment-Principles and Frameworks, 14040, Genene, Switzerland.
ISO, 2006b. Environmental Management—Life Cycle Assessment—Principles and Framework., 14040-2006.
ISO, 2006c. Environmental management—life cycle assessment—requirements and guidelines. In: O.F. Standardization (Ed.) 14044, Switzerland.
Jørgensen, A., 2013. Social LCA—a way ahead? Int. J. Life Cycle Assess. 18, 296–299.
Khan, F.N., Alan, P.T.L., Zhaohui, L., Chao, L., Wai, P.K.A., 2010. Statistical analysis of optical signal-to-noise ratio monitoring using delay-tap sampling. IEEE Photonic Tech. Lett. 22, 149–151.
Khang, D.S., Tan, R.R., Uy, O.M., Promentilla, M.A.B., Tuan, P.D., Abe, N., Razon, L.F., 2017. Design of experiments for global sensitivity analysis in life cycle assessment: the case of biodiesel in Vietnam. Resour. Conserv. Recy. 119, 12–23.
Kloepffer, W., 2008. Life cycle sustainability assessment of products. Int. J. Life Cycle Assess. 13, 89.
Klöpffer, W., 2003. Life-cycle based methods for sustainable product development. Int. J. Life Cycle Assess. 8, 157.
Klöpffer, W., Ciroth, A., 2011. Is LCC relevant in a sustainability assessment? Int. J. Life Cycle Assess. 16, 99–101.
Lehmann, A., Zschieschang, E., Traverso, M., Finkbeiner, M., Schebek, L., 2013. Social aspects for sustainability assessment of technologies—challenges for social life cycle assessment (SLCA). Int. J. Life Cycle Assess. 18, 1581–1592.
Lindahl, M., Sundin, E., Sakao, T., 2014. Environmental and economic benefits of integrated product service offerings quantified with real business cases. J. Clean. Prod. 64, 288–296.
Liu, J., Ma, X., 2009. The analysis on energy and environmental impacts of microalgae-based fuel methanol in China. Energ. Policy 37, 1479–1488.
Liu, Z., Li, T., Jiang, Q., Zhang, H., 2014. Life cycle assessment–based comparative evaluation of originally manufactured and remanufactured diesel engines. J. Ind. Ecol. 18, 567–576.
Lopes, E., Dias, A., Arroja, L., Capela, I., Pereira, F., 2003. Application of life cycle assessment to the Portuguese pulp and paper industry. J. Clean. Prod. 11, 51–59.
Lundie, S., Peters, G.M., 2005. Life cycle assessment of food waste management options. J. Clean. Prod. 13, 275–286.
Luu, L.Q., Halog, A., 2016a. Chapter Fourteen—Life cycle sustainability assessment: a holistic evaluation of social, economic, and environmental impacts. In: Sustainability in the Design, Synthesis and Analysis of Chemical Engineering Processes. Butterworth-Heinemann, Oxford, pp. 327–352.
Luu, L.Q., Halog, A., 2016b. Rice husk based bioelectricity vs. coal-fired electricity: life cycle sustainability assessment case study in Vietnam. Procedia CIRP 40, 73–78.
Manik, Y., Leahy, J., Halog, A., 2013. Social life cycle assessment of palm oil biodiesel: a case study in Jambi Province of Indonesia. Int. J. Life Cycle Assess. 18, 1386–1392.

Margaret, K.M., Pamela, L.S., Kevin, R.C., 1996. Economic and life cycle assessment of an integrated biomass gasification combined cycle system. In: IECEC 96. Proceedings of the 31st Intersociety Energy Conversion Engineering Conference, vol. 3, pp. 2134–2139.

Martínez-Blanco, J., Lehmann, A., Muñoz, P., Antón, A., Traverso, M., Rieradevall, J., Finkbeiner, M., 2014. Application challenges for the social life cycle assessment of fertilizers within life cycle sustainability assessment. J. Clean. Prod. 69, 34–48.

Mathe, S., 2014. Integrating participatory approaches into social life cycle assessment: the SLCA participatory approach. Int. J. Life Cycle Assess. 19, 1506–1514.

Mihelcic, J.R., Crittenden, J.C., Small, M.J., Shonnard, D.R., Hokanson, D.R., Zhang, Q., Chen, H., Sorby, S.A., James, V.U., Sutherland, J.W., Schnoor, J.L., 2003. Sustainability science and engineering: the emergence of a new metadiscipline. Environ. Sci. Technol. 37, 5314–5324.

Ministry of Housing, 2000. Spatial Planning and the Environment. In: Eco-Indicator 99 Manual for Designers: A Damage Oriented Method for Life Cycle Assessment.

Mistry, M., Koffler, C., Wong, S., 2016. LCA and LCC of the world's longest pier: a case study on nickel-containing stainless steel rebar. Int. J. Life Cycle Assess. 21, 1637–1644.

Nemecek, T., Dubois, D., Huguenin-Elie, O., Gaillard, G., 2011. Life cycle assessment of Swiss farming systems: I. Integrated and organic farming. Agr. Syst. 104, 217–232.

Neugebauer, S., Martinez-Blanco, J., Scheumann, R., Finkbeiner, M., 2015. Enhancing the practical implementation of life cycle sustainability assessment—proposal of a Tiered approach. J. Clean. Prod. 102, 165–176.

Norris, G.A., 2001. Integrating economic analysis into LCA. Environ. Quality Manage. 10, 59–64.

Norris, G.A., 2006. Social impacts in product life cycles—towards life cycle attribute assessment. Int. J. Life Cycle Assess. 11, 97–104.

Owens, J.W., 1996. LCA impact assessment categories. Int. J. Life Cycle Assess. 1, 151–158.

Park, K., Hwang, Y., Seo, S., Seo, H., 2003. Quantitative assessment of environmental impacts on life cycle of highways. J. Constr. Eng. M 129, 31.

Pehnt, M., 2006. Dynamic life cycle assessment (LCA) of renewable energy technologies. Renew. Energ. 31, 55–71.

Pe-International, 2014.

Peng, S., Li, T., Tang, Z., Shi, J., Zhang, H., 2016. Comparative life cycle assessment of remanufacturing cleaning technologies. J. Clean. Prod. 137, 475–489.

Perugini, F., Mastellone, M.L., Arena, U., 2004. Environmental aspects of mechanical recycling of PE and PET: a life cycle assessment study. Prog. Rubber Plast. Recy. Technol. 20, 69–84.

Politano, D., Frohlich, K., 2006. Calculation of stress-dependent life cycle costs of a substation subsystem—demonstrated for controlled energization of unloaded power transformers. IEEE Tran. Power Deliver. 21, 2032–2038.

PRé-Consultants, 2014.

Rafaschieri, A., Rapaccini, M., Manfrida, G., 1999. Life cycle assessment of electricity production from poplar energy crops compared with conventional fossil fuels. Energ. Convers. Manage. 40, 1477–1493.

Rebitzer, G., Ekvall, T., Frischknecht, R., Hunkeler, D., Norris, G., Rydberg, T., Schmidt, W.P., Suh, S., Weidema, B.P., Pennington, D.W., 2004. Life cycle assessment: Part 1: framework, goal and scope definition, inventory analysis, and applications. Environ. Int. 30, 701–720.

Renó, M.L.G., Lora, E.E.S., Palacio, J.C.E., Venturini, O.J., Buchgeister, J., Almazan, O., 2011. A LCA (life cycle assessment) of the methanol production from sugarcane bagasse. Energy 36, 3716–3726.

Restianti, Y.Y., Gheewala, S.H., 2012. Life cycle assessment of gasoline in Indonesia. Int. J. Life Cycle Assess. 17, 402–408.

Robert, K.W., Parris, T.M., Leiserowitz, A.A., 2005. What is Sustainable Development? Goals, Indicators, Values, and Practice. Environ. Sci. Policy Sustain. Dev. 47, 8–21.

Ross, S., Evans, D., Webber, M., 2002. How LCA studies deal with uncertainty. Int. J. Life Cycle Assess. 7, 47–52.

Roy, P., Nei, D., Orikasa, T., Xu, Q., Okadome, H., Nakamura, N., Shiina, T., 2009. A review of life cycle assessment (LCA) on some food products. J. Food Eng. 90, 1–10.

Sala, S., Farioli, F., Zamagni, A., 2013. Life cycle sustainability assessment in the context of sustainability science progress (part 2). Int. J. Life Cycle Assess. 18, 1686–1697.

Schau, E.M., Traverso, M., Finkbeiner, M., 2012. Life cycle approach to sustainability assessment: a case study of remanufactured alternators. J. Remanuf. 2, 5.

Shi, J., Li, T., Peng, S., Liu, Z., Zhang, H., Jiang, Q., 2015. Comparative life cycle assessment of remanufactured liquefied natural gas and diesel engines in China. J. Clean. Prod. 101, 129–136.
Shiels, S., Garner, D., Clift, R., 2002. Using life-cycle assessment to inform the nuclear debate. Nucl. Energy 41, 375–381.
Spriensma, R., 1999. The Eco-Indicator 99: A Damage Oriented Method for Life Cycle Impact Assessment: Methodology Report.
Steen, B., Arvidsson, P., Nobel, A., Höjding, P., Karlson, L., Louis, S., Rydberg, T., Swan, G., Enso, S., Weiner, D., 2019. A Systematic Approach to Environmental Priority Strategies in Product Development.
Swarr, T.E., Hunkeler, D., Klöpffer, W., Pesonen, H., Ciroth, A., Brent, A.C., Pagan, R., 2011. Environmental life-cycle costing: a code of practice. Int. J. Life Cycle Assess. 16, 389–391.
Thomassen, M.A., van Calker, K.J., Smits, M.C.J., Iepema, G.L., de Boer, I.J.M., 2008. Life cycle assessment of conventional and organic milk production in the Netherlands. Agr. Syst. 96, 95–107.
Traverso, M., Asdrubali, F., Francia, A., Finkbeiner, M., 2012. Towards life cycle sustainability assessment: an implementation to photovoltaic modules. Int. J. Life Cycle Assess. 17, 1068–1079.
Tukker, A., Kleijn, R., Oers, L., Smeets, E., 1997. Combining SFA and LCA.
Upham, P., 2000. An assessment of the natural step theory of sustainability. J. Clean. Prod. 8, 445–454.
Vahdat Aboueshagh, H., Nazif, S., Shahghasemi, E., 2014. Development of an algorithm for sustainability based assessment of reservoir life cycle cost using fuzzy theory. Water Resour. Manag. 28, 5389–5409.
Valderrama, C., Granados, R., Cortina, J.L., Gasol, C.M., Guillem, M., Josa, A., 2012. Implementation of best available techniques in cement manufacturing: a life-cycle assessment study. J. Clean. Prod. 25, 60–67.
Van Genderen, E., Wildnauer, M., Santero, N., Sidi, N., 2016. A global life cycle assessment for primary zinc production. Int. J. Life Cycle Assess. 21, 1580–1593.
van Schooten, M., Vanclay, F., Slootweg, R., 2003. Conceptualizing social change processes and social impacts. In: The International Handbook of Social Impact Assessment: Conceptual and Methodological Advances. Edward Elgar Publishing, Cheltenham, UK.
van Westenenk, L., Ligthart, T., Korenromp, R., 2019. Improvement of LCA Characterization Factors and LCA Practice for Metals.
Warsen, J., Laumer, M., Momberg, W., 2011. In: Hesselbach, J., Herrmann, C. (Eds.), Comparative life cycle assessment of remanufacturing and new manufacturing of a manual transmission. Springer, Berlin, Heidelberg, pp. 67–72.
Weidema, B.P., 2006. The integration of economic and social aspects in life cycle impact assessment. Int. J. Life Cycle Assess. 11, 89–96.
Wenzel, H., Hauschild, M.Z., Alting, L., 2000. Environmental Assessment of Products: Volume 1: Methodology, Tools and Case Studies in Product Development. Springer.
Wilson, J.M., Piya, C., Shin, Y.C., Zhao, F., Ramani, K., 2014. Remanufacturing of turbine blades by laser direct deposition with its energy and environmental impact analysis. J. Clean. Prod. 80, 170–178.
Woodward, D.G., 1997. Life cycle costing—Theory, information acquisition and application. Int. J. Proj. Manag. 15, 335–344.
Xiang, D., Liu, X., Wu, Y., Wang, J., Duan, G., 2003. Life cycle assessment tool for electromechanical products green design. IEEE Int. Symp. Electr. Environ. 124.
Yang, J.X., Wang, R.S., Fu, H., Liu, J.R., 2004. Life cycle assessment of mobile phone housing. J. Environ. Sci.-China 16, 100–103.
You, F., Tao, L., Graziano, D.J., Snyder, S.W., 2012. Optimal design of sustainable cellulosic biofuel supply chains: multiobjective optimization coupled with life cycle assessment and input–output analysis. AICHE J. 58, 1157–1180.
Yue, D., Kim, M.A., You, F., 2013. Design of sustainable product systems and supply chains with life cycle optimization based on functional unit: general modeling framework, mixed-integer nonlinear programming algorithms and case study on hydrocarbon biofuels. ACS Sustain. Chem. Eng. 1, 1003–1014.
Zamagni, A., Pesonen, H., Swarr, T., 2013. From LCA to life cycle sustainability assessment: concept, practice and future directions. Int. J. Life Cycle Assess. 18, 1637–1641.
Zhang, Y., Singh, S., Bakshi, B.R., 2010. Accounting for ecosystem services in life cycle assessment, part I: a critical review. Environ. Sci. Technol. 44, 2232–2242.

CHAPTER

6

Life cycle thinking for sustainable development in the building industry

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6.1 Introduction

In various production activities, the construction and operation of buildings have an important impact on energy consumption and environmental damage (Birch, 2014). Research shows that the energy consumption of building activities accounts for more than 50% of the world's total energy consumption, and around 40% of the total energy consumption in both the United States and the European Union (Sun et al., 2017). Since the reform and opening up 40 years ago, China's economy has developed rapidly and has become the second largest in the world. With the rapid development of the economy, the proportion of the construction industry in the national economy is getting higher. China's building materials consume more than 5 billion tons of clay, limestone, and sandstone resources each year, consuming 230 million tons of standard coal energy (Xie and Yan, 2015). The production and transportation of building materials, the manufacture of building components, the construction, operation, and demolition of buildings, and other processes will bring many environmental problems. At present, building life cycle assessment (LCA) has become a common tool in the building sector to evaluate and contribute to sustainable building development (Khasreen et al., 2009a).

Regarding the environmental impact assessment of buildings materials, multiple studies have quantified energy consumption during the building production process and assessed corresponding greenhouse gas (GHG) emissions (Hong et al., 2014; Ng et al., 2013; Estokova and Porhinc ˊăk, 2012). Huang et al. (2015) and Hong et al. (2016a) used material flow analysis (MFA) models to calculate material stock of urban infrastructure materials, and predict building material stock in China from 2010 to 2050 (Huang et al., 2015; Hong et al., 2016a). Tanikawa et al. (2015) used 4D GIS technology to investigate and analyze the consumption

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of building materials and the impact of building environment in Japan. Based on the input-output life cycle assessment (IOLCA) method, Chang et al. (2016) assessed the overall energy consumption of China's construction industry and the emission of environmental pollutants such as CO_2 and NO_x. He et al. (2013) evaluated the impact of China's urban residential materialization environment in 2010 based on the LCA method, and used the scenario analysis method to explore the environmental emissions of urban residential buildings in China in 2020.

6.2 Life cycle of building materials

Based on previous studies, buildings may be classified as residential or nonresidential in accordance with the Chinese statistical yearbooks (NBSC (National Bureau of Statistics of China), 2001-2016). There are seven types of nonresidential buildings: office, education and cultural, research, plant and warehouse, commercial, healthcare and medicine, and other buildings. The key building material categories are identified as steel, concrete, cement (for nonconcrete uses, i.e., plaster and mortar), wood, brick, sand (nonconcrete use), gravel (nonconcrete use), limestone, glass, and ceramic tiles (Hong et al., 2016b; Huang et al., 2017a).

The whole life cycle of product is always a dynamic and comprehensive process. The life cycle of a building includes the acquisition of raw materials, the processing and manufacturing of building materials, the production of building components, the construction of buildings, the use of operations, and the entire process of demolition. As shown in Fig. 6.1. As for the building materials, the main life cycle processes are acquisition and processing of raw materials, processing of building components, and disposal after demolition.

6.3 Green building materials

The concept of "green materials" was first proposed by the international society for materials science in 1988. Yamamoto put forward the concept of "eco-environmental materials" in 1990 (Wang and Bao, 2015). In 1999, the first Chinese national green building materials

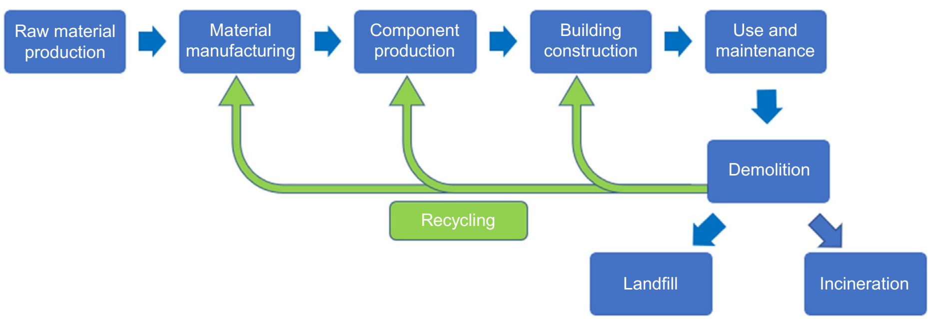

FIG. 6.1 Life cycle processes of buildings.

development and application conference put forward the definition of green building materials, pointing out that these materials should adopt cleaner production technology. Compared with traditional building materials, green building materials should be cleaner in the production, construction, use, and dismantling processes.

Green building materials can vary in types and functions. However, in the selection of green building materials, the characteristics and construction environment of buildings should be fully considered. In the process of traditional building construction, a large amount of dust will be generated, causing serious environmental pollution problems, and its input cost is large. Compared with traditional building materials, green building materials have obvious advantages in terms of service performance and economic benefits.

Nowadays, resources are increasingly scarce and sustainable development has strategic significance. Green building materials are in line with the strategic planning for sustainable development (Chen, 2019). In order to enhance people's awareness of building environmental protection, China has been increasing the implementation of green building projects, the purpose of which is to encourage residents to actively use environmentally friendly building materials, to build a harmonious green home. In addition, China also actively encourages relevant personnel to continuously develop more environment-friendly building materials, and continuously increase environmental protection efforts to promote the healthy development of China's environment (Cheng and Li, 2019).

At present, environmentally friendly building materials have been recognized by more and more residents. The development of environment-friendly building materials plays a great role in actively advocating environmental protection for the country.

6.3.1 Life cycle assessment of building materials

6.3.1.1 Methodology

Life cycle assessment is an important application of life cycle thinking in the field of environmental protection. It is used to explore the energy consumption of buildings and their environmental impacts. There is a relevant definition of life cycle assessment in the world, pointing out that it is a method of objectively evaluating the environmental load of a product, activity, or process. According to the LCA methodology framework proposed by the International Society of Environmental Toxicology and Chemistry (SETAC) in 1993, its basic structure is divided into four organic parts: the setting of evaluation purpose and investigation scope (ISO14041), inventory analysis (ISO14041), life cycle impact assessment (ISO14042), and result analysis (ISO14043) (ISO, 2006).

Surveys are usually carried out to collect necessary data and information for life cycle assessment. The scope of a building's survey includes building functions, building area, year of assessment, building structure, recycling of building materials, and use of construction equipment. The survey of basic building data also includes energy consumption for construction equipment and a list of released gases. Based on life cycle thinking, inventory analysis is a process of data collection for environmental load of the system based on material balance and energy balance. The analysis of building materials inventory should be collected from seven phases, including design, construction, replacement, energy consumption, maintenance and management, repair and renewal, and disposal phase (Xiong and Deng, 2016). Impact assessment is a process of categorizing environmental impacts and characterization and quantifying their potential environmental impacts based on inventory analysis. The resource input, renew cycle, renew times, energy consumption, and material recycling rate per unit area of each stage shall be displayed in detail to provide data support for subsequent environmental impact assessment.

The environmental impact assessment of buildings mainly combines the recycling of resources, and analyzes the input of raw resources, the amount of recycled resources, the amount of waste materials and waste generated, and the final treatment volume at each stage. There are usually three steps to evaluate the environmental impact burden of building materials: (1) classify buildings and building materials into types; (2) calculate the annual building material use; and (3) estimate environmental impacts associated with building materials production (Huang et al., 2018a). Furthermore, resource life cycle reduction (LCR) and material life cycle waste (LCW) evaluation are the final step of life cycle assessment for building materials. The evaluation method is as follows: comparing the results of the building environmental impact with the benchmark case, comprehensively analyzing the difference between the evaluation case and the reference case in the materials input, waste generation and other items, thereby calculating the corresponding reduction rate or increase rate (Yi and De, 2005).

In our previous research (Huang et al., 2018a), we have tried to answer the following research questions: (1) What kinds, how much, and in which building types have building materials been used in recent years in China, and what are their development trends? (2) What are the primary environmental impacts caused by producing these building materials? (3) What is the spatial distribution of the key embodied environmental impacts? In addition to the nationwide time series of 16 years, we also explored the variation of material use and associated environmental impact across different provinces of Mainland China. The research method uses the three steps mentioned above: (1) classify buildings and building materials into types; (2) calculate the annual building material use from 2000 to 2015; and (3) estimate environmental impacts associated with building materials production.

Environmental impacts per kg of building material (*Ek*) are evaluated by applying the midpoint parameters of the ReCiPe 2016 H method, which is a commonly used life cycle impact assessment method with the most up to date environmental impact indicators and normalization values (Huijbregts et al., 2016). Moreover, the ReCiPe method covers China with its global scope impact mechanism.

In addition to the evaluation of environmental impact per kg of building material (*Ek*), we also should analyze environmental impact considering the annual material use amount (*Ev*). This is important because materials are used in different amounts for specific buildings, and it is quite possible that materials with higher per-mass environmental burden are consumed at lower rates and vice versa. *Ev* was calculated according to:

$$Ev_{i,x}^{t} = MU_{i}^{t} \times Ek_{i,x}$$

where $Ev_{i,x}^{t}$ is the total magnitude of environmental impact x formaterial i in year t from all construction, summed for the eight building types j; $Ek_{i,j,x}$ is the per-kg environmental impact x of material i in building type j; and $MU_{i}^{t,k}$ is the annual use of material i of year t in province k.

We carried out environmental impact characterization and normalization on the midpoint level (Strauss et al., 2006; Bueno et al., 2016), to compare the contribution of building materials to the total global impacts in different impact categories. In normalization, the characterized results of each impact category are divided by a selected reference value, which brings all the results to the same scale. Such normalization facilitates the interpretation of the results and helps us link the relative contributions of each building material to each type of environmental impact.

6.3.1.2 Results

Existing researches show that around the world, 20% of the building life cycle energy consumption and environmental impacts are from the building materials (Adalberth, 1997). The environmental impact of building materialization on the environment includes 15 environmental impact categories such as climate change, surface acidification, the formation of photochemical oxidants, particulates, ozone depletion, ionizing radiation, eutrophication of fresh water, ocean eutrophication, human toxicity, freshwater, marine ecological toxicity, land ecological toxicity, fossil fuel consumption, and the loss of metal (Huang et al., 2017b). The main impacts of the built environment are water consumption, metal pollution, and global warming (Minho et al., 2015; Li et al., 2016).

In our analysis targeting China as a case study, we found that 2 billion tons of building materials were used in China in 2000; this increased to 10 billion tons by 2014 (Fig. 6.2). The key building materials were concrete, sand, and gravel, followed by bricks, and cement for nonconcrete applications. Steel, limestone, and wood were used in relatively lower quantities.

Our environmental impact assessment result show that steel, lime, glass, wood, and cement have comparatively higher environmental impacts per kg (*Ek*) than the other materials. Scaling up the ReCiPe environmental impacts from 1 kg to annual use amounts (*Ev*), we aggregate the contribution of each material to every impact category. The environmental impact indicators associated with the production of building material used in China in 2015

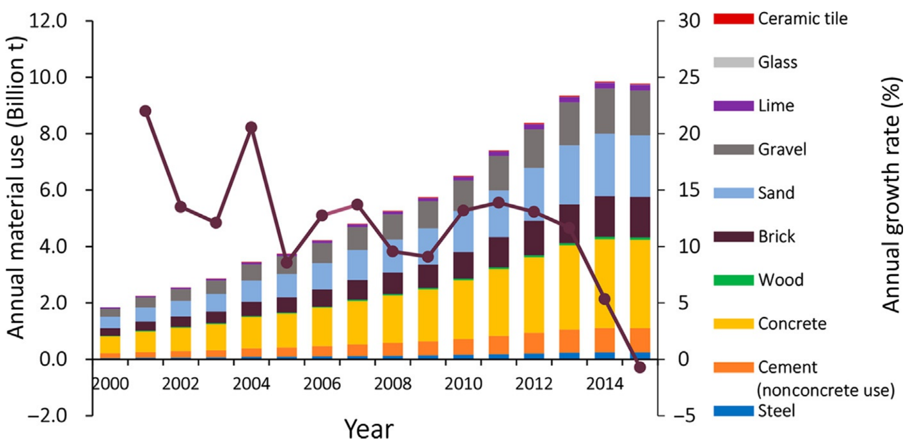

FIG. 6.2 Annual use of building materials for newly constructed buildings, 2000–15 (*bar plot*, left-hand axis) and annual growth rate of building material use (*line plot*, right-hand axis) (Huang et al., 2018a).

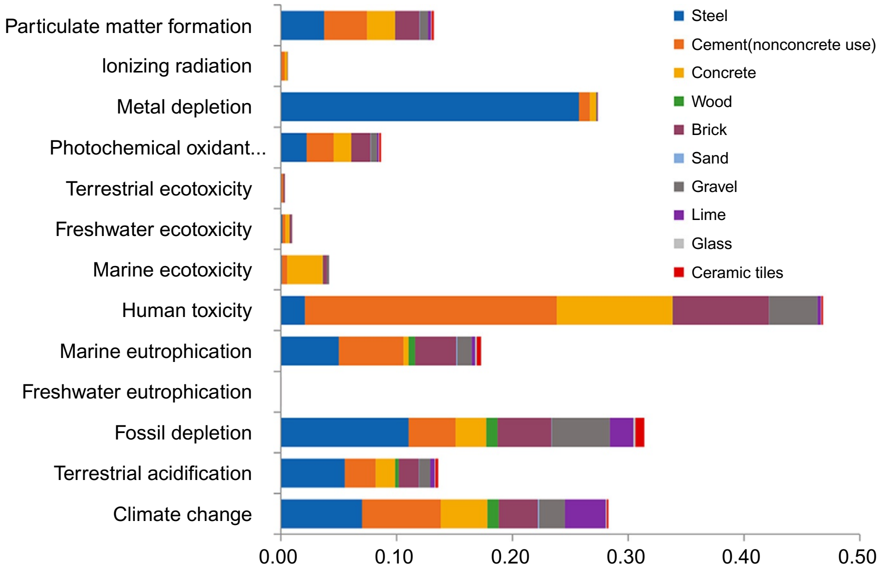

FIG. 6.3 Environmental impact indicators associated with the production of building material used in 2015, using the ReCiPe method, normalized to global indicators in 2000 (*Nv*, cf. Eq. 2) (Huang et al., 2018a).

are illustrated in Fig. 6.3 (the 13 highest of the 18 environmental indicators are presented). Overall, the most severe environmental impacts are found to be human toxicity, fossil fuel depletion, global warming, and metal depletion, emphasizing that greenhouse gas emissions should not be the sole focus of research on environmental impacts of building materials.

In general, cement, steel, concrete, and bricks are the key contributors to the environmental impacts of building materials. The contributions of some materials are due to their high use (e.g., concrete, sand, gravel, and brick). Other materials have disproportionate contribution to various impacts despite their comparatively low use by mass (cf. Fig. 6.2). Steel is the most prominent example, but also lime, glass, and wood. Cement stands out as a material whose high contribution to impacts is a combination of both high usage and high impacts per kg.

Tracing the sources of these key environmental indicators, human toxicity is primarily caused by heavy metals (including arsenic, cadmium, zinc, lead, etc. (Huijbregts et al., 2000)) emitted in the mining and manufacturing processes of cement, concrete, and bricks. Fossil depletion is mainly caused by the large demand for coal, petroleum, electricity, and natural gas in the manufacturing process of steel, brick, gravel, and cement. The largest contributions to global warming come from steel and cement production and each account for around 25% of total impact from building materials (Fig. 6.4).

Global warming burdens originate in the large energy consumption during the production processes of steel, cement, and concrete (Guo et al., 2016; You et al., 2011; Dodoo et al., 2009) and in the chemical reactions of clinker production for cement manufacture (Yücel, 2013). Thus, reducing the energy use and using less CO_2-intensive energy sources in steel and lime production are presumably the most effective approaches. Whereas in the case of concrete, gravel, and bricks, the focus should be on reducing consumption or looking for substitute

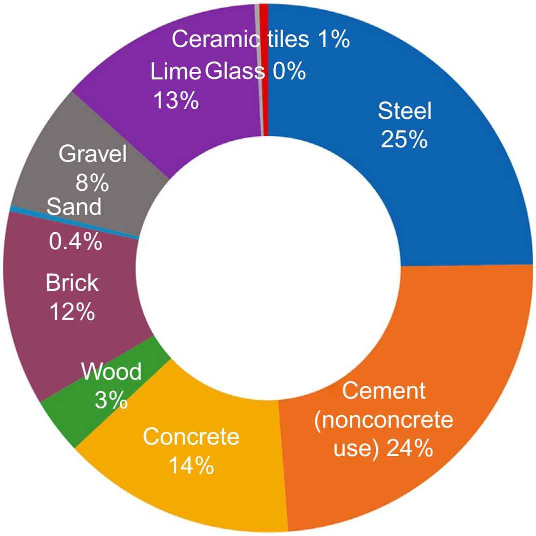

FIG. 6.4 Share of global warming impacts from building material use in China in 2015 (Huang et al., 2018a).

materials with lower GHG burden such as hollow concrete blocks, stabilized soil blocks, or fly ash (Huberman and Pearlmutter, 2008).

Previous studies often analyzed the impacts of individual building materials or the magnitudes of consumption, but rarely combined both. The research results of our research group indicate that building materials with high environmental impacts per kg in China are steel, lime, glass, wood, and cement, consistent with international studies (Thormark, 2006; Khasreen et al., 2009b; Chau et al., 2015). Our research approach enables to identify the contribution of specific impact and to see whether it is from a material's per-unit associated impact or from the magnitude of usage. Our results indicate that although steel, cement, and concrete are key contributing materials for the estimated impacts and have similar magnitudes of *Ev*, the origin of each impact is different and thus also the potential measures to reduce impacts. Concrete's impact per kg (*Ek*) is relatively low and the magnitude of impacts is mostly from the sheer amount used. In comparison, steel's high impacts are due to its high per-kg impacts rather than the masses used. Cement's contributions to impacts are a combination of both the scale of use and the per-kg associated impacts.

Based on the findings and the fact that GHG emission burdens are currently the only environmental indicator for evaluating green building material products in the current certification program, other key environmental indicators such as human toxicity and fossil depletion are highly recommended to be included in the certification system. Policy strategies such as green building materials certification programs should be given attention and reinforced to promote the cleaner production for building materials, since currently the program covers only concrete, glass, and ceramic tile, and is in the initial stage of implementation in China (MIIT, n.d.). Extended producer responsibility may also be an option for the high recycling potential materials such as concrete, steel, and wood (Guggemos and Horvath, 2003). Holding building material producers responsible for managing certain building waste encourages manufacturers to design more environmentally friendly and recyclable materials.

6.3.2 Application of green building materials

The selection of materials for green buildings not only needs to consider the resource consumption and environmental emissions in the process related to material production, but also needs to consider the performance of materials in the specific building structure, both of which are indispensable.

At present, a variety of green building materials have been developed globally, and these have been widely used in the application process. Insulating glass is a new kind of green building material invented in the United States. It not only has the characteristics of adequate quality, low energy consumption, but also has the advantages of high intensity and high density. In practical green building applications, the use of insulating glass not only meets the actual requirements of green building materials, but also plays a very good role in promoting green city construction (Tang, 2016).

Domestic study found that, the annual average environmental impact on building windows, recycled wood-plastic composite windows, and wood-plastic composite windows is lower—about 23% lower than aluminum alloy windows. Using green materials to reduce the heat transfer coefficient of building windows is an effective way to improve the environmental protection performance of building windows in its whole life cycle (Liu and Zhang, 2016).

The materials for door and window design in China have gradually emerged as new energy-saving materials such as unplasticized polyvinyl chloride (UPVC) plastics and aluminum alloy heat-insulation; this not only improves the thermal insulation performance of doors and windows, but also improves the sealing performance (Tang, 2014). For building wall materials, with the same insulation performance, the total environmental impact of aerated concrete blocks is the least, and that of solid clay bricks is the largest, which is more than three times that of aerated concrete blocks (Liu and Zhang, 2016). Green wallboard can be made by processing and compressing renewable raw materials. The use of green wallboard not only reduces air pollution, but also reuses waste products, which greatly saves costs and reduces pollution. Common green wallboard raw materials include straw, precast concrete, etc. (Chen, 2018).

The life cycle evaluation method can scientifically and comprehensively evaluate the environmental impact of the whole life cycle of building materials, and determine which building materials can obtain the best environmental benefits under the premise of realizing the same function. The environmental impact analysis of energy-saving building materials based on the life cycle assessment method not only contributes to the development of new building materials, but also provides a powerful method and data support for the selection of green building materials.

6.4 Recycling of construction and demolition waste

Construction and demolition wastes (CDW) are the status of building materials after the end life of buildings. CDW could be concrete, steel, wood products, asphalt shingles, and bricks from building. Old buildings approaching the end of their lives are demolished, producing millions of tons of concrete wastes, large quantities of construction products rejected for noncompliance with the required specifications, as well as marble deposits provided a large quantity of aggregates of different sizes induced by fragmentation operations, sawing of large stones; furthermore, processing plants proliferate a very large quantity of wastes consisting mainly of powders and sludge (Belachia and Hebhoub, 2018). Most of the CDW comes from the demolition process while minor portions (around 10%–30%) are generated during the construction process (EPA, 2018; Ning, 2017). Reducing, reusing, and recycling of CDW has become an urgent and essential issue, as inappropriate CDW treatment will cause severe environmental issues and land use threats.

Countries around the world reduce CDW by introducing different legislation and raising awareness. Japan, Singapore, and some European countries are at the forefront in the treatment and reuse of construction waste. In Japan, there are more than 20 subdivisions of "construction by-products," which are scientifically processed according to categories. The main principle of treating CDW in Japan is to reduce the generation of waste on the construction site and reuse it as much as possible. Singapore focuses on setting standards for green buildings to reduce the generation of construction waste from the source. According to the European Union statistics office, the total amount of waste generated in the European Union was over 2.5 billion tons, of which almost 860 million tons belonged to construction and demolition activities (Bravo et al., 2015). Some European countries have achieved the goal of 70% CDW recycling. Statistics show that the total mass flow of recovered waste accounts for more than 80% of the total waste generation in member states such as the Netherlands, Germany, and Denmark (Eurostat, 2017).

Current CDW processing and recycling techniques can be considered to be common across Europe. A common CDW recycling plant usually consists of:

(1) reception, weighing, and visual inspection;
(2) manual preselection (for unsegregated streams), rejection, and diversion to alternative treatments;
(3) screening of large materials;
(4) magnetic separation;
(5) manual separation of plastic, wood, and other waste streams, if required;
(6) crushing; and
(7) screening and secondary crushing, which is applied depending on the goal product mix (Gálvez-Martos et al., 2018).

However, in some regions there is a significant amount of illegal dumping and a heterogeneous market for secondary materials, which hinders the development of the secondary materials market.

In order to achieve zero landfill for CDW in the United Kingdom, the construction industry in the United Kingdom usually begins to estimate the total life cycle of waste production at the design stage and gives a plan for recycling construction waste. The United Kingdom enacted specific construction waste management regulations to record the production and type of construction waste on the construction site in order to achieve the purpose of recycling construction and demolition waste. The UK Government sponsored the Waste Resources Action Program, which led to a variety of work undertaken to increase recycling for CDW. This included working with the construction sector to specify higher levels of recycled content, a major program of assisting companies to reduce CDW to landfill and assisting in financing recycling plants. At the same time, the United Kingdom has adopted a landfill tax on construction waste, and the annual landfill tax has gradually increased, which has driven the development of CDW recycling technology in the United Kingdom.

In Australia, the largest components of the CDW stream, and the most commonly recycled materials in Australia are concrete, bricks, asphalt, soil, timber, and ferrous metals, because they are usually demolished in large quantities and have an existing market for reuse and recycling (e.g., concrete, bricks, and asphalt), or they have a relatively high commercial value (e.g., metals) (EPHC, 2010). At the same time, Australia has also enacted some laws and regulations to strengthen the management and recycling of construction waste.

Japan was the first country in the Asian region to formulate regulations on construction waste. Through continuous supplementation and improvement, it has formed a legal system that is in line with its national conditions. Meanwhile, Japan has established a recycling system for construction waste, developed classification and treatment technologies, and implemented a zero emission strategy for construction waste, greatly promoting the recovery of construction waste, with the recovery rate increasing from 42% in 1995 to 97% in 2011 (Pu and Tang, 2012). In Japan, the utilization rate of waste concrete blocks is high, and it is generally used as aggregate for asphalt concrete after crushing and separation. For discarded wood, according to its quality, it can be used as papermaking raw materials, hot pressing plates, fuels, etc. About 25% of the sludge produced goes to sanitary landfills, 65% incinerated, and only about 9% is used for agriculture (Zhang and Sun, 2018). For waste plastics, except for a small part of recycling, the rest are incinerated.

The United States is also one of the countries with the highest construction waste production. In the United States, 30% of CDW will be transported to landfills for landfill. In an EPA report, there is a quick guide to reducing waste from building demolition, allowing construction waste to be sorted out and recycled. There are also some promotion policies for the recycling of construction waste in the United States, such as:

(1) the government has clearly stipulated the requirements for recycling of construction waste, including the requirements and proportions of material recycling, the specific requirements of green buildings, etc.;
(2) establishing market incentives, including tax relief, recycling subsidies, tax rebates, etc.; and
(3) providing education to construction companies and the general public to improve awareness of construction waste recycling.

Most states and local governments in the United States encourage enterprises and the public to recycle construction waste. There are also national projects; for example, the US Environmental Protection Agency, have created a web page for solid waste disposal to provide information on the recycling of construction waste in demolition, renovation, and new construction projects to stakeholders.

Currently, China's recycling of construction waste is based mainly on inert construction waste. According to the Ministry of Construction in 2003, "urban construction garbage and waste residue management regulations (revised)" regulation, according to the source classification, CDW can be divided into land excavation, the excavation of roads, old building demolition, construction and building materials production, mainly using sediment, crushed stone, waste mortar, brick and tile fragments, concrete, asphalt, plastic, scrap metal, waste wood, etc. Different structures and building construction types generate waste of which the components are different; the basic composition is classified, mainly by the soil, sediment, scattered mortar, concrete, carved masonry, and reinforced concrete pile under the concrete debris heading, piling, scrap metal, waste from bamboo timber, decoration, all kinds of packaging materials, and other wastes, etc.

In recent years, China's annual CDW emissions are about 1.55 billion tons to 2.4 billion tons, accounting for about 30%–40% of urban waste, causing a serious ecological crisis. For a long time, due to the lack of unified and perfect CDW management methods, and the lack of scientific, effective, economic, and feasible disposal technology, the vast majority of CDW without any treatment will be shipped to the suburbs for open stacking or simple landfill. In 2017, CDW generated in China was about 2.379 billion tons, among which only 119,000 tons were recycled. Recycled aggregates are mainly processed from discarded concrete, mortar, bricks, etc.

The average recovery rate of construction waste in China is around 5%. Based on the literature review and survey, challenges of CDW management in China were analyzed by interviews with relevant stakeholders, including researchers, building designers, construction and demolition company staff, and CDW treatment/recycling company managers (in total, 40 people). We explored the problems of construction waste management in China based on the 3R principle of circular economy, and summarized some of the following existing problems of building recycling in China (Huang et al., 2018b).

- Barriers for *Reducing*
  Lack of design standards for reducing CDW
  Low cost for CDW disposal
  Inappropriate urban planning
- Barriers for *Reusing*
  Informal collection
  Lack of guidance for effective CDW collection and sorting
  Lack of standards for reused CDW
- Barriers to *Recycling*
  Ineffective government regulation
  Immature recycling technology
  Lack of standards for recycled CDW products

Based on these findings, suggestions to promote CDW management based on 3R principle were proposed. Firstly, effective circular economy models in building and other related industries should be designed. In order to enforce reduction of CDW, it is necessary to reinforce the source control. For example, building design and construction stakeholders should sign an agreement to develop green construction programs in which they jointly manage the CDW. Enhanced supervision and management is in urgent necessary for implementing reuse and recycling of CDW. Approaches of this aspect include establishing a coherent "top-down" regulatory system, carrying out process monitoring of CDW, and implementing strict punishment for illegal CDW treatment behaviors. Innovative technologies are also essential for promoting circular economy of CDW. Other than promoting technologies of classification and separating of CDW, contributions of joint technologies such as precast construction and BIM should also be explored. Last but not least, government should encourage economic

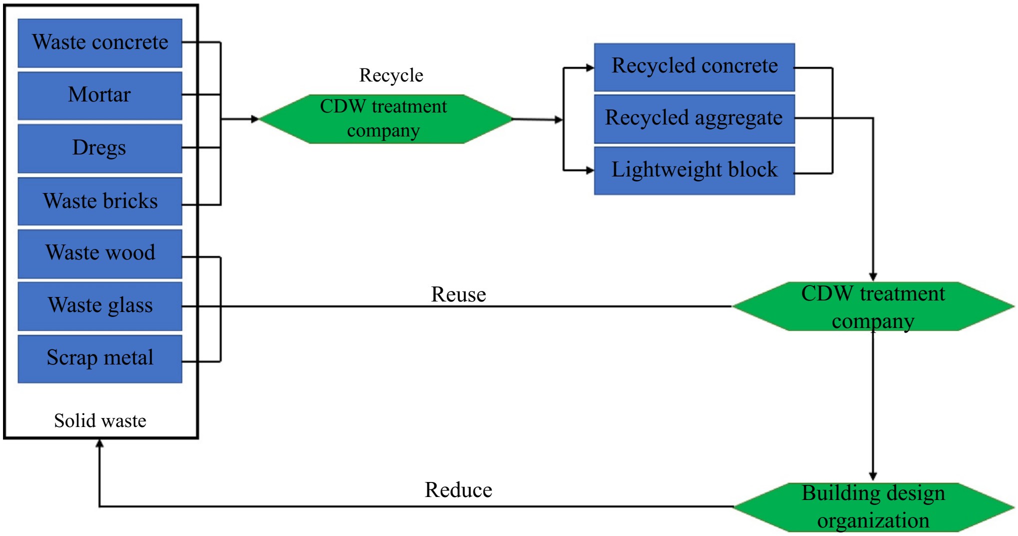

FIG. 6.5 Circular economy mode for construction and demolition waste in the building industry.

incentive measures such as shortening the application period for loan and lowering rent of land for CDW management businesses. New market modes such as the public-private-partnership should also be encouraged for relieving the economic pressures for CDW treatment/recycling companies.

It is important to establish an effective circular economy model for CDW, because recycled CDW can be utilized not only for the building industry, but also for other industries. As shown in Fig. 6.5, reducing and reusing CDW should be carried out by stakeholders and professionals in building design and construction. For recycling and reusing CDW within the building industry, waste concrete, bricks, dregs, and mortar can be converted into recycled materials, such as recycled concrete, lightweight block, and recycled aggregate.

As a global issue, appropriate CDW treatment and reduce, reuse, recycle approaches are essential for each country. Reasonable CDW treatment based on the 3R principle can enhance the efficiency for building materials in their life cycle, and accordingly contribute to enhancing the efficiency of buildings.

References

Adalberth, K., 1997. Energy use during the life cycle of single-unit dwellings: examples. Build. Environ. 32 (4), 321–329.

Belachia, M., Hebhoub, H., 2018. Valorization of marble waste in the manufacture of concrete. IOP Conf. Ser. Mater. Sci. Eng. 431. https://doi.org/10.1088/1757-899X/431/3/032005.

Birch, E.L., 2014. A review of "climate change 2014: impacts, adaptation, and vulnerability" and "climate change 2014: mitigation of climate change". J. Am. Plan. Assoc. 80 (2), 184–185.

Bravo, M., De Brito, J., Pontes, J., Evangelista, L., 2015. Mechanical performance of concrete made with aggregates from construction and demolition waste recycling plants. J. Clean. Prod. 99, 59–74.

Bueno, C., Hauschild, M.Z., Rossignolo, J.A., Ometto, A.R., Mendes, N.C., 2016. Sensitivity analysis of the use of life cycle impact assessment methods: a case study on building materials. J. Clean. Prod. 112, 2208–2220.

Chang, Y., Huang, Z.Y., Ries, R.J., et al., 2016. The embodied air pollutant emissions and water footprints of buildings in China: a quantification using disaggregated input-output life cycle inventory model. J. Clean. Prod. 113, 274–284.

Chau, C.K., Leung, T.M., Ng, W.Y., 2015. A review on life cycle assessment, life cycle energy assessment and life cycle carbon emissions assessment on buildings. Appl. Energy 143 (1), 395–413.

Chen, Q., 2018. Importance and feasibility analysis of the application of modern building environmental protection materials. Green Environ. Protect. Build. Mater. (1), 6 (in Chinese).

Chen, Q., 2019. Application of green building materials in civil engineering construction. Ju she (13), 36 (in Chinese).

Cheng, S.J., Li, W., 2019. Characteristics and application development trend of environmentally friendly building energy-saving materials. Shandong Ind. Technol. (15), 91 (in Chinese).

Dodoo, A., Gustavsson, L., Sathre, R., 2009. Carbon implications of end-of-life management of building materials. Resour. Conserv. Recycl. 53 (5), 276–286.

EPA, 2018. Advancing Sustainable Materials Management: 2015 Fact Sheet. US EPA, Washington.

EPHC, 2010. National Waste Report. Department of the Environment, Water, Heritage and the Arts.

Estokova, A., Porhinc ́ăk, M., 2012. Reduction of primary energy and ́ CO2 emissions through selection and environmental evaluation of building materials. Theor. Found. Chem. Eng. 46, 704–712.

Eurostat, 2017. Generation of Waste by Waste Category. Hazardousness and NACE Rev 2 Activity.

Gálvez-Martos, J.-L., Styles, D., Schoenberger, H., 2018. Construction and demolition waste best management practice in Europe. Resour. Conserv. Recy. 136, 166–178.

Guggemos, A.A., Horvath, A., 2003. Strategies of extended producer responsibility for buildings. J. Infrastruct. Syst. 9 (2), 65.

Guo, B., Geng, Y., Dong, H., Liu, Y., 2016. Energy-related greenhouse gas emission features in China's energy supply region: the case of Xinjiang. Renew. Sustainable Energy Rev. 54 (2047), 15–24 (in Chinese).

He, X.S., Liu, Y., Li, T.W., et al., 2013. Does the rapid development of Chinas urban residential buildings matter for the environment. Build. Environ. 64, 130–137.

Hong, T., Ji, C., Jang, M., Park, H., 2014. Assessment model for energy consumption and greenhouse gas emissions during the construction phase. J. Manage. Eng. 30, 226–235.

Hong, L.X., Zhou, N., Feng, W., et al., 2016a. Building stock dynamics and its impacts on materials and energy demand in China. Energy Pol. 94. 47e55. (in Chinese).

Hong, L., Zhou, N., Feng, W., Khanna, N., Fridley, D., Zhao, Y., Sandholt, K., 2016b. Building stock dynamics and its impacts on materials and energy demand in China. Energy Policy 94, 47–55 (in Chinese).

Huang, C., Han, J., Chen, W.Q., 2015. Changing patterns and determinants of infrastructures' material stocks in Chinese cities. Resour. Conserv. Recycl. 123, 47e53 (in Chinese).

Huang, C., Han, J., Chen, W.Q., 2017a. Changing patterns and determinants of infrastructures' material stocks in Chinese cities. Resour. Conserv. Recycl. 123, 47–53 (in Chinese).

Huang, B.J., et al., 2017b. Implicit environmental impact assessment of building materials. Res. Environ. Sci. 30 (6), 929–936.

Huang, B.J., Zhao, F., Fishman, T., Chen, W.Q., Heeren, N., Hertwich, E.G., 2018a. Building material use and associated environmental impacts in China 2000-2015. Environ. Sci. Technol. https://doi.org/10.1021/acs.est.8b04104.

Huang, B.J., Wang, X.Y., Harnwei, K., et al., 2018b. Construction and demolition waste management in China through the 3R principle. Resour. Conserv. Recy. 129, 36–44.

Huberman, N., Pearlmutter, D., 2008. A life-cycle energy analysis of building materials in the Negev desert. Energy Build. 40, 837–848.

Huijbregts, M.A.J., Thissen, U., Guinee, J.B., Jager, T., Kalf, D., 2000. Chemosphere priority assessment of toxic substances in life cycleassessment. Part I: calculation of toxicity potentials for 181 substances with the nested multimedia fate, exposure and effects model USESLCA. Chemosphere 41, 541–573.

Huijbregts, M.A.J., Steinmann, Z.J.N., Elshout, P.M.F., Stam, G., Verones, F., Vieira, M.D.M., Hollander, A., Van Zelm, R., 2016. ReCiPe2016: A harmonized life cycle impact assessment method at midpoint and endpoint level. RIVM Report 2016-0104. Bilthoven, The Netherlands.

ISO, 2006. Environmental Management. Life Cycle Assessment. Principles and Framework (ISO 10040:2006).

Khasreen, M.M., Banfill, P.F.G., Menzies, G.F., 2009a. Life-cycle assessment and the environmental impact of buildings: a review. Sustainability 1, 674–701. https://doi.org/10.3390/su1030674.

Khasreen, M.M., Banfill, P.F.G., Menzies, G.F., 2009b. Life-cycle assessment and the environmental impact of buildings: a review. Sustainability 1 (3), 534–544.

Li, D.Z., Cui, P., Lu, Y.J., 2016. Development of an automated estimator of life-cycle carbon emissions for residential buildings: a case study in Nanjing, China. Habitat Int. 57, 154–163 (in Chinese).

Liu, Y., Zhang, Y.F., 2016. Research on green building materials selection based on life cycle evaluation. Prog. Chinese Mater. 35 (10). 769–775+790. (in Chinese).

MIIT, Instruction for Promoting Green Building Material Standard, Certification and Labeling. http://www.miit.gov.cn/n1146285//content.html. (Accessed 15 October 2018).

Minho, J., Twaheon, H., Changyoon, J., 2015. Hybrid LCA model for assessing the embodied environmental impacts of buildings in South Korea. Environ. Impact Assess. Rev. 50, 143–155.

NBSC (National Bureau of Statistics of China), 2001-2016. China Statistical Yearbook on Construction. China Statistics Press, Beijing (in Chinese).

Ng, S.T., Chen, Y., Wong, J.M.W., 2013. Variability of building environmental assessment tools on evaluating carbon emissions. Environ. Impact Assess. 38, 131–141.

Ning, J., 2017. China Statistical Yearbook-2017. China Statistics Press, Beijing.

Pu, Y.H., Tang, J.L., 2012. The enlightenment of recycling construction waste in Japan to China [J]. Construct. Technol. 21, 43–45 (in Chinese).

Strauss, K., Brent, A., Hietkamp, S., 2006. Characterisation and normalisation factors for life cycle impact assessment mined abiotic resources categories in South Africa: the manufacturing of catalytic converter exhaust systems as a case study. Int. J. Life Cycle Assess. 11, 162–171.

Sun, C.J., Zhou, G., Ge, Y.J., 2017. Application research of BIM technology in the whole life cycle of sustainable green building. Proj. Manag. Technol. (2) (in Chinese).

Tang, L.B., 2014. Application analysis of green building materials in building energy conservation. Shanghai Build. Mater. (2), 37–38 (in Chinese).

Tang, 2016. Application of new energy-saving materials for green buildings in modern urban development. USA & Times (4), 34–35.

Tanikawa, H., Fishman, T., Okuoka, K., et al., 2015. The weight of society over time and space-a comprehensive account of the construction material stock of Japan, 1945-2010. J. Ind. Ecol. 19, 778–791.

Thormark, C., 2006. The effect of materials choice on the total energy need and recycling potential of a building. Build. Environ. 41, 1019–1026.

Wang, J., Bao, Y.Y., 2015. Review the use of ecological environment materials. Technol. Innov. Appl. 5, 23–26.

Xie, H.T., Yan, Y.F., 2015. Application analysis of green building materials and building energy conservation in engineering. SME Manag. Technol. (late issue) (9), 118.

Xiong, W., Deng, J.Y., 2016. Study on the life cycle assessment method of Japanese green buildings. Const. Econ. 37 (02), 95–99.

Yi, P., De, H., 2005. Methods of Life Cycle Environmental Impact Assessment. Association of Industrial environmental management, pp. 27–256.

You, P., Hu, D., Zhang, H., Guo, Z., Zhao, Y., Wang, B., Yuan, Y., 2011. Carbon emissions in the system in China-a case study of residential life cycle of urban building system in China-a case study of residential buildings. Ecol. Complex. 8 (2), 201–212 (in Chinese).

Yücel, G., 2013. Extent of inertia caused by the existing building stock against an energy transition in the Netherlands. Energy Build. 56, 134–145.

Zhang, C., Sun, L.N., 2018. Status quo of sludge treatment and utilization and development prospects [J]. Heilongjiang Agri. Sci. 09, 158–161 (in Chinese).

CHAPTER

7

MCDM for sustainability ranking of district heating systems considering uncertainties

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7.1 Introduction

A district heating (DH) system is one of the most important components of city infrastructures in cold climate areas. In China, the DH area expanded from about 250 million square meters to more than 2600 million square meters between 1991 and 2006 (Huang, 2011). In the past 10 years, since China has been undergoing a fast urbanization, the total DH demand has grown even faster. Combined heat and power (CHP), heat only boiler (HOB), and heat pumps (HPs) are the most popular heat sources in China to satisfy the huge amount of heat demand. According to Tsinghua University Building Energy Research Center (2014), coal-fired CHP supplies approximately 48% of the district heat, followed by the coal-fired heat only boiler that produces about 42% of district heat. Gas-fired heat only boiler takes the third place with a contribution of nearly 8%; the share from other heating technologies including heat pump is very small—less than 2%. In China, CHP plants are mainly coal-based and almost a half of the total heat is produced by coal-fired HOB. This is mainly because coal is the primary fuel for DH due to the connatural energy structure (Lin, 2002). More and more DH systems will be built in the near future, and the DH area will be gradually expanded from north China to south China, specifically from the extremely cold area and cold area to the hot summer and cold winter area.

Different DH technologies have different characteristics from the economics, environment, and energy (3E) points of view, which can reflect the sustainabilities of them; therefore

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choosing the most suitable DH system for a specific area is very important to decision makers (DMs) and managers (Cao, 2002). The evaluation of DH systems is not a single objective problem; on the contrary, it is a typical multicriteria decision making (MCDM) problem, and the use of MCDM method in heating ventilating and air conditioning (HVAC) systems are of more importance (Jiang et al., 2007). It can help the DMs make more consistent decisions by considering all important factors, which often include conflicting criteria and usually have some uncertainties (Troldborg et al., 2014; Shan et al., 2013; Gang et al., 2015).

In this chapter, seven popular DH systems (Wei et al., 2010) are evaluated with more emphasis on the uncertainties in criteria performance values (PVs) and the weighting. These DH systems cover a wide range: (1) coal-fired CHP; (2) gas-fired HOB; (3) oil-fired HOB; (4) coal-fired HOB; (5) solar energy HP; (6) water source heat pump (WSHP); and (7) ground source heat pump (GSHP). Most of the data for these DH systems are based on real-life existing DH installations.

Some previous studies have been carried out to develop multicriteria evaluation methods for choosing the optimal DH systems or heating technologies from the standpoints of technology, economy, and environment. Ghafghazi et al. (2010) have done a multicriteria evaluation for choosing the energy sources of a DH system in Vancouver, Canada; possible energy sources are natural gas, wood pellets, sewer heat, and geothermal heat. The evaluation criteria are: GHG emissions, particulate matter emissions, maturity of technology, traffic load, and local source. Kontu et al. (2015) carried out a multicriteria evaluation of heating systems including many renewable energy forms for a sustainable residential area in southern Finland. In their study, altogether 11 alternative heating systems were evaluated in terms of 15 criteria. The stochastic multicriteria acceptability analysis (SMAA) method was also used to analyze this problem, but the study did not take into account the uncertainty in weighting.

Soltero et al. (2016) developed a framework to evaluate the potential for natural gas cogeneration to reach decarbonization economy in Spain. The evaluation was implemented by environmental, economic, and regulatory analyses at four levels, including national, regional, municipality, and district, using a proposed top-down/bottom-up methodology. Li et al. (2016) evaluated the CCHP systems for hotels, offices, and residential buildings in Dalian, China, from energetic analysis, economic operation, and environment effect viewpoints. They use fuzzy optimum selection theory to evaluate the integrated performances of CCHP systems with various operation strategies, but the uncertainties in weighting the process are not well defined. The abovementioned methods worked well in the application-oriented case studies, but it could be better if uncertainties in criteria and weighting were considered in their studies.

In general, different kinds of uncertainties in criteria PVs and in subjective judgments (Zarghami and Szidarovszky, 2009; Durbach and Stewart, 2012) as well as policy and technology uncertainties (Tylock et al., 2012) are very common and thus should be treated carefully. In this study, we adopt the stochastic multicriteria acceptability analysis (SMAA) model to evaluate the DH systems, because it can handle the uncertainties by using a probability distribution function (PDF) and a Monte Carlo simulation (Wang and Haves, 2014). Moreover, we also propose to use the "feasible weight space" (FWS) but not a deterministic weight vector in MCDM, because the weights should indicate all DMs' preference information (Wang et al., 2015). In fact, FWS is a union of all weight vectors obtained from DMs' judgment matrices.

This study develops a more efficient method to the multicriteria decision analysis of heating, ventilating, and air conditioning systems, and to solve the problems of the uncertainties in criteria PVs and weighting, which were not treated carefully in previous studies. This paper is organized as following. Firstly, the SMAA models and FWS concept, as well as the way to handle the uncertainties, are introduced; followed by a case study in China, where the developed methods are demonstrated with seven candidate DH system options; finally the conclusion is drawn according to the results and discussion of the study.

7.2 Methods

SMAA is a family of models that encompasses many different variants (Tervonen and Figueira, 2008). This paper proposes to use SMAA-2 and SMAA-O models to solve the multicriteria decision making problems that have both quantitative and qualitative criteria (Lahdelma et al., 2001).

7.2.1 The SMAA-2 model

Let's take an MCDM problem, which has m alternatives $A=\{x_1, x_2, x_3, \ldots, x_m\}$ and n criteria. SMAA-2 model assumes that DM's preference can be expressed by a utility function $u(x_i,w)$; this function calculates the utility value for alternative x_i when using weight vector $\boldsymbol{w}$. We introduce a rank acceptability index to evaluate each alternative's acceptability according to the utility calculation results. A ranking function is defined to determine the ranking sequences from the best (1) to the worst (m), as in Lahdelma and Salminen (2001):

$$\operatorname{rank}(\xi_i, \boldsymbol{w}) = 1 + \sum\nolimits_k \rho[u(\xi_k, \boldsymbol{w}) > u(\xi_i, \boldsymbol{w})] \tag{7.1}$$

where $\rho(\text{true})=$and $\rho(\text{false})=0$, $\boldsymbol{\xi}$ is used to stand for criteria PVs having a stochastic distribution of $f_X(\boldsymbol{\xi})$; similarly $\boldsymbol{w}$ has a stochastic distribution of $f_W(\boldsymbol{w})$. Then the favorable rank weights, $W_i^r(\boldsymbol{\xi})$ is defined:

$$W_i^r(\boldsymbol{\xi}) = \{\boldsymbol{w} \in W : \operatorname{rank}(\xi_i, \boldsymbol{w}) = r\}, \text{ where } W = \left\{\boldsymbol{w} \in R^n : w_j \geq 0, \ \sum\nolimits_{j=1}^n w_j = 1\right\} \tag{7.2}$$

If a weight vector $\boldsymbol{w} \in W_i^r(\boldsymbol{\xi})$, then it makes that alternative x_i obtains rank r. Based on this, the rank acceptability index, b_i^r, can be defined as:

$$b_i^r = \int_X f_X(\boldsymbol{\xi}) \int_{W_i^r(\boldsymbol{\xi})} f_W(\boldsymbol{w}) d\boldsymbol{w} d\boldsymbol{\xi} \tag{7.3}$$

In fact, b_i^r indicates all the different valuations that make alternative x_i rank r. It is not possible to calculate b_i^r directly from the integral formula, but it can be calculated by using the Monte Carlo simulation. From this point of view, rank acceptability also can be explained as the share (%) of Monte Carlo simulations that make alternative x_i rank r. SMAA-2 uses a holistic acceptability index shown in Eq. (7.4) to consider contributions of all ranks; this is an improvement compared to the original SMAA model (Lahdelma et al., 1998).

$$a_i^h = \sum\nolimits_{r=1}^m \alpha b_i^r \tag{7.4}$$

where α_r are the meta-weights, which means the contribution of each rank acceptability index to the holistic evaluation. In general, first ranks contribute most and the worst ranks contribute least to the holistic acceptability index.

The central weight vector, $\boldsymbol{w}_i^c$, can be expressed as in Eq. (7.5).

$$\boldsymbol{w}_i^c = \frac{\int_X f_X(\boldsymbol{\xi}) \int_{W_i^1(\boldsymbol{\xi})} f_W(\boldsymbol{w}) \boldsymbol{w} d\boldsymbol{w} d\boldsymbol{\xi}}{b_i^1} \tag{7.5}$$

The central weight vector can be deemed as the best single representation of the preference from a DM supporting x_i. $\boldsymbol{w}_i^c$ is actually the average value of the weight vectors that make alternative i the best.

The confidence factor, p_i^c, is the probability that x_i ranks first when its central weight vector is used. That is to say, only the first rank acceptability b_i^1 has the confidence factor. It is expressed as:

$$p_i^c = \int_{\boldsymbol{\xi} \in X : \mathrm{rank}(\boldsymbol{\xi}, \boldsymbol{w}_i^c) = 1} f_X(\boldsymbol{\xi}) d\boldsymbol{\xi} \tag{7.6}$$

The confidence factor is used to evaluate whether the criteria PVs are accurate to differentiate alternatives using the central weight vectors.

In addition, we also can calculate the confidence factors for different alternatives using each other's' central weight vectors, which are called cross confidence factors. Better discrimination capability can be observed based on these cross confidence factors. The cross confidence factor for alternative x_i with respect to target alternative x_k is defined as:

$$p_{ik}^c = \int_{\xi \in X, b_k^1 \neq 0 : w_k^c \in W_i^1(\xi)} f_X(\xi) d\xi \tag{7.7}$$

The cross confidence factor measures the probability that x_i will obtain the first rank when the central weight vector of x_k is used. Nonzero cross confidence factors means that the alternative x_i will compete for the first rank with the central weight vector of alternative x_k and the competence extent can also be determined. Note that the cross confidence factor p_{ii}^c is exactly the confidence factor p_i^c. In all, rank acceptability indices, holistic acceptability indices, central weight vectors, and confidence factors are used to facilitate the evaluation of DH systems.

7.2.2 The SMAA-O model

The SMAA-O model was developed for problems with ordinal criteria (Lahdelma et al., 2003). It uses rank level numbers, $r_j = 1, 2, \ldots, j^{max}$, to sort the alternatives in terms of each criterion. It is clear that 1 is the best and j^{max} is the worst rank level. In reality, two or more alternatives may be considered equally good; so that $j^{max} \leq m$. In SMAA-O, the ordinal measurements are mapped into the cardinal values. All consistent mappings between the ordinal scales and cardinal values are considered. Monte Carlo simulations are used to generate random cardinal values corresponding to the ordinal values. Let γ_j be the cardinal values for rank levels, r_j, then the mapping (David and Nagaraja, 2003) is:

$$\gamma_j = v_j(r_j) \tag{7.8}$$

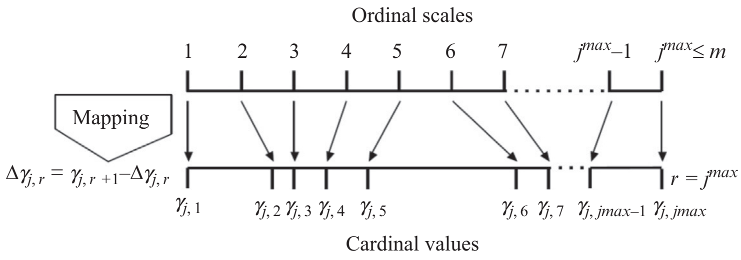


FIG. 7.1 The mapping from ordinal scales to cardinal values in SMAA-O.

The lower the rank is, the better for an alternative; therefore, $\nu(\bullet)$ should be a monotone decreasing mapping. In this study, γ_j is in the interval [0, 1]. The mapping process is shown in Fig. 7.1. The sum of the scale intervals can be expressed as:

$$\sum_{r=1}^{j^{max}-1} \Delta\gamma_{j,r} = \sum_{r=1}^{j^{max}-1} \left(\gamma_{j,r+1} - \gamma_{j,r}\right) = 1 \tag{7.9}$$

Therefore, the problem becomes to simulate all cardinal scales that satisfy:

$$\Gamma_j = \left\{ \Delta\gamma_j \in R^{j^{max}-1} : \Delta\gamma_{j,r} > 0, \sum_{r=1}^{j^{max}-1} \Delta\gamma_{j,r} = 1 \right\} \tag{7.10}$$

The valid interval space will expand as the mapping numbers (K) increase; this is illustrated in Fig. 7.2 for $j^{max} = m = 11$. It is clear that the mapping from ordinal scales to cardinal values can cover more and more interval space with more iterations.

If there is no information about the scale intervals, then we can use a uniform distribution in the simulation. During the simulation, $j^{max} - 2$ distinct random numbers will be generated according to the uniform distribution in [0, 1] and be sorted in decreasing order so that $1 = \gamma_j, 1 > \gamma_j, 2 > \ldots > \gamma_j, j^{max} = 0$. SMAA-O also has rank acceptability indices, the central weight vectors, and the confidence factors.

7.2.3 Feasible weight space

A weight vector is only one point in the weight space, but only one point is not a good representation for the preferences of a group of DMs (Liu et al., 2017) in real life. This is why we propose to use the feasible weight space (FWS) concept. FWS is actually a part of the general weight space; it assumes random variables with certain probability distributions in the feasible subspace. Therefore, weight vectors are taken with certain probability distributions from the FWS in the Monte Carlo simulation. For example, in a three criteria problem, the general weight space can be shown as a plane in Fig. 7.3A; but a possible FWS with interval constraints is demonstrated as a polygon shaded area shown in Fig. 7.3B. This FWS can be expresses as:

$$W = \left\{ \boldsymbol{w} \in R^n : w_j \geq 0,\ w_j^{\min} \leq w_j \leq w_j^{\max}, \sum_{j=1}^{n} w_j = 1 \right\} \tag{7.11}$$

FWS identifies a more accurate subspace than the general weight space and covers. For group decision making, it is necessary to obtain this subspace to cover all DMs' preferences. However, if there are not too many DMs, then we can set an interval for each criterion based on the calculated weight vector to represent the uncertainties.

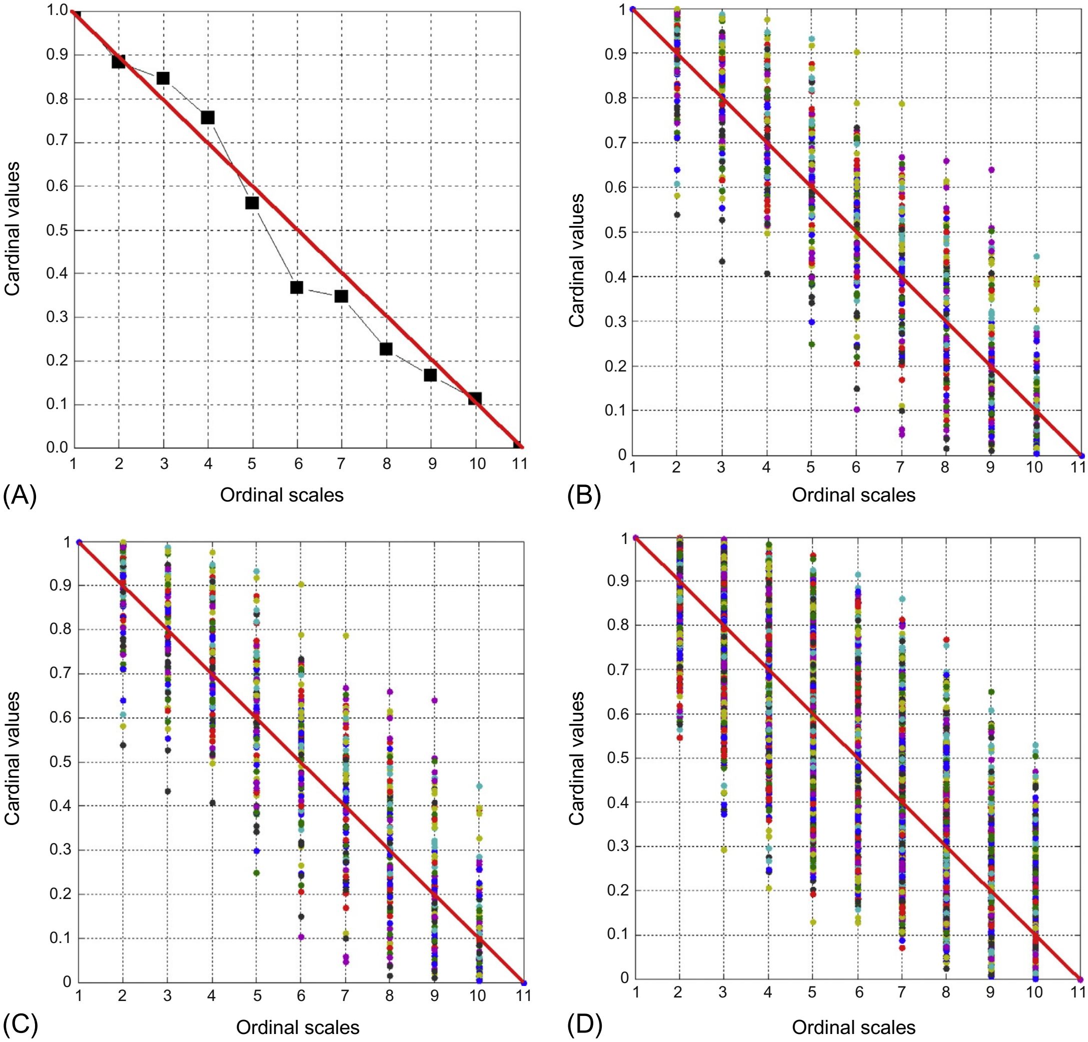


FIG. 7.2 Variation of Γ_j mapping ordinal scales onto cardinal values with simulation iterations K from 1 to 1000 in SMAA-O when $j^{max} = m = 11$ (the *straight lines* stand for linear mapping) (Wang, 2013). (A) $K = 1$, (B) $K = 100$, (C) $K = 500$, and (D) $K = 1000$.

7.2.4 Handling the uncertainties

A certain probability distribution around the expected values of the criteria PVs is used to express the uncertainties. The most popular distributions are uniform and normal distributions (Lahdelma et al., 1998) and the former one is used in this study. The SMAA-O model already takes into account the uncertainties when simulating the mapping processes; therefore, we only focus on how to treat the uncertainties in weighting by taking a 3-criterion example. However, the same technique can be used in higher dimensions.

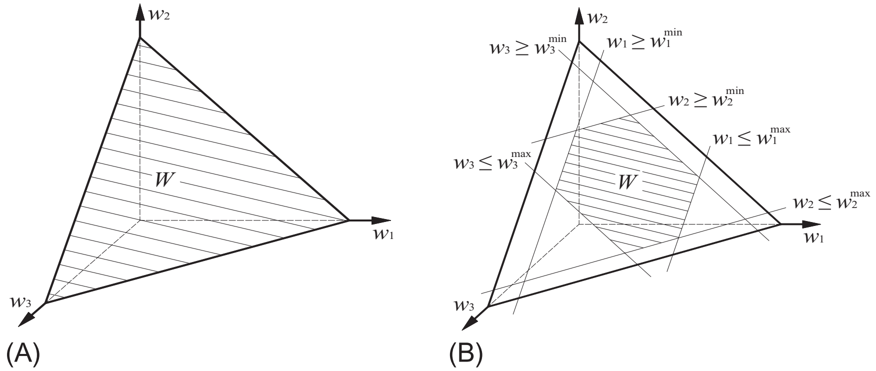


FIG. 7.3 A deterministic weight vector A in a general weight space of a three criteria case and an FWS with interval constraints on each criterion. (A) A general 3D weight space and (B) FWS with interval constraints in 3D weight space.

If there is no weighting information in the extreme cases, a uniform distribution is assumed. In 3-criterion case, the FWS is a $(n-1)$-dimensional simplex. Fig. 7.4 shows the projection onto w_1-w_2 plane for the FWS in Fig. 7.3, respectively.

The weight intervals $w_j \in [w_j^{min}, w_j^{max}]$ may come from direct preference statements of the DMs or from judgments matrices. The intervals can be obtained by restricting the uniform weight distribution with linear inequality constraints.

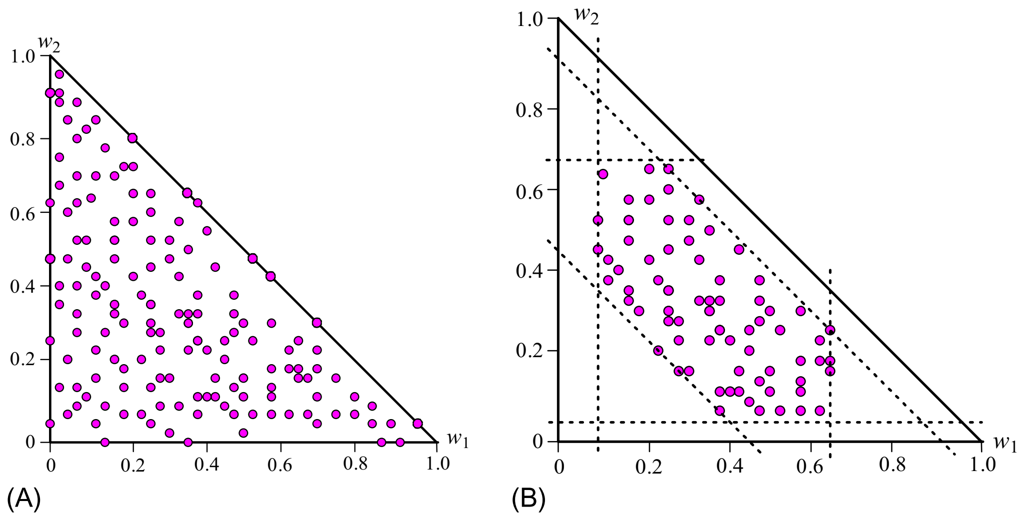


FIG. 7.4 Projection onto w_1-w_2 plane of the FWS shown in Fig. 7.3. (A) a general 3D weight space and (B) FWS with interval constraints in 3D weight space.

7.3 Results and discussion

7.3.1 The case of seven DH systems in a city of north China

The seven DH systems are planned based on the same DH area in Baoding, a city in the north of China (Wei et al., 2010). The floor heating area is 251,746 m^2 with a design heat load of 16.6 MW. Space heating season is 120 days a year and the average outdoor air temperature is −1.6°C. The design indoor and outdoor air temperatures are 18°C and −9°C respectively. Assume that all DH systems provide the same DH capacity for this area, and then the properties of economy, environment, and energy for the seven DH systems are shown in Table 7.1 (Wei et al., 2010).

There are both quantitative and qualitative criteria in Table 7.1. The uncertainty of economic indices is assumed as 10% (Hokkanen et al., 2000); because the emission data has large flexibility, so an uncertainty of ±20% is used for the environmental criteria. However, for the qualitative (ordinal) energy criteria, the uncertainty will be handed by SMAA-O automatically using the Monte Carlo simulation. The uncertainty in weighting is considered by an FWS in this chapter. The FWS is obtained by giving ±50% linear constraints using uniform distribution (Wang et al., 2015) to each criterion based on the weight vector elicited by Wei et al. (2010). The FWS can cover more possible preference information, indicated in Fig. 7.5.

7.3.2 Results of stochastic multicriteria acceptability analysis

The criteria PVs of the seven DH systems in Table 7.1 should be normalized first before being used in SMAA. In this study, we used 100,000 Monte Carlo iterations to calculate the statistic variables in the simulation (Wang et al., 2016), and this will result in error limits

TABLE 7.1 Properties of the seven DH systems.

Criteria	Coal-fired CHP	Gas-fired HOB	Oil-fired HOB	Coal-fired HOB	Solar energy HP	WSHP	GSHP
Total cost per floor area (¥/m^2)[a]	26.96	46.85	78.40	32.19	67.88	54.95	62.27
NO_x (g/m^2)[b]	588.0	92.9	116.0	840.0	91.9	78.4	87.8
SO_2 (g/m^2)[b]	179.0	94.0	127.0	255.7	162.0	138.1	154.8
CO (g/m^2)[b]	8.9	1.9	3.5	40.9	1.14	0.85	0.95
CO_2 (g/m^2)[b]	40871	31920	40314	58224	24054	20504	22985
Other (g/m^2)[b]	73.9	27.1	18.1	105.6	22.2	19.0	21.3
Technical merits[c]	Good	Good	Good	Little bad	Neutral	Good	Good
Mentality effect[c]	Better	Good	Good	bad	Good	Good	Good
Heating charge[c]	Better	Neutral	Bad	Better	Bad	Little bad	Neutral

[a] *Includes the annuity of initial investment and annual operating cost, ¥ means Chinese currency RMB yuan.*
[b] *Emission is calculated based on the floor heating area.*
[c] *These three properties are deemed as qualitative (ordinal) criteria.*

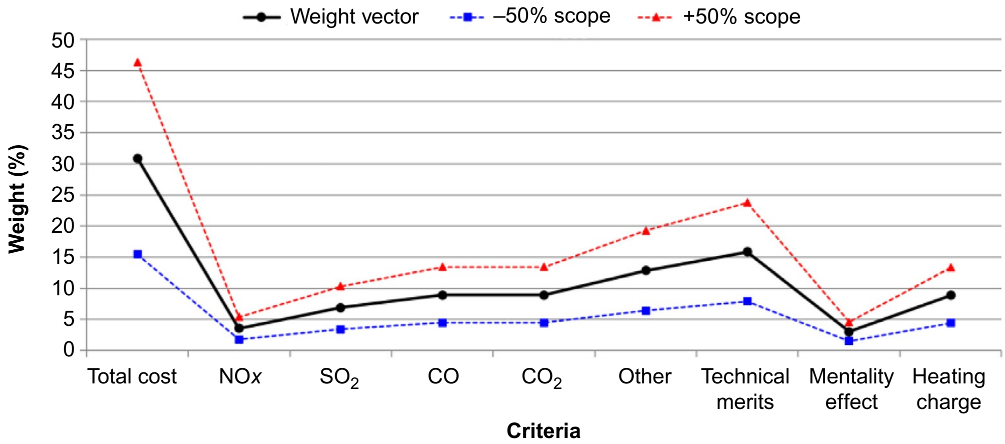


FIG. 7.5 The FWS with ±50% interval constraints using uniform distribution on each criterion for evaluation of the DH systems.

smaller than 0.01 (Tervonen and Lahdelma, 2007). The confidence factors, holistic acceptability, and rank acceptability indices are shown in Table 7.2. All rank acceptability indices and average utilities of each DH system are also illustrated graphically in Fig. 7.6. Here the average utility is defined as the central-weighted average utility function value based on the criteria PVs. Central weight vectors and the cross confidence factors are shown in Fig. 7.7 and Table 7.3. Note that there is no central weight vector for DH systems having zero confidence factors.

According to Table 7.2, coal- and oil-fired HOBs can be rejected from the most qualified DH systems, because their confidence factors are zero. This means that they never obtain the first rank even considering uncertainties. Similarly, solar energy HP is currently not a good choice for DH because of its nearly zero confidence factor and very low holistic acceptability index. GSHP has a 4.87% confidence factor, which is also deemed too small to be the best alternative, but it still can be a compromise DH system, especially if a weight vector close to its central weights is used. However, coal-fired CHP has a high confidence factor of 75.75%, followed by

TABLE 7.2 Confidence factors (p^c), holistic (a^h) and rank acceptability indices (b^r) in percentage.

DH system	p^c	a^h	b^1	b^2	b^3	b^4	b^5	b^6	b^7
Coal-fired CHP	75.75	73.7	51.2	24.9	13.3	10.0	0.5	0	0
Gas-fired HOB	57.43	72.5	39.2	48.8	10.8	1.3	0	0	0
WSHP	18.92	45.7	8.3	20.3	54.2	17.1	0.1	0	0
GSHP	4.87	31.0	1.4	5.9	21.6	69.5	1.7	0	0
Solar energy HP	0.02	12.5	0.001	0	0.1	1.5	68.5	27.7	2.3
Oil-fired HOB	0	6.6	0	0	0	0.1	18.6	55.6	25.7
Coal-fired HOB	0	2.9	0	0	0.1	0.6	10.8	16.6	72.0

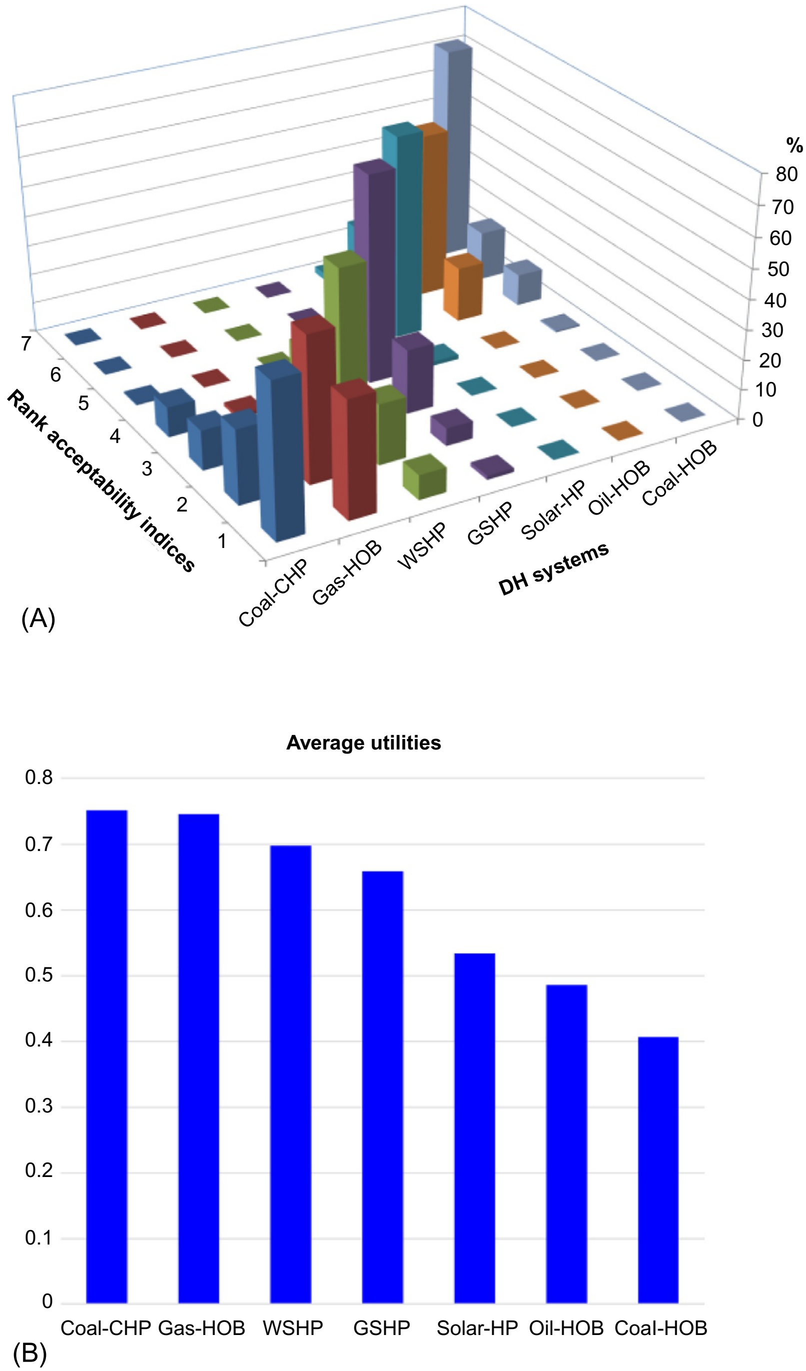
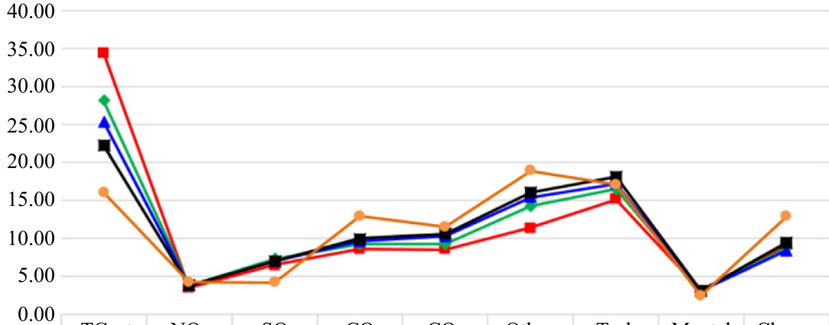


FIG. 7.6 Rank acceptability indices (b^r) and the average utilities of the seven DH systems. (A) Rank acceptability indices and (B) average utilities.



	TCost	NO_x	SO_2	CO	CO_2	Other	Tech	Mental	Charge
Coal-CHP	34.41	3.49	6.49	8.59	8.50	11.37	15.07	3.05	9.04
Gas-HOB	28.08	3.71	7.29	9.22	9.20	14.22	16.52	3.06	8.69
WSHP	25.34	3.76	7.00	9.64	10.27	15.40	17.16	3.07	8.36
GSHP	22.22	3.79	6.93	9.97	10.55	15.99	18.09	3.09	9.37
Solar-HP	16.06	4.19	4.12	12.90	11.50	18.86	17.07	2.46	12.83

FIG. 7.7 Central weights (w^c) favoring different DH systems.

TABLE 7.3 Cross confidence factors (%), confidence factors are in bold, and the biggest cross confidence factors are underlined.

Alt.	Coal-CHP	Gas-HOB	Oil-HOB	Coal-HOB	Solar-HP	WSHP	GSHP	Sum
Coal-CHP	**<u>75.752</u>**	23.187	–	–	0	1.048	0.013	100
Gas-HOB	31.994	**<u>57.432</u>**	–	–	0	9.806	0.768	100
Oil-HOB	–	–	–	–	–	–	–	–
Coal-HOB	–	–	–	–	–	–	–	–
Solar-HP	3.225	<u>44.57</u>	–	–	**0.021**	32.316	19.868	100
WSHP	15.445	<u>63.084</u>	–	–	0	**18.916**	2.555	100
GSHP	9.598	<u>62.412</u>	–	–	0	23.118	**4.872**	100

gas-fired HOB (57.43%), and WSHP (18.92%). In addition, the first rank acceptability index of coal-fired CHP is 51.2%, which already dominates the other DH systems. Nevertheless, the second rank acceptability index is 24.9% and zero for worst ranks, which means that coal-fired CHP is the most preferred DH technology in the study area with such a big DH load (16.6 MW) considering uncertainties. Gas-fired HOB and WSHP also can be the compromise DH systems if their central weight vectors are used.

As can be seen from Fig. 7.6, coal-fired CHP favors criteria of total cost, but if the DMs are not emphasizing the total cost, then gas-fired HOB is very suitable for DH in this area. This is also justified by the SMAA result that gas-fired HOB competes well with the three HP systems and has a good chance of being the best alternative, even when three HP systems' central weights are finally used. This can be justified by Table 7.3, e.g., when the central weight of solar HP system is finally chosen, the cross confidence factor of gas-HOB is far bigger than the confidence factor itself (0.021%) for solar HP. In fact, it is the smallest confidence factor, because gas-HOB, WSHP, GSHP, and coal-CHP all have bigger cross confidence factors when solar HP is the target alternative. This means that solar HP will not be the most preferred or compromise alternative. The situations are similar when other two HPs' central weights are used, because gas-HOB will dominate the WSHP and GSHP. In other words, HPs only have small chances to be the best alternative even if weights are close to their central weights (close to the central weight of gas-HOB too), as shown in Fig. 7.7.

The ranking of the DH systems based on the average utility can be found in Fig. 7.6. The same ranking can be obtained through holistic acceptability in Table 7.2. We found that the first three rankings are the same as the result given by Wei et al. (2010). The ranking sequences of GSHP, solar energy HP, and oil-fired HOB are also the same. The only difference is that a coal-fired HOB ranks 4 in their conclusion, but it is apparently the worst alternative in our study. The reason is that if total cost is emphasized, then coal-fired HOB is dominated by coal-fired CHP, otherwise it is dominated by other DH technologies characterized by lower emissions.

7.3.3 Discussion

Pairwise winning indices can also be defined if the above statistic variables are still not enough to differentiate the alternatives. The pairwise winning index c_{ij} is the probability for alternative i to score better than alternative j considering the uncertainty in the preference statements. It can be calculated by the times that alternative i is better than j divided by the total Monte Carlo simulation iterations. For the seven DH technologies, their pairwise winning indices are shown in Table 7.4. The pairwise winning indices of one alternative to itself is zero. It can be found that coal-CHP is highly certain to dominate other alternatives, but not the gas-HOB, which is the second best alternative in this study. In fact, gas-HOB is even more certain to dominate other alternatives, however the pairwise winning index for gas-HOB compared to coal-CHP is 46.90% (less than 50%), which means that coal-CHP is more likely better than gas-HOB.

TABLE 7.4 Pairwise winning indices of the seven DH systems (%).

Alt.	Coal-CHP	Gas-HOB	Oil-HOB	Coal-HOB	Solar-HP	WSHP	GSHP
Coal-CHP	0	53.10	99.26	99.99	98.06	71.41	82.14
Gas-HOB	46.90	0	99.88	99.94	99.30	72.92	86.94
Oil-HOB	0.74	0.12	0	70.87	28.14	0.43	1.37
Coal-HOB	0.01	0.06	29.13	0	15.67	0.60	1.97
Solar-HP	1.94	0.71	71.86	84.33	0	2.16	6.44
WSHP	28.59	27.08	99.57	99.40	97.84	0	69.60
GSHP	17.86	13.06	98.63	98.03	93.56	30.40	0

A full ranking sequence of all DH systems can also be obtained according to the simulation results. However, this may lead to some kind of misunderstanding and thus is not encouraged. DMs may believe that the alternative with largest utilities dominates all the others, disregarding the fact that ranking sequence is subject to uncertainties in weighting. Therefore, SMAA method is used to help understand the evaluation by using the rank acceptability indices and the confidence factors. SMAA plus FWS can make clear that what kind of weight information will favor what kind of alternative taking into account of the probability. This makes the combination of SMAA and FWS more reliable in the multicriteria evaluation. Therefore, the proposed method can also be used in other MCDM problems.

7.4 Conclusions

In this study, the stochastic multicriteria analysis (SMAA) method in combination with the feasible weight space (FWS) are used to evaluate the DH systems considering uncertainties in criteria performance values (PVs) and weighting. The uncertainties in criteria PVs are treated using uniform distribution function within the uncertainty range of each criterion, and uncertainties in weighting are addressed directly by the Monte Carlo simulation in SMAA. The statistic variables of rank acceptability indices, confidence factors, central weight vectors, and cross confidence factors of SMAA can help DMs understand what kind of weight information will favor what kind of best alternatives to what extent and of better discriminations.

The method was demonstrated in a city in the north of China, and the results indicate that at present, the coal-fired CHP has the best chance to be the most preferred DH technology in this area with DH load at 10MW level or higher, especially the economic criterion is emphasized. Otherwise the gas-fired HOB can be more suitable for DH, because it dominates all three heat pump (HP) technologies. Among the HP systems, the water source heat pump (WSHP) is better than ground source heat pump (GSHP) and solar energy heat pump in the case study area. SMAA also helps reveal the inefficient alternatives of oil- and coal-fired HOBs, because they are dominated by other DH technologies even considering the uncertainties. A full ranking of the alternatives is not recommended in this study, because it will very easily lead to a misunderstanding that the best alternative dominates all the others in any situation. On the contrary, we should bear in mind that the any ranking is subject to uncertainties, which should be well considered.

Acknowledgments

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References

Cao, Y., Chen, G.M., Xu, W.F., 2002. Optimization of heating modes for medium and large cities. J. HVAC 32 (1), 7–10.
David, H.A., Nagaraja, H.N., 2003. Order Statistics. Wiley Series in Probability and Statistics.
Durbach, I.N., Stewart, T.J., 2012. Modeling uncertainty in multi-criteria decision analysis. Eur. J. Oper. Res. 223, 1–14.

Gang, W.J., Wang, S.W., Yan, C.C., Xiao, F., 2015. Robust optimal design of building cooling systems concerning uncertainties using mini-max regret theory. Sci. Technol. Built Environ. 21 (6), 789–799.

Ghafghazi, S., Sowlati, T., Sokhansanj, S., Melin, S., 2010. A multicriteria approach to evaluate district heating system options. Appl. Energy 87 (4), 1134–1140.

Hokkanen, J., Lahdelma, R., Salminen, P., 2000. Multicriteria decision support in a technology competition for cleaning polluted soil in Helsinki. J. Environ. Manage. 60, 339–348.

Huang, J., 2011. Comprehensive evaluation for the setup of peak heating using gas fired boiler. Master thesis, Beijing University of Civil Engineering and Architecture, Beijing, China.

Jiang, W., Reddy, T.A., Gurian, P., 2007. General methodology combining engineering optimization of primary HVAC&R plants with decision analysis methods—part II: uncertainty and decision analysis. HVAC&R Res. 13 (1), 119–140.

Kontu, K., Rinne, S., Olkkonena, V., Lahdelma, R., Salminen, P., 2015. Multicriteria evaluation of heating choices for a new sustainable residential area. Energy Build. 93, 169–179.

Lahdelma, R., Salminen, P., 2001. SMAA-2: stochastic multicriteria acceptability analysis for group decision making. Oper. Res. 49 (3), 444–454.

Lahdelma, R., Hokkanen, J., Salminen, P., 1998. SMAA—stochastic multiobjective acceptability analysis. Eur. J. Oper. Res. 106 (1), 137–143.

Lahdelma, R., Salminen, P., Simonen, A., Hokkanen, J., 2001. Choosing a reparation method for a landfill using the SMAA-O multicriteria method. In: Multiple Criteria Decision Making in the New Millennium, Lecture Notes in Economics and Mathematical Systems, 2001, 507. Springer-Verlag, Berlin, pp. 380–389.

Lahdelma, R., Miettinen, K., Salminen, P., 2003. Ordinal criteria in stochastic multicriteria acceptability analysis (SMAA). Eur. J. Oper. Res. 147, 117–127.

Li, M., Mu, H.L., Li, N., Ma, B.Y., 2016. Optimal design and operation strategy for integrated evaluation of CCHP (combined cooling heating and power) system. Energy 99, 202–220.

Lin, Z.H., 2002. Influences of the energy policy reformation of our country on developments of industrial boilers. Ind. Boiler 3, 4–7.

Liu, C., Tang, G.L., Liu, P.D., 2017. An approach to multicriteria group decision-making with unknown weight information based on Pythagorean fuzzy uncertain linguistic aggregation operators. Math. Probl. Eng. 6414020, https://doi.org/10.1155/2017/6414020.

Shan, K., Wang, S.W., Xiao, F., Sun, Y.J., 2013. Sensitivity and uncertainty analysis of cooling water control strategies. HVAC&R Res. 19 (4), 435–443.

Soltero, V.M., Chacartegui, R., Ortiz, C., Velázquez, R., 2016. Evaluation of the potential of natural gas district heating cogeneration in Spain as a tool for decarbonisation of the economy. Energy 115, 1513–1532.

Tervonen, T., Figueira, J.R., 2008. A survey on stochastic multicriteria acceptability analysis methods. J. Multi-Crit. Decis. Anal. 15 (1–2), 1–14.

Tervonen, T., Lahdelma, R., 2007. Implementing stochastic multicriteria acceptability analysis. Eur. J. Oper. Res. 178 (2), 500–513.

Troldborg, M., Heslop, S., Hough, R.L., 2014. Assessing the sustainability of renewable energy technologies using multi-criteria analysis: suitability of approach for national-scale assessments and associated uncertainties. Renew. Sustain. Energy Rev. 39, 1173–1184.

Tsinghua University Building Energy Research Center, 2014. Annual Report on China Building Energy Efficiency. China Building Industry Press.

Tylock, S.M., Seager, T.P., Snell, J., Bennett, E.R., Sweet, D., 2012. Energy management under policy and technology uncertainty. Energy Policy 47, 156–163.

Wang, H.C., 2013. Development and Application of a Multicriteria Decision Support Framework for Planning or Retrofitting District Heating Systems. Aalto University publication series Doctoral dissertations.

Wang, L.P., Haves, P., 2014. Monte Carlo analysis of the effect of uncertainties on model-based HVAC fault detection and diagnostics. HVAC&R Res. 20 (6), 616–627.

Wang, H.C., Jiao, W.L., Lahdelma, R., Zhu, C.Z., Zou, P.H., 2015. Stochastic multicriteria acceptability analysis for evaluation of combined heat and power units. Energies 8 (1), 59–78.

Wang, H.C., Zhou, Z.G., Jiao, W.L., Wang, P., Lahdelma, R., 2016. Stochastic multicriteria evaluation of district heating systems considering the uncertainties. In: Proceedings of the 9th International Symposium on Heating, Ventilation and Air Conditioning (ISHVAC)—The 3rd International Conference on Building Energy and Environment (COBEE), July 12-15, Tianjin, China.

Wei, B., Wang, S.L., Li, L., 2010. Fuzzy comprehensive evaluation of district heating systems. Energy Policy 38, 5947–5955.

Zarghami, M., Szidarovszky, F., 2009. Stochastic-fuzzy multi criteria decision making for robust water resources management. Environ. Res. Risk Assess. 23, 329–339.

CHAPTER

8

Framework of life cycle sustainability assessment

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6.1 Introduction

With continuous growth of population globally, the conflict between growing resources demand and worsening environment has become obvious and urgent to solve. Most environmental issues have been found to be worsened by overusing of limited resources and unsuitable waste treatments. To alleviate this trend, the sustainable development concept has been raised and attracted a lot of attention. In the "Our Common Future" conference published by World Commission on Environment and Development (WCED) in 1987, the Brundtland Commission defined sustainable development as "development that meets the needs of the present without compromising the ability of future generations to meet their own". This definition has been used until today. For this purpose, tools need to be developed to assist practitioners or stakeholders in developing more sustainable products, services, or production processes. As researchers are aware of the importance of tool development, the life cycle concept was developed to emphasize the significance of overall assessment. Life cycle assessment (LCA) was the first and the widest accepted tool developed to evaluate environmental sustainability. With revision and further studies, LCA has been standardized as ISO 14040 and ISO 14044 and been adapted in several areas such as sustainability assessment of projects in the construction industry, sustainability assessment of new technologies in manufacturing industries, and environmental assessment of new production development. However, since LCA only considers the environmental impacts of the research targets, it leads to ignorance of economic benefits and social impacts. Because of this, life cycle sustainability assessment (LCSA) considering environmental, economic, and social aspects was raised.

Life Cycle Sustainability Assessment for Decision-Making
https://doi.org/10.1016/B978-0-12-818355-7.00008-7

LCSA consists of LCA for environmental assessment, life cycle costing (LCC) for economic assessment, and social life cycle assessment (SLCA) for social impact assessment (Heijungs et al., 2010). Due to the completeness of overall analysis of sustainability, the studies related to LCSA have increased in recent years. To apply LCSA in more situations, LCSA has been extended to the fuzzy framework (Kouloumpis and Azapagic, 2018), tiered framework (Chen and Holden, 2018), dynamic system (Onat et al., 2016a), and by combining with other theories. Instead of examining the sustainability performances for a certain product or a service, the studies related to LCSA focus more on the comparison of multiple objects. LCSA combined with multicriteria decision making (MCDM) has been widely used in energy (Yu and Halog, 2015; Moslehi and Reddy, 2019), transportation (Onat et al., 2016c, Onat et al., 2016a), construction (Ferrari et al., 2019; Zheng et al., 2019), logistics (van Kempen et al., 2017), agriculture (Moriizumi et al., 2010; Chen and Holden, 2018), and other fields for alternatives selection or ranking.

In this chapter, the framework of LCSA is illustrated in detail and attached with corresponding examples for further explanation in Section 6.2; the research trend and the current progress of LCSA are summarized by reviewing literature related to LCSA in the Section 6.3; and Section 6.4 concludes the chapter.

6.2 Framework of LCSA

LCA, LCC, and SLCA follow the same framework for analysis. In this chapter, the framework of LCA defined by the international standard ISO 14044 is introduced with examples to illustrate the operation process of LCSA.

6.2.1 Goal and scope definition

The definitions of goal and scope in LCSA are important to be clarified at the beginning of the project. They are the keys to keep all procedures afterward following the consistent research standards and boundary. According to ISO 14044, the goal for an LCSA test should clearly state the intended application, the reasons for carrying out the study, the intended audience, and whether the results are intended to be used in comparative assertions intended to be disclosed to the public. Furthermore, the scope part should define the product system to be studied, the functions of the product system or, in the case of comparative studies, the systems, the functional unit, the system boundary, allocation procedures, life cycle impact (LCI) assessment methodology and types of impacts, interpretation to be used, data requirements, assumptions, value choices and optional elements, limitations, data quality requirements, type of critical review, and type and format of the report required for the study.

Taking a solid waste management system in Regione Campania (Arena et al., 2003) as an example, the goal and scope of this project are summarized in Table 8.1. As not all information is provided in the article, some hypothetical adjustments had been made.

The details of this procedure are adjustable depending on the real situations of the research objective. It is worth mentioning that if two alternatives are supposed to be compared for their sustainability performances by conducting LCA, the goal and scope need to be corresponding in the two assessments.

TABLE 8.1 The goal and scope of solid waste management in Regione Campania.

Goal	
The intended application	To find out municipal solid wastes (MSW) management options in Regione Campania
The reasons for carrying out the study	To develop information and tools to evaluate the environmental performance of alternative MSW management options in the area of Regione Campania
The intended audience	The Italian Committee for Waste Emergency in Campania
Whether the results are intended to be used in comparative assertions intended to be disclosed to the public	The results are intended to be used in comparative assertions intended to be disclosed to the public
Scope	
The product system to be studied	The municipal solid wastes management options in the Regione Campania
The functions of the product system or, in the case of comparative studies, the systems	To manage solid rest waste, i.e., the MSW residual from separate household collection, having a given quantity and composition
The functional unit	The management of 1 kg of rest waste of the composition measured as average in Campania, shown in Table 8.2
The system boundary	The system boundary is shown in Fig. 8.1
Allocation procedures	The allocation of material and energy flows starts with listing out all material and energy inputs and outputs from the examination system. Then every single material or energy source should be traced from its beginning to the end in the system. The material flow and the energy flow can be determined and be summarized in a flow chart
Life cycle impact (LCI) assessment methodology and types of impacts	The LCI assessment is planned to be conducted through LCA software GaBi. The types of impacts include abiotic depletion potential of elements (ADPe), abiotic depletion potential of fossil fuels (ADPf), acidification potential (AP), eutrophication potential (EP), freshwater aquatic ecotoxicity potential (FAETP), global warming potential (GWP), human toxicity potential (HTP), marine aquatic ecotoxicity potential (MAEP), ozone-layer depletion potential (ODP), photochemical oxidants creation potential (POCP), and terrestrial ecotoxicity potential (TETP)
Interpretation to be used	Consistency check, completeness check, and sensitivity check
Data requirements	The generic data is provided by the most valued international data bank, the Boustead Ltd., and the specific data is collected by site investigations
Assumptions	The inputs and outputs are constant every year

Continued

TABLE 8.1 The goal and scope of solid waste management in Regione Campania—cont'd

Value choices and optional elements	The weighting process will be conducted to show the preference of each criterion. The preference will reflect on the final results of LCSA
Data quality requirements	A data quality assessment with 6 criteria will be conducted for the data collected. The criteria are completeness, parameter uncertainty, methodological appropriateness and consistency, technological, geographical, and time-related representativeness

Adapted from Arena, U., Mastellone, M.L., Perugini, F., 2003. The environmental performance of alternative solid waste management options: a life cycle assessment study. Chem. Eng. J. DOI: https://doi.org/10.1016/j.cej.2003.08.019.

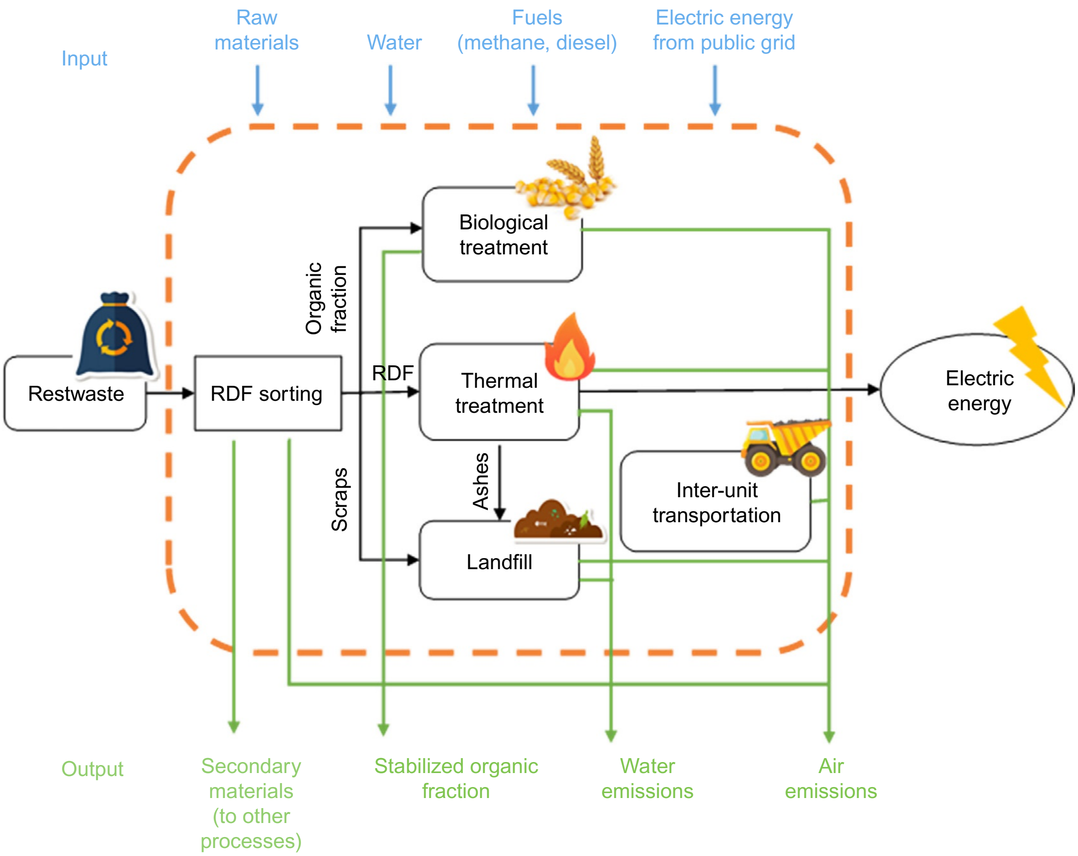

FIG. 8.1 The system boundary of solid waste management in Regione Campania. *Adapted from Arena, U., Mastellone, M.L., Perugini, F., 2003. The environmental performance of alternative solid waste management options: a life cycle assessment study. Chem. Eng. J. https://doi.org/10.1016/j.cej.2003.08.019 (web archive link).*

TABLE 8.2 Composition of MSW in Campania, as obtained from a specific investigation made by the National Committee for Waste Emergency.

Waste component	Content in rest waste (wt%)
Glass	5.7
Metals	3.25
Wood	1.75
Food wastes	30.1
Greens	3.88
Paper and paperboard	23.15
Plastics, light	7.92
Plastics, hard	2.84
Textiles	4.48
Leather	1.76
Oversize	0.7
Inert materials	1.26
Miscellaneous	4.49
Fines	8.7

Adapted from Arena, U., Mastellone, M.L., Perugini, F., 2003. The environmental performance of alternative solid waste management options: a life cycle assessment study. Chem. Eng. J. https://doi.org/10.1016/j.cej.2003.08.019.

6.2.2 Life cycle inventory

The life cycle inventory is designed to clarify the energy flows and material flows of the systems. Information should be collected completely and clearly in the system to avoid redundancy or missing information. It is important to conduct life cycle inventory accurately, as its precision affects significantly the accuracy of examination results. The procedures of life cycle inventory include preparing for data collection, data collection, validation of data, relating data to unit process, relating data to a functional unit, data aggregation, refining the system boundary, and completed inventory, as shown in Fig. 8.2. The steps mentioned need to be conducted repeatedly if any revisions are made.

Taking Kalundborg symbiosis in Denmark as an example, the life cycle inventory is assessed. Kalundborg Symbiosis is a great illustrator of the circular economy concept. It achieved with a group of functional corporations run with mutual material supplies and mutual waste disposal in Kalundborg, Denmark (Valentine, 2016). This project started in 1959 as a pioneer experimental project with considerable support from Denmark's government (John and Nicholas, 2018). This is an organically evolving, self-sustaining environmental collaboration in Denmark with 12 enterprises currently participating (Gulipac, 2016), and it has become a typical case of industrial symbiosis learned by other countries (Jacobsen, 2006). Kalundborg Symbiosis consists of Lake Tisse, Argo, Gyproc, Kalundborg Utility, Ørsted (also

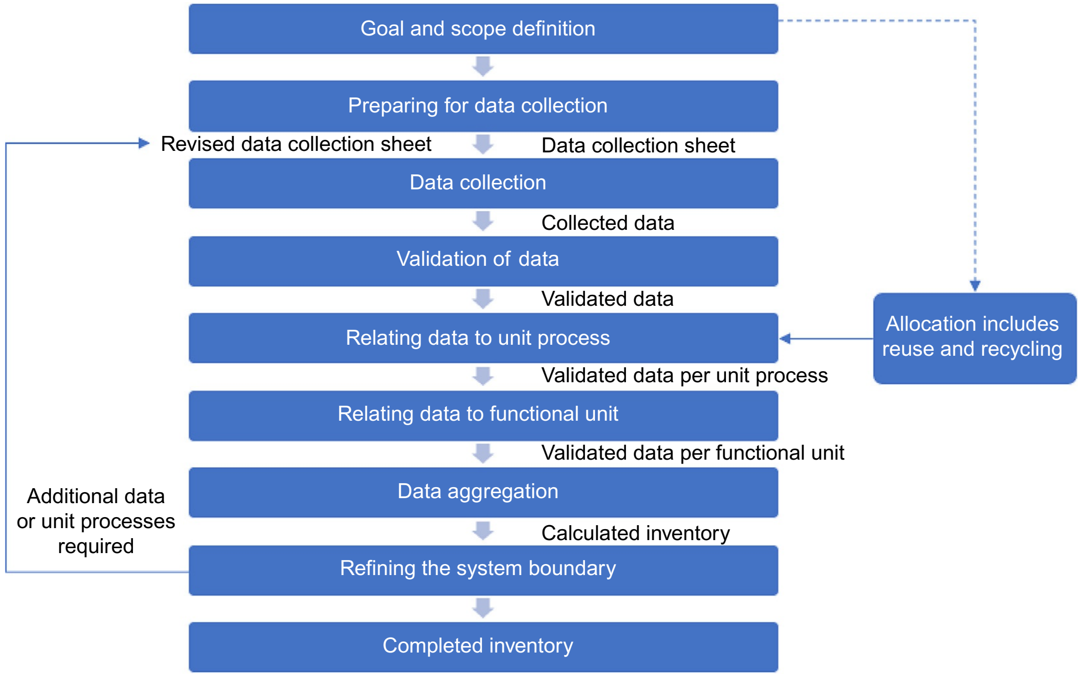

FIG. 8.2 Life cycle inventory.

known as Dong Energy), Statoil, Novo Nordisk & Novozymes Land Owner's Association, Novo Nordisk, Inbicon, Kalundborg Utility Heat pump, Novozymes, and Novozymes Wastewater & Biogas (Symbiosis Centre Denmark, 2010). Among all the companies in Kalundborg Symbiosis, Dong Energy, Statol, Gyproc, Novozymes, and Novo Nordisk run the five core businesses in this symbiosis system. Dong Energy is a power generation company providing electricity, steam, and heat to other companies and local residences. Statoil serves as a refinery factory that provides the cleanest petrol and diesel as the output products. The main service of Gyproc is manufacturing plasterboard. In addition, Novozymes produces enzymes and Novo Nordisk produces insulin. Other companies in symbiosis support these five cores by providing material processing, waste treatment, and management coordination services. For example, Kalundborg utility takes charge of water treatment and supply for the whole system and Argo handles wastes produced in the system.

The Kalundborg symbiosis system can be described by two resource exchange subsystems, namely energy exchange system (shown in Fig. 8.3) and material exchange system (shown in Fig. 8.4). This categorization method helps to explain how resources flow in the system more clearly.

The data in this step should be collected for every single arrow in Figs. 8.3 and 8.4 after transferring those data into functional units.

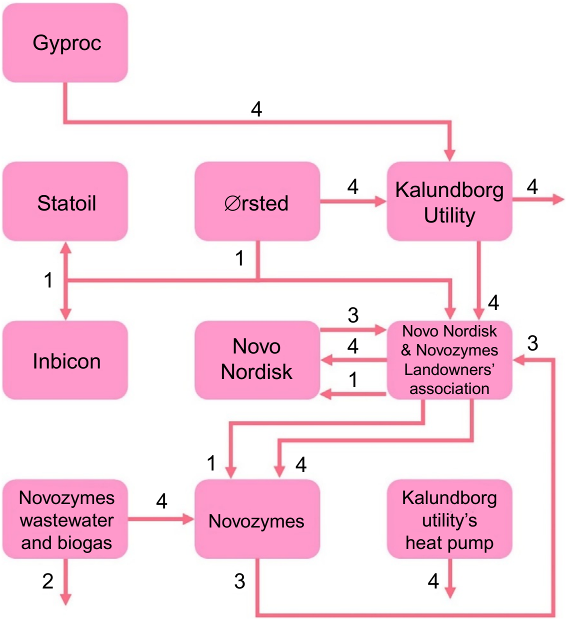

FIG. 8.3 Kalundborg symbiosis energy exchange system (Note: 1—Steam; 2—Power to the grid; 3—Warm condensate; 4—District heating).

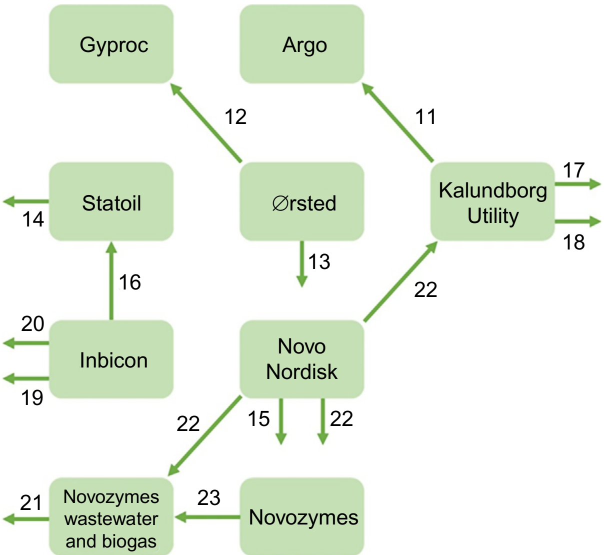

FIG. 8.4 Kalundborg symbiosis material exchange system (Note: 11—Waste; 12—Gypsum; 13—Fly ash; 14—Sulfur; 15—Slurry; 16—Bioethanol; 17—Sand; 18—Sludge; 19—C5/C6 Sugars; 20—Lignin; 21—NovoGro 30; 22—Ethanol waste; 23—Biomass).

6.2.3 Life cycle impact assessment

This step is to transform materials and energy input into equivalent impacts on the environment. The researchers have clarified impacts of each substance on the environment. Some software, such as SimaPro, openLCA, and GaBi, can help researchers to generate the impact

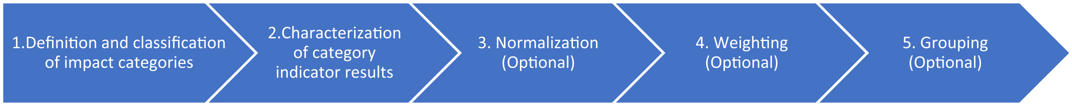

FIG. 8.5 Life cycle impact assessment.

values with respect to the inventory data based on a database. The procedures of LCI assessment include definition and classification of impact categories, characterization of category indicator results, normalization, weighting, and grouping, as shown in Fig. 8.5.

6.2.4 Interpretation

Since data cannot provide decision maker a clear view of the sustainability performance of the research object, interpretation plays an important role in evaluating and further analyzing the results data. The general procedures of interpretation can be demonstrated by Fig. 8.6.

The evaluation process is adapted to ensure that life cycle inventory and LCI assessment are conducted in a manner consistent with goal and scope definition made in the first step. This process includes a completeness check, consistency check, and so on. The completeness check and consistency check are the compulsory parts of LCA, which are designed to check for missing parts of procedures and to check for consistency with the goal and scope. The differences between the completeness check and the consistency check are shown in Fig. 8.7.

From those checks, or other methods such as MCDM methods, decision makers can obtain a clear result of the sustainability of the target process or service.

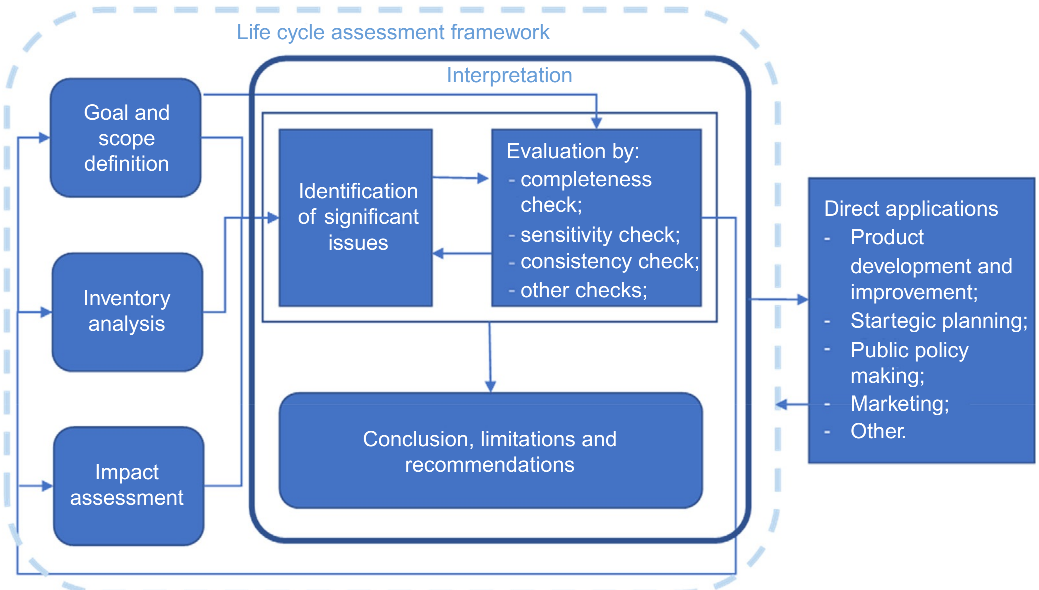

FIG. 8.6 Interpretation of life cycle assessment.

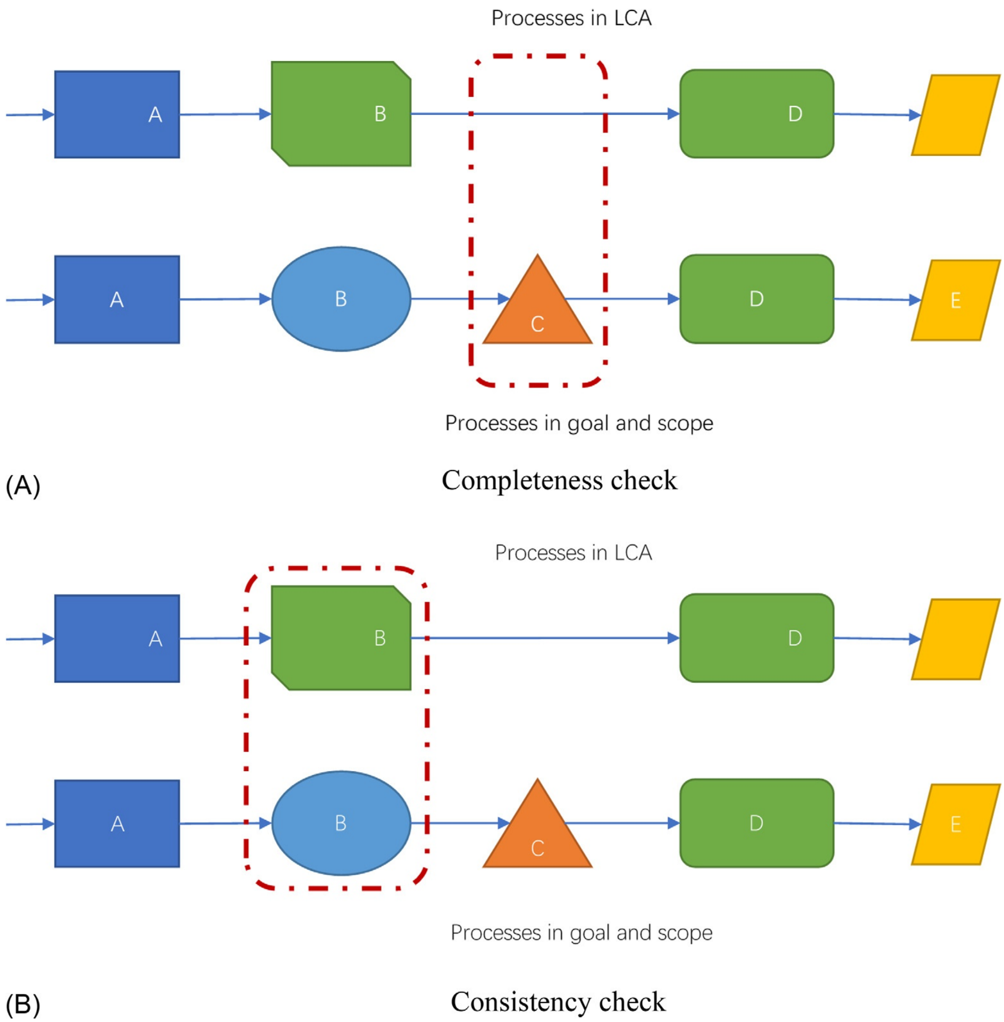

FIG. 8.7 The differences between (A) the completeness check and (B) the consistency check.

6.2.5 Life cycle costing and social life cycle assessment

The overall framework of LCC and SLCA are the same with LCA, which includes goal and scope definition, life cycle inventory, LCI assessment, and interpretation, as shown in Fig. 8.8. The LCC and SLCA of the same object should share the same research boundary. The material flows and energy flows should be replaced by cash flow for LCC and be replaced by social related indexes for SLCA. Similarly, in the impact assessment for LCC and SLCA, the data should be transformed into economic-related and social-related impact values.

As there are no international standards for LCC and SLCA, there is more space for researchers to create innovative methods or adjust the current method within the LCA framework to create more suitable sustainability assessment methods that match the practical situations.

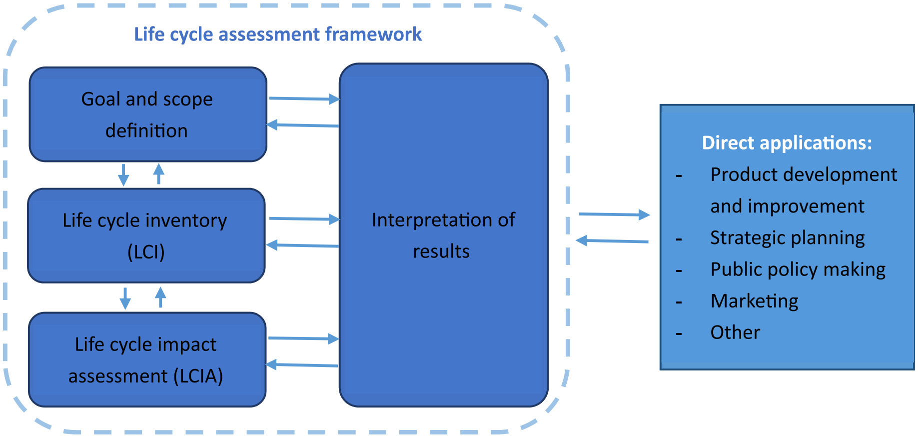

FIG. 8.8 Life cycle assessment framework.

6.3 Recent studies of LCSA

The LCSA has become popular in recent years. The peer-reviewed articles conducted with LCA, LCC, SLCA, or LCSA as the main purpose were reviewed and summarized according to the year, as shown in Fig. 8.9. The articles were searched and selected from Web of Science. The search keywords were set as "life cycle assessment," "life cycle costing," "social life cycle assessment," and "life cycle sustainability assessment" for title searching, respectively. In this chapter, only published articles were reviewed, since published articles provide more reliable information as they are reviewed by peers.

From Fig. 8.9, the increasing number of sustainability assessment researches can be observed. Sustainability assessment remains a popular topic for study and application. Although the number of articles conducting LCC, SLCA, and LCSA is small compared with that of LCA, the increasing trend of research of LCC, SLCA, and LCSA indicates the great potential for development and improvement in this area.

As LCA has the longest history among the three assessments in LCSA, its application range can, to some extent, reflect the research focuses for past years and the potential trend for LCSA in the future. The research categories of LCA for 2000 to 2018 are summarized in Fig. 8.10. The data was collected from Web of Science by title searching "life cycle assessment" for articles. The information has been categorized into nine categories that including mathematics and computing, physics and material, chemistry, sociology, marine engineering, biology and medicine, electricity, geology and civil engineering, and agriculture.

Observed from Fig. 8.10, the analysis related to electricity generation technologies, chemistry, and materials were the most popular ones for sustainability assessment. It is understandable that more studies were related to those three categories, since those three categories represent the major consumptions of our daily lives. The research trend of LCA indicates the major industries that were trying to improve their environmental

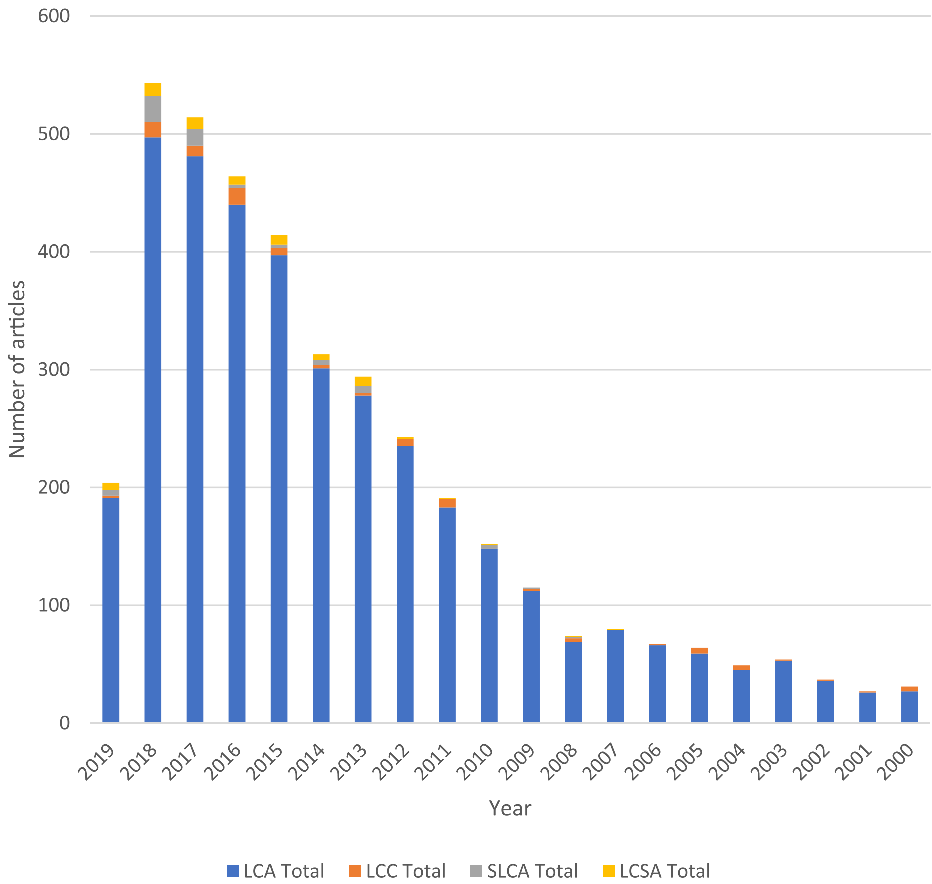

FIG. 8.9 Summarization of literatures for LCSA from 2000 to 2018.

performance in the past years. The success of LCA application in those fields provides the basis for other industries to improve their sustainability performances by using LCA and LCSA. More LCA researches are expected to be carried out for industries like marine engineering.

Although LCA has been widely used in sustainability management, the articles conducting complete LCSA are very few. To figure out the current progress of LCSA results, all 61 articles containing the keywords "life cycle sustainability assessment" in their titles were reviewed. To clarify, only scientific journal articles were selected in this chapter, because the information in the published articles that have been reviewed by peers is more reliable. Therefore, proceedings papers, reviews, editorial material, early access, and conference abstracts have been excluded from the review list. From those 61 articles, 10 articles mentioning only parts of LCSA in their article were excluded from the list. The remaining 51 articles were analyzed for the development trend of LCSA and the application fields.

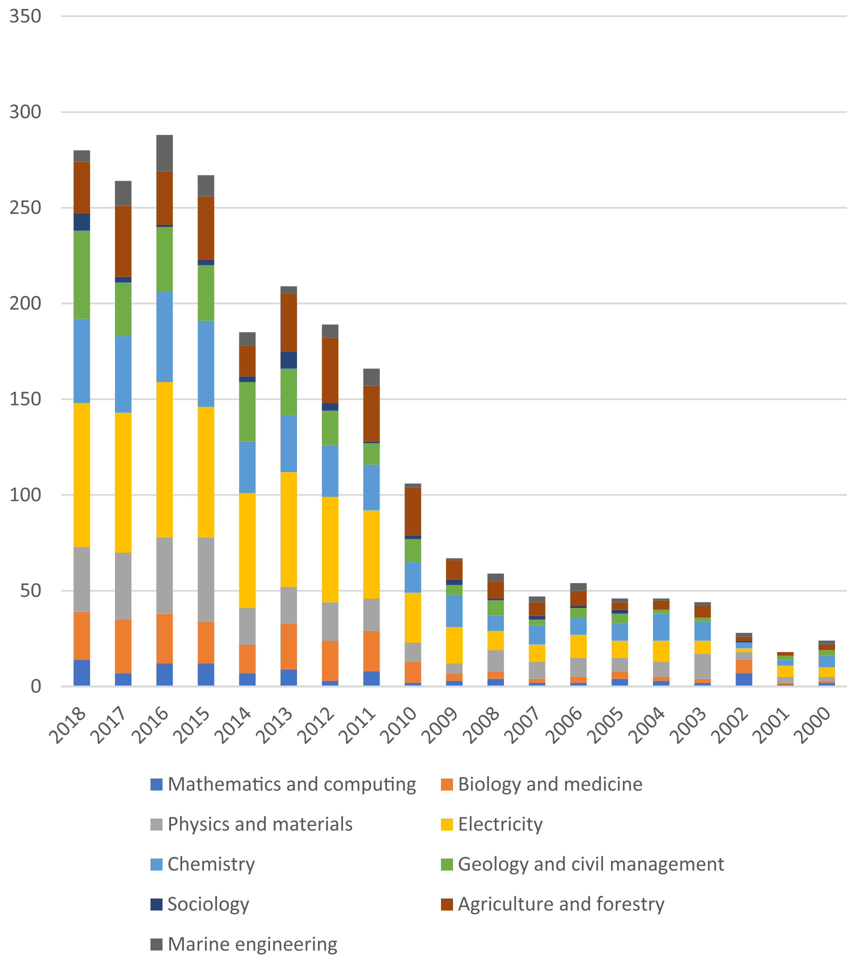

FIG. 8.10 The research categories of LCA for 2000 to 2018.

Among the remaining 51 articles, they were categorized into three different categories, which include concept, method, and analysis. The concept type article introduces sustainability concepts and reviews of relative articles without mentioning methodologies and case studies. The method type article introduces a new framework of LCSA without mentioning specific application fields and application cases. The analysis type articles have specific application objects and clear problems to be solved. To better illustrate the development trend of LCSA, the number of each type of article was summarized according to published years, as shown in Fig. 8.11.

It can be observed from Fig. 8.11 that the articles purely discussing concepts and methods were limited. Most of the articles attempted to create new methods for a certain industry or a case. The number of LCSA analyzing sustainability performances increased with the years. Therefore, although LCSA is a new field of sustainability assessment, it is observed to be a useful and feasible tool that attracts further study.

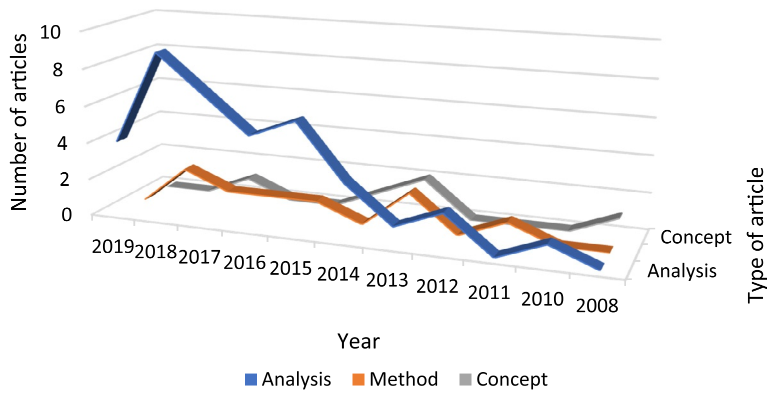

FIG. 8.11 The number of each types of articles reviewed published from 2008 to 2019.

To summarize the application fields of LCSA, LCSA literature with case studies are listed in Table 8.3 with indicating the purpose of LCSA used in this article. There are nine categories summarized from all literature including agriculture, chemistry, construction, energy, environment protection, food, logistics, manufacturing, and transportation. Among all categories, energy sources selection, construction related cases, and transportation are the three major application fields of LCSA. This application trend of LCSA matches the core application fields of LCA, as discussed previously.

TABLE 8.3 Summarization of LCSA literature with case studies.

Case category	Title	Purpose	Year	Reference
Agriculture	Simplified life cycle sustainability assessment of mangrove management: a case of plantation on wastelands in Thailand	Improvement	2010	(Moriizumi et al., 2010)
	Tiered life cycle sustainability assessment applied to a grazing dairy farm	Selection or ranking	2018	(Chen and Holden, 2018)
	Evaluation of sustainable innovations in olive growing systems: a life cycle sustainability assessment case study in southern Italy	Selection or ranking	2018	(De Luca et al., 2018)
Chemistry	Life cycle sustainability assessment of chemical processes: a vector-based three-dimensional algorithm coupled with AHP	Evaluation of technology	2017	(Xu et al., 2017)
	Life cycle sustainability assessment for sustainability improvements: a case study of high-density polyethylene production in Alberta, Canada	Improvement	2017	(Hannouf and Assefa, 2017)
Construction	Building a sustainability benchmarking framework of ceramic tiles based on life cycle sustainability assessment (LCSA)	Benchmarking	2019	(García-Muiña et al., 2019)

Continued

TABLE 8.3 Summarization of LCSA literature with case studies—cont'd

Case category	Title	Purpose	Year	Reference
	Agent-based modeling of temporal and spatial dynamics in life cycle sustainability assessment	Evaluation of technology	2017	(Wu et al., 2017)
	Life cycle sustainability assessment of RC buildings in seismic regions	Evaluation of technology	2016	(Gencturk et al., 2016)
	Integrating triple bottom line input–output analysis into life cycle sustainability assessment framework: the case for US buildings	Evaluation of technology	2014	(Onat et al., 2014a)
	Multidimensional Pareto optimization as an approach for site-specific building refurbishment solutions applicable for life cycle sustainability assessment	Evaluation of technology	2013	(Ostermeyer et al., 2013)
	Life cycle sustainability assessment of pavement maintenance alternatives: methodology and case study	Selection or ranking	2019	(Zheng et al., 2019b)
	Spatial life cycle sustainability assessment: a conceptual framework for net-zero buildings	Selection or ranking	2015	(Hossaini et al., 2015a)
	AHP based life cycle sustainability assessment (LCSA) framework: a case study of six-story wood frame and concrete frame buildings in Vancouver	Selection or ranking	2015	(Hossaini et al., 2015b)
	Life cycle sustainability assessment (LCSA) for selection of sewer pipe materials	Selection or ranking	2015	(Akhtar et al., 2015)
Energy	Multi-criteria decision making for the prioritization of energy systems under uncertainties after life cycle sustainability assessment	Selection or ranking	2018	(Ren, 2018)
	Multi-actor multi-criteria decision making for life cycle sustainability assessment under uncertainties	Selection or ranking	2018	(Ren et al., 2018)
	Prioritization of bioethanol production pathways in China based on life cycle sustainability assessment and multi-criteria decision-making	Selection or ranking	2015	(Ren et al., 2015)
	Life cycle sustainability assessment of ground source heat pump in Shanghai, China	Evaluation of technology	2016	(Huang and Mauerhofer, 2016)
	Integrated life cycle sustainability assessment: a practical approach applied to biorefineries	Evaluation of technology	2015	(Keller et al., 2015)
	Life cycle sustainability assessment of electricity generation in Pakistan: policy regime for a sustainable energy mix	Improvement	2017	(Akber et al., 2017)
	Life cycle sustainability assessment of UK electricity scenarios to 2070	Prediction	2014	(Stamford and Azapagic, 2014)

TABLE 8.3 Summarization of LCSA literature with case studies—cont'd

Case category	Title	Purpose	Year	Reference
	A new quantitative life cycle sustainability assessment framework: application to integrated energy systems	Selection or ranking	2019	(Moslehi and Reddy, 2019)
	Integrated life cycle sustainability assessment of the Greek interconnected electricity system	Selection or ranking	2019	(Roinioti and Koroneos, 2019)
	Life cycle sustainability assessment of grid-connected photovoltaic power generation: a case study of Northeast England	Selection or ranking	2018	(Li et al., 2018)
	Inclusive impact assessment for the sustainability of vegetable oil-based biodiesel Part I: linkage between inclusive impact index and life cycle sustainability assessment	Selection or ranking	2017	(Nguyen et al., 2017)
	An integrated life cycle sustainability assessment of electricity generation in Turkey	Selection or ranking	2016	(Atilgan and Azapagic, 2016)
	Solar photovoltaic development in Australia; a life cycle sustainability assessment study	Selection or ranking	2015	(Yu and Halog, 2015)
	Towards life cycle sustainability assessment: an implementation to photovoltaic modules	Selection or ranking	2012	(Traverso et al., 2012)
Environment protection	Life cycle sustainability assessment of DMSO solvent recovery from hazardous waste water	Evaluation of technology	2018	(Zajáros et al., 2018)
	Framework for life cycle sustainability assessment of municipal solid waste management systems with an application to a case study in Thailand	Selection or ranking	2012	(Menikpura et al., 2012)
Food	Framework for integrating animal welfare into life cycle sustainability assessment	Selection or ranking	2018	(Scherer et al., 2018)
Logistics	Using life cycle sustainability assessment to trade off sourcing strategies for humanitarian relief items	Selection or ranking	2017	(van Kempen et al., 2017)
Manufacturing	Lessons learned from a life cycle sustainability assessment of rare earth permanent magnets	Selection or ranking	2017	(Wulf et al., 2017)
	An exploratory investigation of additively manufactured product life cycle sustainability assessment.	Evaluation of technology	2018	(Ma et al., 2018)
Transportation	Combined application of multi-criteria optimization and life cycle sustainability assessment for optimal distribution of alternative passenger cars in the US.	Selection or ranking	2016	(Onat et al., 2016c)
	Developing life cycle sustainability assessment methodology by applying values-based sustainability weighting: tested on biomass based and fossil transportation fuels	Selection or ranking	2018	(Ekener et al., 2018)

Continued

TABLE 8.3 Summarization of LCSA literature with case studies—cont'd

Case category	Title	Purpose	Year	Reference
	Uncertainty-embedded dynamic life cycle sustainability assessment framework: an ex-ante perspective on the impacts of alternative vehicle options	Selection or ranking	2016	(Onat et al., 2016b)
	Towards life cycle sustainability assessment of alternative passenger vehicles	Selection or ranking	2014	(Onat et al., 2014b)

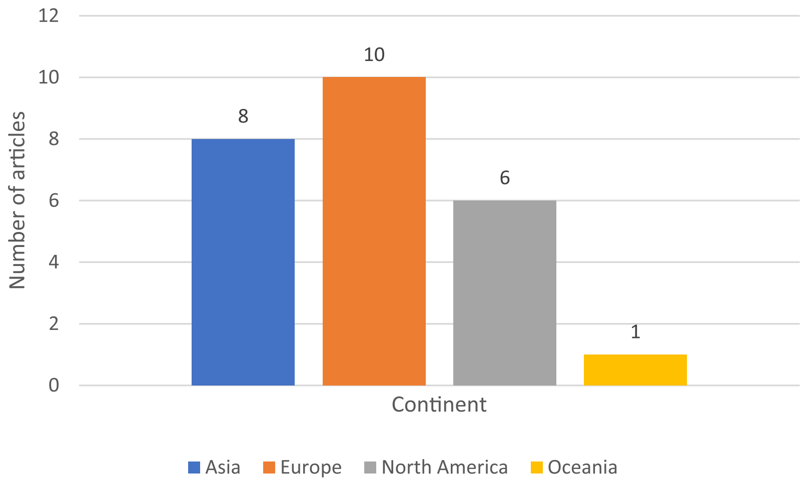

FIG. 8.12 The summarization of regions of LCSA studies.

To analyze the case studies of LCSA further, the regions for LCSA application were counted as shown in Fig. 8.12. From the 51 reviewed articles, the research regions include Australia, China, Germany, Greece, Hungary, Ireland, Italy, Kenya, Pakistan, Thailand, Turkey, the United Kingdom, the United States, and Canada. It is observed that European countries and Asian countries realize the reliability of LCSA to assess sustainability more than other areas. However, since the case studies, especially for certain countries, were few, this conclusion cannot be made. It is true that LCSA has great potential to be revised and applied in more cases for sustainability analysis.

6.4 Conclusions

LCSA which involves LCA for environmental assessment, LCC for economic analysis, and SLCA for social performance evaluation was raised to compensate for the shortage, and limited research perspectives, of LCA. In this chapter, the framework of LCA was introduced in detail with case studies, and differences between the frameworks of LCC and SLCA and that of LCA were emphasized. Then the relevant articles of LCSA were reviewed and summarized for research trend analysis. The analysis of literature related to LCSA led to the conclusions that:

- more work can be done to revise and modify the LCSA frameworks for application in more complex environments;
- LCSA is a useful and feasible tool that will attract more studies in the future; and
- LCSA has great potential to be revised and applied in more cases for sustainability analysis.

References

Akber, M.Z., Thaheem, M.J., Arshad, H., 2017. Life cycle sustainability assessment of electricity generation in Pakistan: policy regime for a sustainable energy mix. Energy Policy. https://doi.org/10.1016/j.enpol.2017.09.022.

Akhtar, S., Reza, B., Hewage, K., et al., 2015. Life cycle sustainability assessment (LCSA) for selection of sewer pipe materials. Clean Techn. Environ. Policy 17. https://doi.org/10.1007/s10098-014-0849-x.

Arena, U., Mastellone, M.L., Perugini, F., 2003. The environmental performance of alternative solid waste management options: a life cycle assessment study. Chem. Eng. J. https://doi.org/10.1016/j.cej.2003.08.019.

Atilgan, B., Azapagic, A., 2016. An integrated life cycle sustainability assessment of electricity generation in Turkey. Energy Policy 93, 168–186. https://doi.org/10.1016/j.enpol.2016.02.055.

Chen, W., Holden, N.M., 2018. Tiered life cycle sustainability assessment applied to a grazing dairy farm. J. Clean. Prod. 172, 1169–1179. https://doi.org/10.1016/j.jclepro.2017.10.264.

De Luca, A.I., Falcone, G., Stillitano, T., et al., 2018. Evaluation of sustainable innovations in olive growing systems: a life cycle sustainability assessment case study in southern Italy. J. Clean. Prod. 171, 1187–1202. https://doi.org/10.1016/j.jclepro.2017.10.119.

Ekener, E., Hansson, J., Larsson, A., Peck, P., 2018. Developing life cycle sustainability assessment methodology by applying values-based sustainability weighting—tested on biomass based and fossil transportation fuels. J. Clean. Prod. 181, 337–351. https://doi.org/10.1016/j.jclepro.2018.01.211.

Ferrari, A., Volpi, L., Pini, M., et al., 2019. Building a sustainability benchmarking framework of ceramic tiles based on life cycle sustainability assessment (LCSA). Resources. https://doi.org/10.3390/resources8010011.

García-Muiña, F., Volpi, L., Ferrari, A., et al., 2019. Building a sustainability benchmarking framework of ceramic tiles based on life cycle sustainability assessment (LCSA). Resources 8, 11. https://doi.org/10.3390/resources8010011.

Gencturk, B., Hossain, K., Lahourpour, S., 2016. Life cycle sustainability assessment of RC buildings in seismic regions. Eng. Struct. https://doi.org/10.1016/j.engstruct.2015.11.037.

Gulipac, S., 2016. Industrial symbiosis: building on Kalundborg's waste management experience. Renew. Energy Focus.

Hannouf, M., Assefa, G., 2017. Life cycle sustainability assessment for sustainability improvements: a case study of high-density polyethylene production in Alberta, Canada. Sustainability. https://doi.org/10.3390/su9122332.

Heijungs, R., Huppes, G., Guinée, J.B., 2010. Life cycle assessment and sustainability analysis of products, materials and technologies. Toward a scientific framework for sustainability life cycle analysis. Polymer Degradation and stability 95 (3), 422–428. https://doi.org/10.1016/j.polymdegradstab.2009.11.010.

Hossaini, N., Hewage, K., Sadiq, R., 2015a. Spatial life cycle sustainability assessment: a conceptual framework for net-zero buildings. Clean Techn. Environ. Policy. https://doi.org/10.1007/s10098-015-0959-0.

Hossaini, N., Reza, B., Akhtar, S., et al., 2015b. AHP based life cycle sustainability assessment (LCSA) framework: a case study of six storey wood frame and concrete frame buildings in Vancouver. J. Environ. Plan. Manag. https://doi.org/10.1080/09640568.2014.920704.

Huang, B., Mauerhofer, V., 2016. Life cycle sustainability assessment of ground source heat pump in Shanghai, China. J. Clean. Prod. https://doi.org/10.1016/j.jclepro.2015.08.048.

Jacobsen, N.B., 2006. Industrial symbiosis in Kalundborg, Denmark: a quantitative assessment of economic and environmental aspects. J. Ind. Ecol. https://doi.org/10.1162/108819806775545411.

John, E., Nicholas, G., 2018. Industrial ecology in practice: the evolution of interdependence at Kalundborg. J. Ind. Ecol. https://doi.org/10.1162/jiec.1997.1.1.67.

Keller, H., Rettenmaier, N., Reinhardt, G.A., 2015. Integrated life cycle sustainability assessment—a practical approach applied to biorefineries. Appl. Energy 154, 1072–1081. https://doi.org/10.1016/j.apenergy.2015.01.095.

Kouloumpis, V., Azapagic, A., 2018. Integrated life cycle sustainability assessment using fuzzy inference: a novel FELICITA model. Sustain. Produc. Consum. 15, 25–34. https://doi.org/10.1016/j.spc.2018.03.002.

Li, T., Roskilly, A.P., Wang, Y., 2018. Life cycle sustainability assessment of grid-connected photovoltaic power generation: a case study of Northeast England. Appl. Energy. https://doi.org/10.1016/j.apenergy.2017.07.021.

Ma, J., Harstvedt, J.D., Dunaway, D., et al., 2018. An exploratory investigation of additively manufactured product life cycle sustainability assessment. J. Clean. Prod. https://doi.org/10.1016/j.jclepro.2018.04.249.

Menikpura, S.N.M., Gheewala, S.H., Bonnet, S., 2012. Framework for life cycle sustainability assessment of municipal solid waste management systems with an application to a case study in Thailand. Waste Manag. Res. https://doi.org/10.1177/0734242X12444896.

Moriizumi, Y., Matsui, N., Hondo, H., 2010. Simplified life cycle sustainability assessment of mangrove management: a case of plantation on wastelands in Thailand. J. Clean. Prod. https://doi.org/10.1016/j.jclepro.2010.07.017.

Moslehi, S., Reddy, T.A., 2019. A new quantitative life cycle sustainability assessment framework: application to integrated energy systems. Appl. Energy. https://doi.org/10.1016/j.apenergy.2019.01.237.

Nguyen, T.A., Kuroda, K., Otsuka, K., 2017. Inclusive impact assessment for the sustainability of vegetable oil-based biodiesel—Part I: Linkage between inclusive impact index and life cycle sustainability assessment. J. Clean. Prod. https://doi.org/10.1016/j.jclepro.2017.08.059.

Onat, N.C., Kucukvar, M., Tatari, O., 2016a. Uncertainty-embedded dynamic life cycle sustainability assessment framework: an ex-ante perspective on the impacts of alternative vehicle options. Energy. https://doi.org/10.1016/j.energy.2016.06.129.

Onat, N.C., Kucukvar, M., Tatari, O., 2014a. Integrating triple bottom line input-output analysis into life cycle sustainability assessment framework: the case for US buildings. Int. J. Life Cycle Assess. https://doi.org/10.1007/s11367-014-0753-y.

Onat, N.C., Kucukvar, M., Tatari, O., 2014b. Towards life cycle sustainability assessment of alternative passenger vehicles. Sustainability. https://doi.org/10.3390/su6129305.

Onat, N.C., Kucukvar, M., Tatari, O., Zheng, Q.P., 2016c. Combined application of multi-criteria optimization and life-cycle sustainability assessment for optimal distribution of alternative passenger cars in U.S. J. Clean. Prod. 112, 291–307. https://doi.org/10.1016/j.jclepro.2015.09.021.

Ostermeyer, Y., Wallbaum, H., Reuter, F., 2013. Multidimensional Pareto optimization as an approach for site-specific building refurbishment solutions applicable for life cycle sustainability assessment. Int. J. Life Cycle Assess. https://doi.org/10.1007/s11367-013-0548-6.

Ren, J., 2018. Multi-criteria decision making for the prioritization of energy systems under uncertainties after life cycle sustainability assessment. Sustain. Produc. Consum. https://doi.org/10.1016/j.spc.2018.06.005.

Ren, J., Manzardo, A., Mazzi, A., et al., 2015. Prioritization of bioethanol production pathways in China based on life cycle sustainability assessment and multicriteria decision-making. Int. J. Life Cycle Assess. 20, 842–853. https://doi.org/10.1007/s11367-015-0877-8.

Ren, J., Ren, X., Dong, L., et al., 2018. Multiactor multicriteria decision making for life cycle sustainability assessment under uncertainties. AICHE J. 64, 2103–2112. https://doi.org/10.1002/aic.16149.

Roinioti, A., Koroneos, C., 2019. Integrated life cycle sustainability assessment of the Greek interconnected electricity system. Sustainable Energy Technol. Assess. 32, 29–46. https://doi.org/10.1016/j.seta.2019.01.003.

Scherer, L., Tomasik, B., Rueda, O., Pfister, S., 2018. Framework for integrating animal welfare into life cycle sustainability assessment. Int. J. Life Cycle Assess. https://doi.org/10.1007/s11367-017-1420-x.

Stamford, L., Azapagic, A., 2014. Life cycle sustainability assessment of UK electricity scenarios to 2070. Energy Sustain. Dev. 23, 194–211. https://doi.org/10.1016/j.esd.2014.09.008.

Symbiosis Centre Denmark, 2010. Explore the Kalundborg Symbiosis. Kalundborg Symbiosis. Available at http://www.symbiosis.dk/en/ (Accessed April 17, 2018).

Traverso, M., Asdrubali, F., Francia, A., Finkbeiner, M., 2012. Towards life cycle sustainability assessment: an implementation to photovoltaic modules. Int. J. Life Cycle Assess. https://doi.org/10.1007/s11367-012-0433-8.

Valentine, S.V., 2016. Kalundborg Symbiosis: fostering progressive innovation in environmental networks. J. Clean. Prod. https://doi.org/10.1016/j.jclepro.2016.01.061.

van Kempen, E.A., Spiliotopoulou, E., Stojanovski, G., de Leeuw, S., 2017. Using life cycle sustainability assessment to trade off sourcing strategies for humanitarian relief items. Int. J. Life Cycle Assess. https://doi.org/10.1007/s11367-016-1245-z.

Wu, S.R., Li, X., Apul, D., et al., 2017. Agent-based modeling of temporal and spatial dynamics in life cycle sustainability assessment. J. Ind. Ecol. https://doi.org/10.1111/jiec.12666.

Wulf, C., Zapp, P., Schreiber, A., et al., 2017. Lessons learned from a life cycle sustainability assessment of rare earth permanent magnets. J. Ind. Ecol. https://doi.org/10.1111/jiec.12575.

Xu, D., Lv, L., Ren, J., et al., 2017. Life cycle sustainability assessment of chemical processes: a vector-based three-dimensional algorithm coupled with AHP. Ind. Eng. Chem. Res. 56, 11216–11227. https://doi.org/10.1021/acs.iecr.7b02041.

Yu, M., Halog, A., 2015. Solar photovoltaic development in Australia—a life cycle sustainability assessment study. Sustainability. https://doi.org/10.3390/su7021213.

Zajáros, A., Szita, K., Matolcsy, K., Horváth, D., 2018. Life cycle sustainability assessment of DMSO solvent recovery from hazardous waste water. Period. Polytech., Chem. Eng. https://doi.org/10.3311/PPch.11097.

Zheng, X., Easa, S.M., Yang, Z., et al., 2019. Life-cycle sustainability assessment of pavement maintenance alternatives: methodology and case study. J. Clean. Prod. https://doi.org/10.1016/j.jclepro.2018.12.227.

CHAPTER

9

Life cycle decision support framework: Method and case study

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9.1 Introduction

Life cycle sustainability assessment (LCSA) is a well-known measurement tool for any service, industrial system, or manufacturing process to standardize and quantify its impacts in environmental, economic, and social aspects (Ren et al., 2015; Wulf et al., 2018). The three individual assessments for environmental, economic, and social analysis, respectively, contained in LCSA are life cycle assessment (LCA), life cycle costing (LCC), and social life cycle assessment (SLCA) (Berriel et al., 2018; Ioppolo et al., 2019). Those assessment methods are instructed by international standards (ISO14040, ISO14044, ISO14047-ISO14049, and ISO14072 for LCA, and ISO15663 for LCC), so that they provide scientific and standardized results for research objects with respect to their impacts on environment, finance, and society. Therefore, LCSA has been widely used in industries such as construction (Caruso et al., 2017; Balasbaneh et al., 2018), transportation (Onat et al., 2016; Ekener et al., 2018), energy generation (Moslehi and Reddy, 2019; Roinioti and Koroneos, 2019), and other industries to assess their sustainability.

However, problems were faced when a comparative LCSA study was studied. The results of LCSA cannot be directly compared for ranking alternatives, since it is hard to judge when each alternative has its own strengths and drawbacks in most cases. For example, in a comparative LCSA study of urban water reuse (Opher et al., 2018a), the no-reuse scenarios score best in LCA, while the semidistributed reclamation performs the best in both LCC analysis and SLCA analysis. It is hard to say the semidistributed reclamation is the best option, because the environmental performance of the semidistributed reclamation is far inferior to the no-use reclamation. In this case, the authors (Opher et al., 2018a) used a decision making model to assist the ranking process.

Life Cycle Sustainability Assessment for Decision-Making
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To deal with the problem, multicriteria decision making (MCDM) methods were adopted in the decision making process after LCSA results for several technologies, processes, or services were provided, because MCDM methods can help to prioritize alternatives or to select the best option based on data of multiple criteria provided (Yu et al., 2018; Ioppolo et al., 2019). In addition, the hierarchical structure of LCSA results (with multiple indicators under three pillars) fits the MCDM processing structure perfectly. Thus, the combination of LCSA and MCDM compensates for the drawbacks of both LCSA and MCDM: improves the data quality and provides a direct decision making result based on all-round sustainability assessment. Many studies have applied MCDM combined with LCSA data and these methods have been used in the construction industry (San-José Lombera and Cuadrado Rojo, 2010; Shahriar et al., 2014), transportation (Bojković et al., 2010; Awasthi et al., 2011), energy generation (Zhang et al., 2015; Zhao and Li, 2016), supply chain (Entezaminia et al., 2016; Luthra et al., 2017a), and manufacturing (He et al., 2019).

In this chapter, the procedures and recent studies of LCSA are summarized in Section 9.2. The MCDM methods used in competitive case studies based on LCSA results are also reviewed and summarized in Section 9.2. A group Z-number best worst method (group ZBWM) combined with the goal programming method is developed and introduced in Section 9.3. The proposed method is adapted to analyze a case study regarding waste oil management technologies selection in Section 9.4, and the results and discussions are illustrated afterward. The conclusions are summarized in Section 9.5.

9.2 Literature reviews

The combination of LCSA and MCDM mutually compensates for the drawbacks of each other. After the LCSA, decision makers find it hard to make the decisions because of inexistence of the most superior alternative, which performs the best in every aspect. MCDM is a useful tool to rank, or select multiple alternatives based on data of multiple criteria (Ishizaka and Siraj, 2018). Therefore, MCDM can help to determine the best alternative based on LCSA results, while LCSA provides a scientific and reliable database for decision making, as shown in Fig. 9.1.

The MCDM method include four main steps: criteria system determination, decision making matrix determination, criteria weighting, and aggregation (Ho et al., 2010).

9.2.1 Criteria system determination

The first step is to select criteria from LCSA indicators that can better describe the options. Criteria in environmental, economic, and social perspectives are selected from LCA, LCC, and LCSA, respectively, to analyze the target. The selection of criteria depends on the situation of industry and the knowledge from experts.

9.2.2 Decision making matrix determination

Prior to the weighting method, a decision-making matrix should be built summarizing all information of alternatives with regard to criteria adapted from the criteria system. As for all

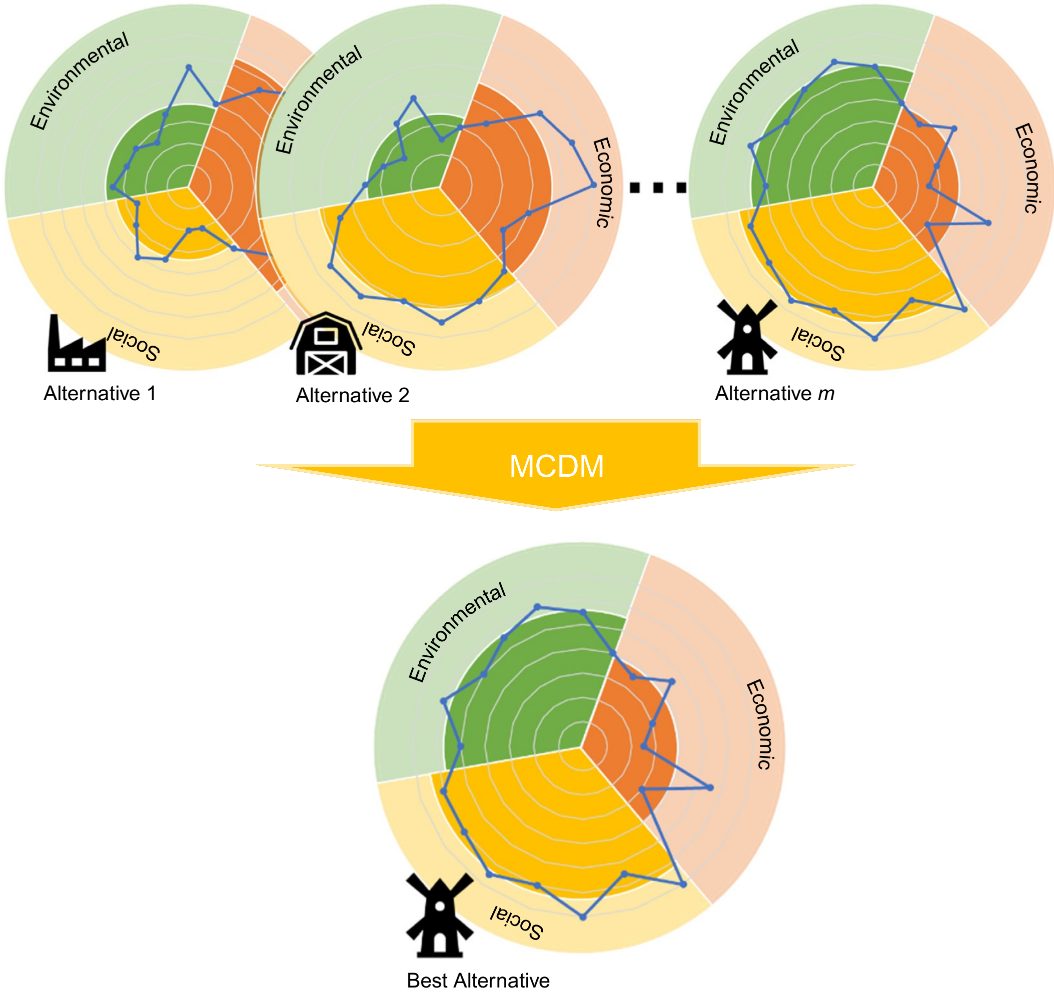

FIG. 9.1 The combination of LCSA and MCDM.

alternatives (a_1, a_2, …, a_n), the decision-making matrix (X) contains the information of data (x_{ij}) of ith criterion with respect to jth alternative where $i=1, 2, \ldots, m$, and $j=1, 2, \ldots, n$, as shown by Eq. (9.1).

$$X = \begin{array}{c} \\ c_1 \\ c_2 \\ \cdots \\ c_m \end{array} \begin{array}{c} \begin{array}{cccc} a_1 & a_2 & \cdots & a_n \end{array} \\ \begin{pmatrix} x_{11} & x_{12} & \cdots & x_{1n} \\ x_{21} & x_{22} & \cdots & x_{2n} \\ \vdots & \vdots & \ddots & \vdots \\ x_{m1} & x_{m2} & \cdots & x_{mn} \end{pmatrix} \end{array} \begin{array}{c} \\ w_1 \\ w_2 \\ \vdots \\ w_m \end{array} \tag{9.1}$$

To engage more possibilities in practice, the data of criteria with respect to multiple alternatives have been extended into different formats.

Some studies realized that uncertainties generated by fluctuation of data regarding time change, and human judgment buffer left because of knowledge limitations, impacted the results of decision making. To illustrate the uncertainties of values, the interval numbers, fuzzy numbers, and rough numbers were adapted in the MCDM methods. The extended methods revised the crisp number (x_{ij}) in decision-making matrix into interval number ($\widetilde{x}_{ij} = \left(x^l, x^u\right)$, where x^l is the lower limit and x^u is the upper limit of this interval number), triangular fuzzy number ($\widetilde{x}_{ij} = \left(x^l, x^m, x^u\right)$, where x^l is the lower limit, x^m is the most possible number, and x^u is the upper limit of this interval number), and other numbers that can express uncertainties. Similarly, the criteria weights can be revised to use fuzzy numbers or rough numbers to involve uncertainties.

9.2.3 Criteria weighting

Criteria weighting is one of the important steps of MCDM to determine the weights of criteria according to different preferences from decision makers. Many studies have been carried out to develop MCDM and to adapt MCDM to LCSA analysis based on real-life case studies. Summarized from the literature, the criteria weighting methods adapted in the combination of MCDM and LCSA method in 2005–2019 are shown in Table 9.1.

As observed from Table 9.1, analytical hierarchy analysis (AHP) raised by Saaty (1987) is the classic but most popular weighting method. Furthermore, some adaptions have been made from original version AHP to make it suitable in more situations. In addition, some studies have extended MCDM to be applied in interval number, fuzzy number, rough number, and other types of number to take reservations of decision makers into consideration.

TABLE 9.1 Weighting methods used in sustainability analysis.

Method	Reference
Crisp number	
AHP	Alwaer and Clements-Croome (2010), Castillo and Pitfield (2010), San-José Lombera and Cuadrado Rojo (2010), Awasthi and Chauhan (2011), Turskis and Zavadskas (2011), Awasthi and Chauhan (2012), Dai and Blackhurst (2012), Del Caño et al. (2012), Zavadskas et al. (2012), Jones et al. (2013), Palevičius et al. (2013), Raslanas et al. (2013), Šiožinytė et al. (2014), Akhtar et al. (2015), Ren et al. (2015), Barić et al. (2016), De La Fuente et al. (2016), Entezaminia et al. (2016), Al Garni and Awasthi (2017), Büyüközkan and Karabulut (2017), Das and Shaw (2017), de la Fuente et al. (2017), Gao et al. (2017), Inti and Tandon (2017), Luthra et al. (2017a), Rashid et al. (2017), Xu et al. (2017), Alhumaid et al. (2018), De Luca et al. (2018), Díaz-Cuevas et al. (2018), Mirjat et al. (2018), Opher et al. (2018b), Tang et al. (2018), Yazdani et al. (2018), Zheng et al. (2019)
BWM	Rezaei et al. (2016), Kusi-Sarpong et al. (2018), Liu et al. (2018)
ANP	Tsai et al. (2013), Ozcan-Deniz and Zhu (2015), Zhang et al. (2015)

TABLE 9.1 Weighting methods used in sustainability analysis—cont'd

Method	Reference
Entropy	Tahmasebi Birgani and Yazdandoost (2018), Tang et al. (2018)
Delphi	Bueno Cadena and Vassallo Magro (2015), Luthra et al. (2017b)
Gray-based weighting method	Manzardo et al. (2012), Su et al. (2016), Jiang et al. (2018)
DEMATEL	Tsai et al. (2013), Zhang et al. (2015), Debnath et al. (2017), Huang et al. (2018), Khoshnava et al. (2018), Pamucar et al. (2018)
DEMATEL-ANP	Dimić et al. (2016), Govindan et al. (2016)
Revised AHP	
Adaptive AHP	Tahmasebi Birgani and Yazdandoost (2018)
REMBRANDT	López and Monzón (2010), Bueno Cadena and Vassallo Magro (2015)
AHP-EW	Wang et al. (2019)
DEMATEL-AHP	Hsu et al. (2014), Kuo et al. (2015)
HEL-AHP	Büyüközkan and Karabulut (2017)
Fuzzy	
FEAHP	Akadiri et al. (2013)
FPP	Fallahpour et al. (2017)
Fuzzy Delphi	Hsu et al. (2014), Dimić et al. (2016), Zhao and Li (2016), Zečević et al. (2017)
ASPID	Jovanovic et al. (2010), Škobalj et al. (2017)
Fuzzy AHP	Azadnia et al. (2015), Ren et al. (2016), Sánchez-Lozano et al. (2016), Inti and Tandon (2017), Sivaraja and Sakthivel (2017), Khoshnava et al. (2018), Padhi et al. (2018), Zhang et al. (2018)
Fuzzy ANP	Ren et al. (2016), Zečević et al. (2017)
Fuzzy SWARA	Mavi et al. (2017)
Fuzzy DEMATEL	Luthra et al. (2017b)
Fuzzy Entropy	Zhao and Guo (2014)
Fuzzy multiplicative AHP	Padhi et al. (2018)
Rough	
Rough BWM	Pamucar et al. (2017), Stević et al. (2018)
R'AMATEL	Chatterjee et al. (2018)

Note: AHP, *analytical hierarchy analysis;* BWM, *best worst method;* ANP, *analytical network analysis;* DEMATEL, *decision-making trial and evaluation laboratory;* REMBRANDT, *ratio estimations in magnitudes or deci-bells to rate alternatives which are non-dominated technique;* FPP, *fuzzy preference programming;* ASPID, *analysis and synthesis of parameters under information deficiency method;* SWARA, *step wise weight assessment ratio analysis;* R'AMATEL, *rough DEMATEL.*

9.2.4 Aggregation

Aggregation is the final step of MCDM to determine the rank or the selection result of the decision-making problem. In this step, a model is applied to integrate the criteria data with criteria weights to generate a clear result of priority of alternatives. To study the development trend of aggregation methods used in studies combining MCDM and LSCA, the ranking or aggregating methods are summarized in Table 9.2.

From Table 9.2, the technique for order preference by similarity to an ideal solution (TOPSIS) method (Yoon and Hwang, 1995) and its interval version are the most acceptable and popularly used in the case studies. Similar to the development of weighting methods, some studies have extended MCDM to be applied in interval number, fuzzy number, rough number, and other types of number to take uncertainties into consideration.

TABLE 9.2 Aggregating methods used in sustainability assessment.

Method	Reference
Crisp number	
WSM/SMART/SAW	Castillo and Pitfield (2010), Jeon et al. (2010), Simongáti (2010), Akadiri et al. (2013), Palevičius et al. (2013), Akhtar et al. (2015), Klein and Whalley (2015), Marzouk and Elmesteckawi (2015), Ozcan-Deniz and Zhu (2015), Mitropoulos and Prevedouros (2016), Osorio-Tejada et al. (2017), Rashidi et al. (2017), Ren et al. (2017a), Opher et al. (2018b), Roinioti and Koroneos (2019)
TOPSIS	Palevičius et al. (2013), Šiožinytė et al. (2014), Validi et al. (2014), Formisano and Mazzolani (2015), Terracciano et al. (2015), Govindan et al. (2016), Baležentis and Streimikiene (2017), Gao et al. (2017), Rashid et al. (2017), Ren et al. (2017a), Škobalj et al. (2017), Jia et al. (2018), Tahmasebi Birgani and Yazdandoost (2018), Tang et al. (2018), Yazdani et al. (2018)
GRA	Manzardo et al. (2012), Šiožinytė et al. (2014), Zhao and Li (2016)
VIKOR	Vučijak et al. (2013), Hsu et al. (2014), Formisano and Mazzolani (2015), Kuo et al. (2015), Ren et al. (2015), Zhao and Li (2016), Büyüközkan and Karabulut (2017), Luthra et al. (2017a), Huang et al. (2018), Zheng et al. (2019)
ELECTRE I/IV/IS/II/III/IV/TRI	Bojković et al. (2010), Khalili and Duecker (2013), Barata et al. (2014), Formisano and Mazzolani (2015)
PROMETHEE I/II	Safaei Mohamadabadi et al. (2009), Tsoutsos et al. (2009), Simongáti (2010), Hayashi et al. (2016), Ren et al. (2016), Gao et al. (2017), Alhumaid et al. (2018), Mahbub et al. (2018), Zhang et al. (2018)
WASPAS	Zhang et al. (2015), Baležentis and Streimikiene (2017)
COPRAS	Palevičius et al. (2013), Zhang et al. (2015), Büyüközkan and Karabulut (2017)
ARAS	Baležentis and Streimikiene (2017)
MABAC	Debnath et al. (2017), Wang et al. (2019)
MAIRCA	Pamucar et al. (2018)
MIVES	San-José Lombera and Cuadrado Rojo (2010), Del Caño et al. (2012), Pons and De La Fuente (2013), Amin Hosseini et al. (2016), De La Fuente et al. (2016), Pujadas et al. (2017)

TABLE 9.2 Aggregating methods used in sustainability assessment—cont'd

Method	Reference
SWING	Vučijak et al. (2013), Hayashi et al. (2016)
SCORE	Diaz-Balteiro et al. (2017)
CRITIC	Gao et al. (2017)
LCASI	Ren (2018)
Fuzzy number	
Fuzzy TOPSIS	Castillo and Pitfield (2010), Awasthi et al. (2011), Awasthi and Chauhan (2012), Govindan et al. (2013), Zhao and Guo (2014), Guo and Zhao (2015), Sánchez-Lozano et al. (2016), Sivaraja and Sakthivel (2017), Das and Shaw (2017), Fallahpour et al. (2017), Ren and Toniolo (2018), Jia et al. (2018), Padhi et al. (2018), Karunathilake et al. (2019)
Fuzzy VIKOR	Sivaraja and Sakthivel (2017), Zečević et al. (2017), Padhi et al. (2018)
Fuzzy ELECTRE	Sivaraja and Sakthivel (2017), Padhi et al. (2018)
MULTIMOORA	Škobalj et al. (2017), Liu et al. (2018)
Fuzzy MOORA	Mavi et al. (2017)
Fuzzy SMART	Padhi et al. (2018)
Fuzzy MIVES	De La Fuente et al. (2016)
Fuzzy MAGDM	Rao et al. (2015)
Fuzzy ARAS	Turskis and Zavadskas (2011)
WASPAS-SVNS	Zavadskas et al. (2015)
Interval SWM	Ren and Toniolo (2018)
ZOGP	Tsai et al. (2013)
Rough number	
Rough WASPAS	Stević et al. (2018)
Rough MAIRCA	Pamucar et al. (2017), Chatterjee et al. (2018)
Stochastic	
Stochastic fuzzy AHP	Promentilla et al. (2018)
TBL-LCA	Kucukvar et al. (2014)
Other methods	
Diagraph model	Ren et al. (2017a)
SNT	Halog and Manik (2011)
Regret-based analysis	Rezaei et al. (2019)
Extension theory	Ren et al. (2017)
Graph theory	He et al. (2019)

Note: WSM, *weighted sum method;* SWM, *simple weighting method;* SMART, *simple multi-attribute rating technique;* GRA, *gray relational analysis;* TOPSIS, *technique for order preference by similarity to an ideal solution;* VIKOR, *multi-criteria optimization and compromise solution;* ELECTRE, *elimination and choice expressing reality;* PROMETHEE, *preference ranking organization method for enrichment of evaluations;* WASPAS, *weighted aggregated sum product assessment method;* COPRAS, *multiple criteria complex proportional assessment;* ARAS, *additive ratio assessment method;* MABAC, *multi-attributive border approximation area comparison;* MIVES, *integrated value model for the sustainability assessment;* MOORA, *multi-objective optimization on the basis of ratio analysis;* ZOGP, *a zero–one goal programming;* CRITIC, *criteria importance through inter-criteria correlation;* LCASI, *life cycle aggregated sustainability index;* SNT, *sustainability network theory.*

Except for the uncertainty considered in MCDM, MCDM used in further analysis of alternatives based on LCSA results has adapted the group decision making concept. The group MCDM can acquire all preferences from multiple stakeholders and reflect them in the decision-making results. For example, group interval BWM and interval TOPDIM have been adapted in hydrogen production technique selection (Ren et al., 2018), group interval AHP has been proposed in sustainability assessment framework (Ren et al., 2017), and group GRA has been raised for hydrogen technologies selections as well (Manzardo et al., 2012).

To illustrate better the process of MCDM based on LCSA, a case study regarding oil management is studied by a group fuzzy MCDM method in the next section.

9.3 Methodology

In this section, a new group MCDM method is introduced. In order to deal with hesitations happened during the judgment process by decision makers, the ZBWM (Aboutorab et al., 2018) was adapted and revised as the weighting method to transform the linguistic preferences of criteria with respect to different decision makers into numerical fuzzy criteria weights. In this revised version, group opinions were considered with various opinions from multiple stakeholders. Thereafter, the goal programming method was adapted as the aggregating method to help aggregate integrated criteria preference and the actual performance of each option. Those options can be ranked according to the scores generated from the proposed method. The detailed methodology, calculation process, and result discussions were provided accordingly below.

9.3.1 Step 1. Determine the criteria system

The first step is to select criteria from LCSA indicators that can better describe the options. Criteria in environmental, economic, and social perspectives are selected from LCA, LCC, and LCSA, respectively, and the hierarchical structure of the criteria system is built in this step. The selection of criteria depends on the situation of industry and the knowledge from experts.

9.3.2 Step 2. Determine the decision-making matrix

As for all alternatives ($a_1, a_2, \ldots, a_n$), the decision-making matrix (X) contains the information of data (x_{ij}) of ith criterion with respect to jth alternative where $i = 1, 2, \ldots, m$, and $j = 1, 2, \ldots, n$, as shown by Eq. (9.1).

The data x_{ij} of criteria are collected from scientific and reliable information resources, for instance, the LCA, LCC, and SLCA results with the same research boundary. The criteria weights indicate the importance of each criterion in the analysis. To avoid large changes in scores due to the unit of weights; the weights (w_i) of ith criterion, as shown in Eq. (9.1), should satisfy Eq. (9.2).

$$\sum_i w_j = 1 \tag{9.2}$$

9.3.3 Step 3. Determine the criteria weights based on opinions of different decision makers by group ZBWM

The criteria weights in this study were determined by multiple stakeholders by using the group ZBWM. The BWM (Rezaei, 2015) is a weighting method to weight criteria by pairwise comparison. However, unlike AHP, BWM simplifies the process by comparing all criteria with the best criterion and comparing all criteria with the worst criterion, instead of comparing every pair of criteria. In this case, the BWM has been widely adapted in ranking and selection in industries such as marketing (Cohen, 2009), energy generation (van de Kaa et al., 2017), and supply chain (Badri Ahmadi et al., 2017).

The first step is to list out the level of priority by pairwise comparison. The number or linguistic term collected for comparison description in BWM is used for describing the pairwise comparison between every criterion and the best criterion, and the pairwise comparison between every criterion and the worst criterion. In order to consider the hesitations of decision makers, fuzzy numbers are adapted to better illustrate the linguistic preferences provided by decision makers. The Z-number, denoted as $\tilde{x}(l, m, u)$, is a fuzzy number raised by Zadeh (2011) that can show both constraints and reliability. In a Z-number linguistic variable for the ZBWM, there are two parts describing pairwise importance and reliabilities of this comparison, respectively. The linguistic variables of constraints include equally important (EI), weakly important (WI), fairly important (FI), very important (VI), and absolutely important (AI). As for linguistic variables of reliabilities, the linguistic terms can be denoted as very low (VL), low (L), medium (M), high (H), and very high (VH). Either linguistic variables of constraints or of reliabilities have their own membership functions. The membership functions for Z-number linguistic variables were calculated as shown in Table 9.3. The detailed calculation process can be referred to the work of Aboutorab et al. (2018).

Assume that p stakeholders participating in the decision-making process, the linguistic opinions need to be collected in the same process as mentioned above.

The optimal fuzzy weights of multiple stakeholders can be solved using Eq. (9.3).

$$
\begin{aligned}
&\text{Min} \sum_{k=1}^{p} \lambda_k \xi_k \\
&\text{s.t.} \begin{cases}
\left| \dfrac{(l_B^W, m_B^W, u_B^W)}{\left(l_j^W, m_j^W, u_j^W\right)} - \left(l_{kBj}, m_{kBj}, u_{kBj}\right) \right| \le \tilde{\xi}_k \\
\left| \dfrac{\left(l_j^W, m_j^W, u_j^W\right)}{(l_W^W, m_W^W, u_W^W)} - \left(l_{kjW}, m_{kjW}, u_{kjW}\right) \right| \le \tilde{\xi}_k \\
\displaystyle\sum_{j=1}^{n} \frac{\left(l_j^W + 4 \times m_j^W + u_j^W\right)}{6} = 1 \\
0 \le l_j^W \le m_j^W \le u_j^W
\end{cases}
\end{aligned} \tag{9.3}
$$

where, λ_k indicates the importance of the opinion of the kth stakeholder participating in decision making and $\sum_{k=1}^{p} \lambda_k = 1$; $\tilde{\xi}_k = (\xi_k, \xi_k, \xi_k)$ represents the upper limits of differences

TABLE 9.3 Transformation rules for Z-number linguistic variables to fuzzy numbers (Aboutorab et al., 2018).

Linguistic terms	Membership function
(EI,VL)	(1, 1, 1)
(EI,L)	(1, 1, 1)
(EI,M)	(1, 1, 1)
(EI,H)	(1, 1, 1)
(EI,VH)	(1, 1, 1)
(WI,VL)	(0.21, 0.32, 0.47)
(WI,L)	(0.37, 0.55, 0.82)
(WI,M)	(0.47, 0.71, 0.82)
(WI,H)	(0.56, 0.84, 1.26)
(WI,VH)	(0.63, 0.95, 1.43)
(FI,VL)	(0.47, 0.63, 0.79)
(FI,L)	(0.82, 1.10, 1.37)
(FI,M)	(1.07, 1.42, 1.78)
(FI,H)	(1.26, 1.68, 2.10)
(FI,VH)	(1.43, 1.90, 2.38)
(VI,VL)	(0.79, 0.95, 1.11)
(VI,L)	(1.37, 1.64, 1.92)
(VI,M)	(1.78, 2.13, 2.49)
(VI,H)	(2.10, 2.52, 2.94)
(VI,VH)	(2.38, 2.85, 3.33)
(AI,VL)	(1.11, 1.26, 1.42)
(AI,L)	(1.92, 2.19, 2.47)
(AI,M)	(2.49, 2.84, 3.20)
(AI,H)	(2.94, 3.36, 3.78)
(AI,VH)	(3.33, 3.80, 4.28)

between pairwise comparison data and the calculated weights based on the opinion of the kth stakeholder; and $i=1, 2, \ldots, m$, $j=1, 2, \ldots, n$, and $k=1, 2, \ldots, p$.

The fuzzy weights obtained from Eq.(9.3) can be defuzzied into crisp weight by Eq. (9.4).

$$w_j = \frac{(l_j + 4 * m_j + u_j)}{6} \tag{9.4}$$

where, $j=1, 2, \ldots, n$ represents the jth criterion.

The global weight of a criterion was determined by the crisp local weight multiplying the crisp perspective weight, as shown in Eq. (9.5).

$$w_{global} = w_{perspective} \times w_{local} \tag{9.5}$$

9.3.4 Step 4. Rank the alternatives by the goal programming method

The goal programming method is a classical scheduling method (Papathanasiou and Ploskas, 2018). The goal programming has been revised to be adapted in problem selection by limiting the assigned alternative as one. In this case study, the goal programming method was adapted to select the best option among three oil management systems.

To begin with, data of criteria with respect to each alternative should be normalized to eliminate the potential impacts brought by different metric units. The benefit-type criteria represent a set of criteria that have the characteristic that the alternative will become better or more superior with the increase of the data with respect to the criteria. On the contrary, the cost-type criterion represents a set of criteria that have the characteristic that the alternative will become better or more superior with the decrease of the data with respect to the criteria. The normalized data is determined by Eq.(9.6).

$$y_{ij} = \begin{cases} \dfrac{x_{ij}}{\left(\dfrac{\sum_{j=1}^{n} x_{ij}}{n}\right)} & \text{for benefit} - \text{type criteria} \\[2ex] \dfrac{\left(\dfrac{\sum_{j=1}^{n} x_{ij}}{n}\right)}{x_{ij}} & \text{for cost} - \text{type criteria} \end{cases} \tag{9.6}$$

where, x_{ij} represents original data of the ith criterion with respect to the jth alternative with $i = 1, 2, \ldots, m$, and $j = 1, 2, \ldots, n$.

The best alternative can be solved by the linear programming Eq. (9.7).

$$\begin{aligned} &\text{Min} \sum_i w_i \left(d_i^+ + d_i^-\right) \\ &\text{s.t.} \begin{cases} \sum_j a_j y_{ij} - d_i^+ + d_i^- = g_i \\ d_i^+ \geq 0 \\ d_i^- \geq 0 \\ a_j = 0 \text{ or } 1 \\ \sum_j a_j = 1 \end{cases} \end{aligned} \tag{9.7}$$

where, g_i is the goal of the ith criterion where the maximum normalized value of the ith criterion is selected; d_i^+ and d_i^- are the redundancy variables to measure the differences between planned alternatives and the goal; w_i is the weight for the ith criterion and y_{ij} is the normalized data of the ith criterion with respect to the jth alternative; and $i=1, 2, \ldots, m$, and $j=1, 2, \ldots, n$.

The only a_{j^*} that satisfies $a_{j^*}=1$ indicates the j^*th alternative is the best option based on its performance in all criteria and preferences from multiple decision makers.

9.4 Case study

In this section, a case study regarding used cooking oil management was studied to better illustrate how to use MCDM to deal with selection problems based on LCSA results. In this case study, the best alternative among three used cooking oil domestic management systems was analyzed according to the performances in 29 criteria from environmental, economic, and social perspectives. The three options analyzed in this chapter include collection through school (SCH), collection from door to door (DTD), and collection through urban collection centers (UCC). The SCH system indicates an oil reuse and recycling system based on schools. In this system, schools are the collection points for the used cooking oil. All participants deliver waste oil generated from their lives to the schools by their own containers. The oils are collected and transported once a month by authorized organizations by van to a special working center. This special working center works as the transport station to collect all used cooking oils in a 1000 t storage container and to transport them further to a biodiesel plant by tanker. During the process, the cleaning of empty containers is conducted by an industrial dishwaster and there are workers with a degree of disability. The DTD system is almost the same with SCH, except that the collection points change from schools to the houses of citizens. In the UCC system, users bring their used cooking oils directly to the urban collection center. The oils in the urban collection center will be transported to biodiesel plant once the 1000 t storage container in the urban collection center is full. In this case, the participants should clean their own containers. The proceeding flows of these three options are shown in Fig. 9.2.

From the view of criteria, the data in environmental, economic, and social aspects were adapted from LCA, LCC, and SLCA respectively. The environmental criteria involve abiotic depletion (AD), acidification (AC), eutrophication (EU), global warming (GW), ozone layer depletion (OPD), human toxicity (HT), fresh water aquatic ecotoxicity (WAE), marine aquatic ecotoxicity (MAE), terrestrial ecotoxicity (TE), photochemical oxidation (PO), and energy consumption (ED). The economic criteria include personnel, transport, collection container, storage container, CO_2 costs, total cost, and total cost without CO_2. As for social aspect, the criteria used to evaluate in this case study were total employees, total working hours, total employees with disabilities, total employees with higher education, total employees with basic education, equal opportunities for sexual reasons, equal opportunities for disabilities, children's environmental education, local employment, public commitments to sustainability issues, and contribution to economic development. The data of LCSA results were adapted from previous study of LCSA with regard to used cooking oil management (Vinyes et al., 2013) as shown in Table 9.4.

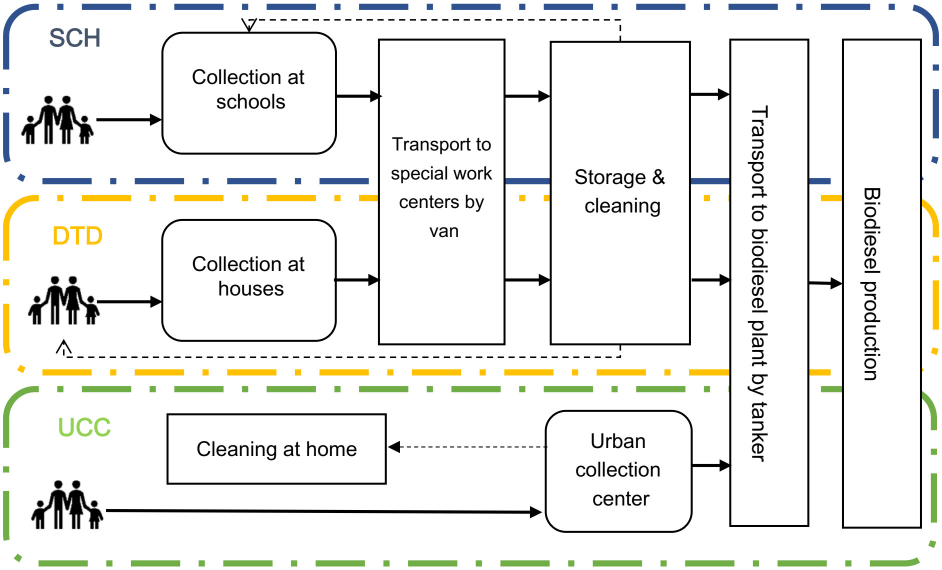

FIG. 9.2 The proceeding flows of three used cooking oils collection systems.

To analyze and select the best option among three alternatives, the revised group ZBWM combined with the goal programming method were used. In this case study, the hierarchical structure of criteria as shown in Fig. 9.3 requires separate comparison judgements. Therefore, four rounds of comparisons and judgements were conducted, which included the comparison among perspectives, the comparison among environmental criteria, the comparison among economic criteria, and the comparison among social criteria.

In this case study, two stakeholders are assumed to be participating in the decision making process. One is an environmentalist, and the other is the top manager of the recycling company. According to the standard of judgment, as shown in Table 9.4, the preferences of criteria and perspectives provided by these two decision makers are presented in Tables 9.5 and 9.6.

In this case study, two decision makers ($p=2$) participated in selection of three used oil treatment systems ($m=3$) based on 29 criteria ($n=29$), and the decision maker weights with respect to the environmentalist and the top manager were set as 0.3 and 0.7, respectively ($\lambda_1=0.3$, $\lambda_2=0.7$). According to Eq. (9.3) and Eq. (9.4), the criteria weights can be solved by using the data listed in the decision-making matrix in Tables 9.5 and 9.6. Taking the perspective comparison as an example, the fuzzy weights of perspectives can be solved by Eq. (9.8).

TABLE 9.4 The data of criteria with respect to three waste oil treatment systems.

From	Indicator	Units	DTD	SCH	UCC
LCA	Abiotic depletion	kg Sb eq	55.07	69.06	44.73
	Acidification	kg SO_2 eq	26.27	31.91	24.09
	Eutrophication	kg PO_4^- eq	7.82	9.57	7.34
	Global warming	kg CO_2 eq	6875.35	8510.98	5651.11
	Ozone layer depletion	kg CFC-11 eq	0.001	0.0013	0.008
	Human toxicity	kg 1,4-DB eq	2549.25	3111.33	2524.86
	Freshwater aquatic ecotoxicity	kg 1,4-DB eq	1144.25	1381.71	1119.02
	Marine aquatic ecotoxicity	kg 1,4-DB eq	2,294,870	2,798,558	2,283,039
	Terrestrial ecotoxicity	kg 1,4-DB eq	54.93	58.26	45.55
	Photochemical oxidation	kg C_2H_4	2.03	2.45	2
	Energy consumption	MJ eq	138,146.2	172,189.4	138,708.3
LCC	Personnel	€	36,104	4360	1587.5
	Transport	€	231.28	254.06	260.01
	Collection container	€	643.5	1271.11	667.33
	Storage container	€	756	840	882
	CO_2 costs	€	85.12	105.37	69.96
	Total cost	€	37,819.89	6830.54	3466.81
	Total cost without CO_2	€	37,734.78	6725.18	3396.85
SLCA	Total employees	–	55	20	9
	Total working hours	–	92,843	29,156	9126
	Total employees with disabilities	–	38	8	0
	Total employees with higher education	–	9	5	2
	Total employees with basic education	–	46	15	12
	Equal opportunities (sex)	–	100%	100%	100%
	Equal opportunities (disabilities)	–	33%	17%	0%
	Children's environmental education	–	17%	100%	25%
	Local employment	–	100%	100%	100%
	Public commitments to sustainability issues	–	100%	100%	100%
	Contribution to economic development	–	100%	100%	100%

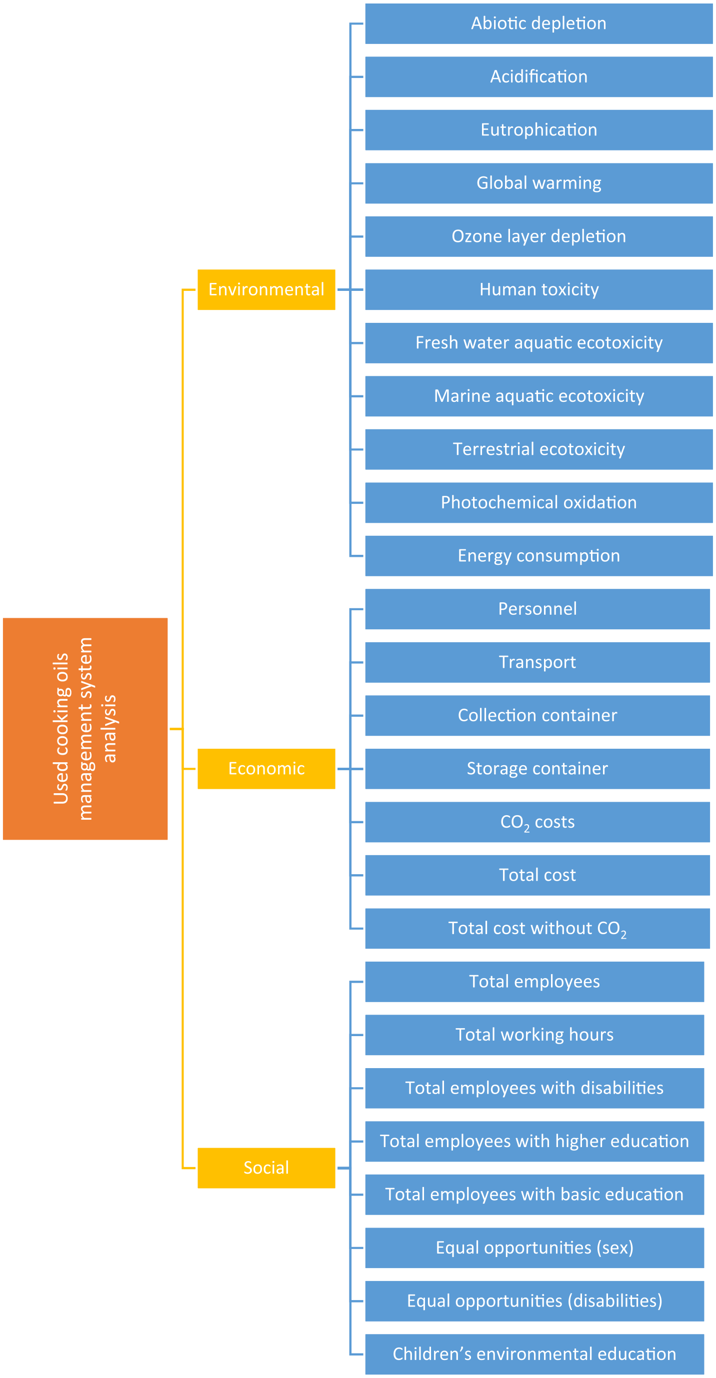

FIG. 9.3 The criteria system for waste oil treatment systems analysis.

TABLE 9.5 The decision-making matrix for best alternative comparison.

Range	DM	The best criteria			
Perspectives	**DM1**		**Environmental**	**Economic**	**Social**
		Environmental	(EI,VH)	(AI,M)	(VI,H)
			(1, 1, 1)	(2.49, 2.84, 3.2)	(2.1, 2.52, 2.94)
	DM2		Environmental	Economic	Social
		Economic	(AI,M)	(EI,VH)	(VI,H)
			(2.49, 2.84, 3.2)	(1, 1, 1)	(2.1, 2.52, 2.94)

Range	DM		AD	AC	EU	GW	ODP	HT	WAE	MAE	TE	PO	ED
Environmental	**DM1**		**AD**	**AC**	**EU**	**GW**	**ODP**	**HT**	**WAE**	**MAE**	**TE**	**PO**	**ED**
		GW	(WI,M)	(EI,H)	(EI,M)	(EI,VH)	(EI,VH)	(FI,H)	(EI,VH)	(EI,VH)	(EI,M)	(EI,M)	(FI,H)
			(0.56, 0.84, 1.26)	(1, 1, 1)	(1, 1, 1)	(1, 1, 1)	(1, 1, 1)	(1.26, 1.68, 2.1)	(1, 1, 1)	(1, 1, 1)	(1, 1, 1)	(1, 1, 1)	(1.26, 1.68, 2.1)
	DM2		AD	AC	EU	GW	ODP	HT	WAE	MAE	TE	PO	ED
		EU	(EI,M)	(VI,M)	(VI,M)	(WI,H)	(VI,M)	(WI,H)	(WI,M)	(VI,VH)	(VI,L)	(VI,M)	(EI,VH)
			(1, 1, 1)	(1.78, 2.13, 2.49)	(1.78, 2.13, 2.49)	(0.56, 0.84, 1.26)	(1.78, 2.13, 2.49)	(0.56, 0.84, 1.26)	(0.47, 0.71, 0.82)	(2.38, 2.82, 3.33)	(1.37, 1.64, 1.92)	(1.78, 2.13, 2.49)	(1, 1, 1)

Range	DM		Personnel	Transport	Collection container	Storage container	CO_2 costs	Total cost	Total cost without CO_2
Economic	**DM1**		**Personnel**	**Transport**	**Collection container**	**Storage container**	**CO_2 costs**	**Total cost**	**Total cost without CO_2**
		Total cost	(FI,H)	(FI,H)	(FI,H)	(FI,H)	(VI,H)	(EI,VH)	(WI, H)
			(1.26, 1.68, 2.1)	(1.26, 1.68, 2.1)	(1.26, 1.68, 2.1)	(1.26, 1.68, 2.1)	(2.1, 2.52, 2.94)	(1, 1, 1)	(0.56, 0.84, 1.26)
	DM2		Personnel	Transport	Collection container	Storage container	CO_2 costs	Total cost	Total cost without CO_2
		Total cost	(FI,H)	(FI,H)	(FI,H)	(FI,H)	(VI,H)	(EI,VH)	(WI, H)
			(1.26, 1.68, 2.1)	(1.26, 1.68, 2.1)	(1.26, 1.68, 2.1)	(1.26, 1.68, 2.1)	(2.1, 2.52, 2.94)	(1, 1, 1)	(0.56, 0.84, 1.26)

Social	DM1		Total employees	Total working hours	Total employees with disabilities	Total employees with higher education	Total employees with basic education	Equal opportunities (sex)	Equal opportunities (disabilities)	Children's environmental education	Local employment	Public commitments to sustainability issues	Contribution to economic development
		Children's environmental education	(FI,L)	(FI,M)	(WI,M)	(FI,M)	(FI,L)	(FI,M)	(WI,M)	(EI,VH)	(WI,M)	(WI,M)	(AI,H)
			(0.82, 1.1, 1.37)	(1.07, 1.42, 1.78)	(0.47, 0.71, 0.82)	(1.07, 1.42, 1.78)	(0.82, 1.1, 1.37)	(1.07, 1.42, 1.78)	(0.47, 0.71, 0.82)	(1, 1, 1)	(0.47, 0.71, 0.82)	(0.47, 0.71, 0.82)	(2.94, 3.36, 3.78)
	DM2		Total employees	Total working hours	Total employees with disabilities	Total employees with higher education	Total employees with basic education	Equal opportunities (sex)	Equal opportunities (disabilities)	Children's environmental education	Local employment	Public commitments to sustainability issues	Contribution to economic development
		Contribution to economic development	(FI,M)	(FI,M)	(VI,H)	(VI,H)	(VI,H)	(VI,H)	(VI,H)	(AI,M)	(WI,M)	(EI,H)	(EI,VH)
			(1.07, 1.42, 1.78)	(1.07, 1.42, 1.78)	(2.1, 2.52, 2.94)	(2.1, 2.52, 2.94)	(2.1, 2.52, 2.94)	(2.1, 2.52, 2.94)	(2.1, 2.52, 2.94)	(2.49, 2.84, 3.2)	(0.47, 0.71, 0.82)	(1, 1, 1)	(1, 1, 1)

Note: *AD, abiotic depletion; AC, acidification; EU, eutrophication; GW, global warming; OPD, ozone layer depletion; HT, human toxicity; WAE, fresh water aquatic ecotoxicity; MAE, marine aquatic ecotoxicity; TE, terrestrial ecotoxicity; PO, photochemical oxidation; and ED, energy consumption.*

TABLE 9.6 The decision-making matrix for worst alternative comparison.

	DM1	Environmentalist		DM2	Top manager	
Perspective		Economic			Environmental	
	Environmental	(AI,M)	(2.49, 2.84, 3.2)	Environmental	(EI,VH)	(1, 1, 1)
	Economic	(EI,VH)	(1, 1, 1)	Economic	(AI,M)	(2.49, 2.84, 3.2)
	Social	(WI,M)	(0.47, 0.71, 0.82)	Social	(WI,M)	(0.47, 0.71, 0.82)
		EU			MAE	
Environmental	AD	(WI,L)	(0.37, 0.55, 0.82)	AD	(VI,M)	(1.78, 2.13, 2.49)
	AC	(FI,H)	(1.26, 1.68, 2.1)	AC	(EI,M)	(1, 1, 1)
	EU	(FI,M)	(1.07, 1.42, 1.78)	EU	(EI,M)	(1, 1, 1)
	GW	(FI,H)	(1.26, 1.68, 2.1)	GW	(FI,H)	(1.26, 1.68, 2.1)
	ODP	(FI,M)	(1.07, 1.42, 1.78)	ODP	(EI,M)	(1, 1, 1)
	HT	(EI,L)	(1, 1, 1)	HT	(FI,VH)	(1.43, 1.9, 2.38)
	WAE	(FI,H)	(1.26, 1.68, 2.1)	WAE	(FI,M)	(1.07, 1.42, 1.78)
	MAE	(FI,H)	(1.26, 1.68, 2.1)	MAE	(EI,VH)	(1, 1, 1)
	TE	(FI,M)	(1.07, 1.42, 1.78)	TE	(EI,M)	(1, 1, 1)
	PO	(FI,M)	(1.07, 1.42, 1.78)	PO	(EI,M)	(1, 1, 1)
	ED	(EI,VH)	(1, 1, 1)	ED	(VI,H)	(2.1, 2.52, 2.94)
		CO_2 costs			CO_2 costs	
Economic	Personnel	(WI,L)	(0.37, 0.55, 0.82)	Personnel	(WI,L)	(0.37, 0.55, 0.82)
	Transport	(WI,H)	(0.56, 0.84, 1.26)	Transport	(WI,H)	(0.56, 0.84, 1.26)
	Collection container	(WI,H)	(0.56, 0.84, 1.26)	Collection container	(WI,H)	(0.56, 0.84, 1.26)

TABLE 9.6 The decision-making matrix for worst alternative comparison—cont'd

	DM1	Environmentalist		DM2	Top manager	
	Storage container	(WI,H)	(0.56, 0.84, 1.26)	Storage container	(WI,H)	(0.56, 0.84, 1.26)
	CO_2 costs	(EI,VH)	(1, 1, 1)	CO_2 costs	(EI,VH)	(1, 1, 1)
	Total cost	(VI,H)	(2.1, 2.52, 2.94)	Total cost	(VI,H)	(2.1, 2.52, 2.94)
	Total cost without CO_2	(FI,L)	(0.82, 1.1, 1.37)	Total cost without CO_2	(FI,L)	(0.82, 1.1, 1.37)
		Contribution to economic development			Children's environmental education	
Social	Total employees	(FI,H)	(1.26, 1.68, 2.1)	Total employees	(FI,M)	(1.07, 1.42, 1.78)
	Total working hours	(FI,M)	(1.07, 1.42, 1.78)	Total working hours	(FI,M)	(1.07, 1.42, 1.78)
	Total employees with disabilities	(VI,L)	(1.37, 1.64, 1.92)	Total employees with disabilities	(WI,L)	(0.37, 0.55, 0.82)
	Total employees with higher education	(FI,M)	(1.07, 1.42, 1.78)	Total employees with higher education	(WI,L)	(0.37, 0.55, 0.82)
	Total employees with basic education	(FI,H)	(1.26, 1.68, 2.1)	Total employees with basic education	(WI,L)	(0.37, 0.55, 0.82)
	Equal opportunities (sex)	(FI,M)	(1.07, 1.42, 1.78)	Equal opportunities (sex)	(WI,L)	(0.37, 0.55, 0.82)
	Equal opportunities (disabilities)	(VI,L)	(1.37, 1.64, 1.92)	Equal opportunities (disabilities)	(WI,L)	(0.37, 0.55, 0.82)
	Children's environmental education	(AI,H)	(2.94, 3.36, 3.78)	Children's environmental education	(EI,VH)	(1, 1, 1)
	Local employment	(VI,L)	(1.37, 1.64, 1.92)	Local employment	(VI,M)	(1.78, 2.13, 2.49)
	Public commitments to sustainability issues	(VI,M)	(1.78, 2.13, 2.49)	Public commitments to sustainability issues	(AI,H)	(2.94, 3.36, 3.78)
	Contribution to economic development	(EI,VH)	(1, 1, 1)	Contribution to economic development	(AI,VH)	(3.33, 3.8, 4.28)

Note: AD, *abiotic depletion;* AC, *acidification;* EU, *eutrophication;* GW, *global warming;* OPD, *ozone layer depletion;* HT, *human toxicity;* WAE, *freshwater aquatic ecotoxicity;* MAE, *marine aquatic ecotoxicity;* TE, *terrestrial ecotoxicity;* PO, *photochemical oxidation;* ED, *energy consumption.*

$$
\begin{aligned}
&\text{Min } (0.3\times\xi_1 + 0.7\times\xi_2)\\
&\text{s.t.}\begin{cases}
-\xi_1\times u_2 \le l_1 - 2.49\times u_2 \le \xi_1\times u_2\\
-\xi_1\times m_2 \le m_1 - 2.84\times m_2 \le \xi_1\times m_2\\
-\xi_1\times l_2 \le u_1 - 3.2\times l_2 \le \xi_1\times l_2\\
-\xi_1\times u_3 \le l_1 - 2.1\times u_3 \le \xi_1\times u_3\\
-\xi_1\times m_3 \le m_1 - 2.52\times m_3 \le \xi_1\times m_3\\
-\xi_1\times l_3 \le u_1 - 2.94\times l_3 \le \xi_1\times l_3\\
-\xi_1\times u_2 \le l_1 - 2.49\times u_2 \le \xi_1\times u_2\\
-\xi_1\times m_2 \le m_1 - 2.84\times m_2 \le \xi_1\times m_2\\
-\xi_1\times l_2 \le u_1 - 3.2\times l_2 \le \xi_1\times l_2\\
-\xi_1\times u_2 \le l_3 - 0.47\times u_2 \le \xi_1\times u_2\\
-\xi_1\times m_2 \le m_3 - 0.71\times m_2 \le \xi_1\times m_2\\
-\xi_1\times l_2 \le u_3 - 0.82\times l_2 \le \xi_1\times l_2\\
-\xi_2\times u_2 \le l_1 - 2.49\times u_2 \le \xi_2\times u_2\\
-\xi_2\times m_2 \le m_1 - 2.84\times m_2 \le \xi_2\times m_2\\
-\xi_2\times l_2 \le u_1 - 3.2\times l_2 \le \xi_2\times l_2\\
-\xi_2\times u_3 \le l_1 - 2.1\times u_3 \le \xi_2\times u_3\\
-\xi_2\times m_3 \le m_1 - 2.52\times m_3 \le \xi_2\times m_3\\
-\xi_2\times l_3 \le u_1 - 2.94\times l_3 \le \xi_2\times l_3\\
-\xi_2\times u_1 \le l_2 - 2.49\times u_1 \le \xi_2\times u_1\\
-\xi_2\times m_1 \le m_2 - 2.84\times m_1 \le \xi_2\times m_1\\
-\xi_2\times l_1 \le u_2 - 3.2\times l_1 \le \xi_2\times l_1\\
-\xi_2\times u_1 \le l_3 - 0.47\times u_1 \le \xi_2\times u_1\\
-\xi_2\times m_1 \le m_3 - 0.71\times m_1 \le \xi_2\times m_1\\
-\xi_2\times l_1 \le u_3 - 0.82\times l_1 \le \xi_2\times l_1\\
\dfrac{(l_1+4\times m_1+u_1)}{6} + \dfrac{(l_2+4\times m_2+u_2)}{6} + \dfrac{(l_3+4\times m_3+u_3)}{6} = 1\\
0 \le l_1 \le m_1 \le u_1\\
0 \le l_2 \le m_2 \le u_2\\
0 \le l_3 \le m_3 \le u_3
\end{cases}
\end{aligned}
\tag{9.8}
$$

The local fuzzy weights for environmental, economic, and social perspectives, respectively, can be calculated by Eq. (9.8) as $\widetilde{w}_{en} = (0.2304, 0.2304, 0.2304)$, $\widetilde{w}_{ec} = (0.4863, 0.5676, 0.6499)$, and $\widetilde{w}_{so} = (0.1958, 0.1958, 0.2320)$. According to Eq. (9.4), the crisp local weights for those three perspectives were calculated as $w_{en} = 0.2304$, $w_{ec} = 0.5678$, and $w_{so} = 0.2018$. Similarly, the local weights for environmental criteria, economic criteria, and social criteria can be determined by Eq. (9.3) and Eq. (9.4), as presented in Table 9.7. For example, the criterion weight with respect to abiotic depletion was calculated by Eq. (9.9).

$$w_{global} = 0.2304 \times 0.1396 = 0.0322 \tag{9.9}$$

The global weights were calculated and presented in Table 9.7.

The global weights were used as the criteria weights in the aggregation process.

In this case study, the benefit-type criteria include some economic criteria (personnel, transport, collection container, and storage container) and all social criteria. The remaining criteria are cost-type criteria. Accordingly, the normalized data are presented in Table 9.8.

Based on Eq. (9.7), the best used cooking oil management system was selected. The results, which were $a_1 = 0$, while $a_2 = 1$ and $a_3 = 0$, show that the second alternative (DTD system) is the most suitable under current situation and based on the preferences of multiple stakeholders.

To evaluate the method, a sensitivity analysis was conducted on this case study. The weight for a major criterion was set as 0.16, and the rest were set as 0.03. The aggregating method was conducted repeatedly with change of the major criterion. The results of sensitivity analysis are shown as Fig. 9.4.

Observed from the sensitivity result, in all situations DTD is the best option to this case, except when personnel is the most preferred criterion, the best option was changed to SCH system. In this case, DTD has absolute priority in this selection problem and the proposed method is feasible and effective for the selection problem.

TABLE 9.7 The criteria weighting results.

Perspective	Criteria	Fuzzy local weights	Crisp local weights	Global weights
Environmental	Abiotic depletion	(0.0947, 0.1461, 0.1585)	0.1396	0.0322
(0.2304, 0.2304, 0.2304)	Acidification	(0.0523, 0.0523, 0.0806)	0.057	0.0131
0.2304	Eutrophication	(0.0527, 0.0806, 0.0806)	0.0759	0.0175
	Global warming	(0.1199, 0.1215, 0.1337)	0.1233	0.0284
	Ozone layer depletion	(0.0634, 0.0634, 0.0634)	0.0634	0.0146
	Human toxicity	(0.082, 0.0998, 0.1527)	0.1057	0.0244
	Fresh water aquatic ecotoxicity	(0.1073, 0.1073, 0.1214)	0.1096	0.0253
	Marine aquatic ecotoxicity	(0.0523, 0.0547, 0.0666)	0.0563	0.013
	Terrestrial ecotoxicity	(0.061, 0.0642, 0.0642)	0.0637	0.0147

Continued

TABLE 9.7 The criteria weighting results—cont'd

Perspective	Criteria	Fuzzy local weights	Crisp local weights	Global weights
	Photochemical oxidation	(0.0713, 0.0713, 0.0713)	0.0713	0.0164
	Energy consumption	(0.1226, 0.1341, 0.1459)	0.1342	0.0309
Economic	Personnel	(0.0943, 0.1142, 0.1142)	0.1109	0.0629
(0.4863, 0.5676, 0.6499)	Transport	(0.0942, 0.1071, 0.1675)	0.115	0.0653
0.5678	Collection container	(0.1157, 0.1467, 0.1678)	0.145	0.0824
	Storage container	(0.1157, 0.1157, 0.1675)	0.1243	0.0706
	CO_2 costs	(0.0971, 0.1121, 0.1125)	0.1097	0.0623
	Total cost	(0.1835, 0.2301, 0.24)	0.224	0.1272
	Total cost without CO_2	(0.1449, 0.1759, 0.1785)	0.1711	0.0972
Social	Total employees	(0.09, 0.0925, 0.1105)	0.0951	0.0192
(0.1958, 0.1958, 0.232)	Total working hours	(0.0815, 0.0866, 0.1032)	0.0885	0.0179
0.2018	Total employees with disabilities	(0.095, 0.095, 0.095)	0.095	0.0192
	Total employees with higher education	(0.0704, 0.0704, 0.0851)	0.0729	0.0147
	Total employees with basic education	(0.0763, 0.0763, 0.117)	0.0831	0.0168
	Equal opportunities (sex)	(0.0707, 0.0707, 0.1045)	0.0763	0.0154
	Equal opportunities (disabilities)	(0.095, 0.095, 0.095)	0.095	0.0192
	Children's environmental education	(0.0997, 0.1187, 0.1476)	0.1203	0.0243
	Local employment	(0.095, 0.095, 0.1199)	0.0992	0.02
	Public commitments to sustainability issues	(0.1135, 0.1293, 0.1456)	0.1294	0.0261
	Contribution to economic development	(0.0452, 0.0452, 0.0452)	0.0452	0.0091

TABLE 9.8 The normalized data with respect to oil management systems.

Indicator	SCH	DTD	UCC
AD	0.81504	1.022093	1.258365
AC	0.859396	1.043903	1.13837
EU	0.861372	1.054135	1.12307
GW	0.823933	1.019945	1.240903
ODP	2.641026	3.433333	0.429167
HT	0.87695	1.070307	1.080646
WAE	0.87934	1.061825	1.085766
MAE	0.878603	1.071443	1.076995
TE	0.908227	0.963287	1.161654
PO	0.881633	1.064039	1.08
ED	0.869283	1.083499	1.079108
Personnel	3.214946	0.388244	8.829711
Transport	0.977919	1.074239	0.95554
Collection container	0.677083	1.337446	1.289687
Storage container	0.983333	1.092593	0.936508
CO_2 costs	0.823922	1.019933	1.240947
Total cost	2.348142	0.424091	4.626466
Total cost without CO_2	2.372021	0.422747	4.696195
Total employees	0.714286	1.964286	0.321429
Total working hours	0.667058	2.124149	0.208793
Total employees with disabilities	0.521739	2.478261	0
Total employees with higher education	0.9375	1.6875	0.375
Total employees with basic education	0.616438	1.890411	0.493151
Equal opportunities (sex)	1	1	1
Equal opportunities (disabilities)	1.02	1.98	0
Children's environmental education	2.112676	0.359155	0.528169
Local employment	1	1	1
Public commitments to sustainability issues	1	1	1
Contribution to economic development	1	1	1

Note: AD, *abiotic depletion;* AC, *acidification;* EU, *eutrophication;* GW, *global warming;* OPD, *ozone layer depletion;* HT, *human toxicity;* WAE, *fresh water aquatic ecotoxicity;* MAE, *marine aquatic ecotoxicity;* TE, *terrestrial ecotoxicity;* PO, *photochemical oxidation;* ED, *energy consumption.*

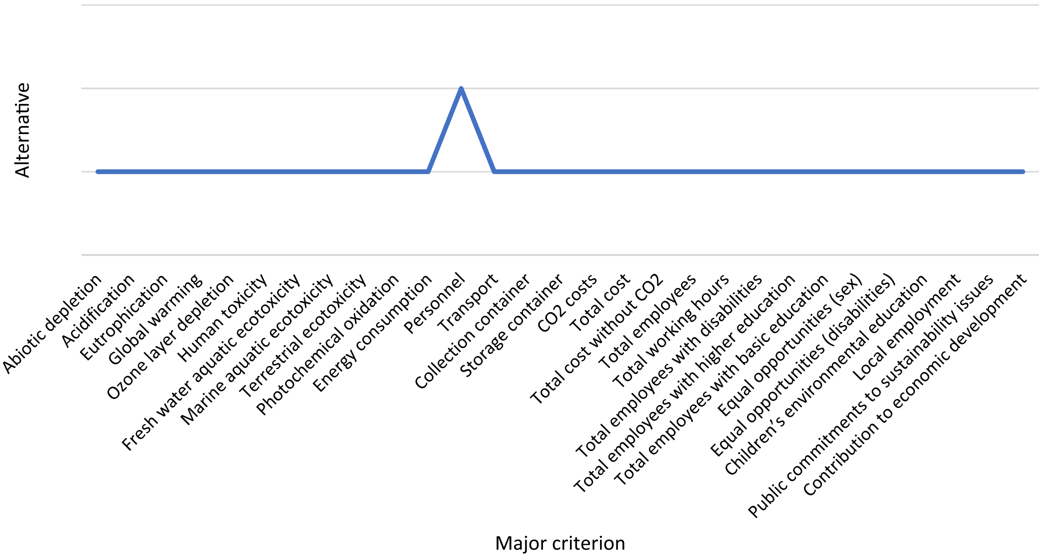

FIG. 9.4 Sensitivity result.

9.5 Conclusions

The combination of LCSA and MCDM improves the data quality and provides a direct decision making result based on all-round sustainability assessment. This chapter summarized MCDM methods including weighting methods and aggregating methods used in competitive case studies based on LCSA results. From the summarization tables, the popularity of weighting methods and aggregating methods used in LCSA analysis can be observed, respectively. To better illustrate the operation of the combination of LCSA and MCDM, a group ZBWM combined with the goal programming method was adapted to analyze a case study regarding waste oil management technologies selection. The method proved feasible and effective to assist group decision makers to achieve a consensus.

In the future, more MCDM will be developed or revised to make those methods suitable for more situations or find out the most efficient tool for a certain industry. More works are expected be done for an overall LCSA.

References

Aboutorab, H., Saberi, M., Asadabadi, M.R., et al., 2018. ZBWM: the Z-number extension of best worst method and its application for supplier development. Expert Syst. Appl. 107, 115–125. https://doi.org/10.1016/j.eswa.2018.04.015.

Akadiri, P.O., Olomolaiye, P.O., Chinyio, E.A., 2013. Multi-criteria evaluation model for the selection of sustainable materials for building projects. Autom. Constr. https://doi.org/10.1016/j.autcon.2012.10.004.

Akhtar, S., Reza, B., Hewage, K., et al., 2015. Life cycle sustainability assessment (LCSA) for selection of sewer pipe materials. Clean Techn. Environ. Policy 17. https://doi.org/10.1007/s10098-014-0849-x.

Al Garni, H.Z., Awasthi, A., 2017. Solar PV power plant site selection using a GIS-AHP based approach with application in Saudi Arabia. Appl. Energy. 206, 1225–1240. https://doi.org/10.1016/j.apenergy.2017.10.024.

Alhumaid, M., Ghumman, A.R., Haider, H., et al., 2018. Sustainability evaluation framework of urban stormwater drainage options for arid environments using hydraulic modeling and multicriteria decision-making. Water (Switzerland) 10. https://doi.org/10.3390/w10050581.

Alwaer, H., Clements-Croome, D.J., 2010. Key performance indicators (KPIs) and priority setting in using the multi-attribute approach for assessing sustainable intelligent buildings. Build. Environ. 45, 799–807. https://doi.org/10.1016/j.buildenv.2009.08.019.

Amin Hosseini, S.M., De La Fuente, A., Pons, O., 2016. Multi-criteria decision-making method for assessing the sustainability of post-disaster temporary housing units technologies: a case study in Bam, 2003. Sustain. Cities Soc. 20, 38–51. https://doi.org/10.1016/j.scs.2015.09.012.

Awasthi, A., Chauhan, S.S., 2011. Using AHP and Dempster-Shafer theory for evaluating sustainable transport solutions. Environ. Model. Software. 26, 787–796. https://doi.org/10.1016/j.envsoft.2010.11.010.

Awasthi, A., Chauhan, S.S., 2012. A hybrid approach integrating affinity diagram, AHP and fuzzy TOPSIS for sustainable city logistics planning. App. Math. Model. 36, 573–584. https://doi.org/10.1016/j.apm.2011.07.033.

Awasthi, A., Chauhan, S.S., Omrani, H., 2011. Application of fuzzy TOPSIS in evaluating sustainable transportation systems. Expert Syst. Appl. 38, 12270–12280. https://doi.org/10.1016/j.eswa.2011.04.005.

Azadnia, A.H., Saman, M.Z.M., Wong, K.Y., 2015. Sustainable supplier selection and order lot-sizing: an integrated multi-objective decision-making process. Int. J. Prod. Res. 53, 383–408.

Badri Ahmadi, H., Kusi-Sarpong, S., Rezaei, J., 2017. Assessing the social sustainability of supply chains using Best Worst Method. Resour. Conserv. Recycl. https://doi.org/10.1016/j.resconrec.2017.07.020.

Balasbaneh, A.T., Bin, M.A.K., Khaleghi, S.J., 2018. Sustainability choice of different hybrid timber structure for low medium cost single-story residential building: Environmental, economic and social assessment. J Build. Eng. https://doi.org/10.1016/j.jobe.2018.07.006.

Baležentis, T., Streimikiene, D., 2017. Multi-criteria ranking of energy generation scenarios with Monte Carlo simulation. Appl. Energy. 185, 862–871. https://doi.org/10.1016/j.apenergy.2016.10.085.

Barata, J.F.F., Quelhas, O.L.G., Costa, H.G., et al., 2014. Multi-criteria indicator for sustainability rating in suppliers of the oil and gas industries in Brazil. Sustainability. https://doi.org/10.3390/su6031107.

Barić, D., Pilko, H., Strujić, J., 2016. An analytic hierarchy process model to evaluate road section design. Transport. 31, 312–321. https://doi.org/10.3846/16484142.2016.1157830.

Berriel, S.S., Ruiz, Y., Sánchez, I.R., et al., 2018. Introducing low carbon cement in Cuba—a life cycle sustainability assessment study. In: RILEM Bookseries, pp. 415–421.

Bojković, N., Anić, I., Pejčić-Tarle, S., 2010. One solution for cross-country transport-sustainability evaluation using a modified ELECTRE method. Ecol. Econ. 69, 1176–1186. https://doi.org/10.1016/j.ecolecon.2010.01.006.

Bueno Cadena, P.C., Vassallo Magro, J.M., 2015. Setting the weights of sustainability criteria for the appraisal of transport projects. Transport. https://doi.org/10.3846/16484142.2015.1086890.

Büyüközkan, G., Karabulut, Y., 2017. Energy project performance evaluation with sustainability perspective. Energy. 119, 549–560. https://doi.org/10.1016/j.energy.2016.12.087.

Caruso, M.C., Menna, C., Asprone, D., et al., 2017. Methodology for life-cycle sustainability assessment of building structures. ACI Struct. J. https://doi.org/10.14359/51689426.

Castillo, H., Pitfield, D.E., 2010. ELASTIC - a methodological framework for identifying and selecting sustainable transport indicators. Transp. Res. Part D: Transp. Environ. 15, 179–188. https://doi.org/10.1016/j.trd.2009.09.002.

Chatterjee, K., Pamucar, D., Zavadskas, E.K., 2018. Evaluating the performance of suppliers based on using the R'AMATEL-MAIRCA method for green supply chain implementation in electronics industry. J. Clean. Prod. 184, 101–129. https://doi.org/10.1016/j.jclepro.2018.02.186.

Cohen, E., 2009. Applying best-worst scaling to wine marketing. Int. J. Wine Bus. Res. https://doi.org/10.1108/17511060910948008.

Dai, J., Blackhurst, J., 2012. A four-phase AHP-QFD approach for supplier assessment: a sustainability perspective. Int. J. Prod. Res. 50, 5474–5490. https://doi.org/10.1080/00207543.2011.639396.

Das, R., Shaw, K., 2017. Uncertain supply chain network design considering carbon footprint and social factors using two-stage approach. Clean Techn. Environ. Policy. 19, 2491–2519. https://doi.org/10.1007/s10098-017-1446-6.

de la Fuente, A., Blanco, A., Armengou, J., Aguado, A., 2017. Sustainability based-approach to determine the concrete type and reinforcement configuration of TBM tunnels linings. Case study: extension line to Barcelona airport T1. Tunn. Undergr. Space Technol. 61, 179–188. https://doi.org/10.1016/j.tust.2016.10.008.

De La Fuente, A., Pons, O., Josa, A., Aguado, A., 2016. Multi-criteria decision making in the sustainability assessment of sewerage pipe systems. J. Clean. Prod. 112, 4762–4770. https://doi.org/10.1016/j.jclepro.2015.07.002.

De Luca, A.I., Falcone, G., Stillitano, T., et al., 2018. Evaluation of sustainable innovations in olive growing systems: A life cycle sustainability assessment case study in southern Italy. J. Clean. Prod. 171, 1187–1202. https://doi.org/10.1016/j.jclepro.2017.10.119.

Debnath, A., Roy, J., Kar, S., et al., 2017. A hybrid MCDM approach for strategic project portfolio selection of agro by-products. Sustainability 9. https://doi.org/10.3390/su9081302.

Del Caño A, Gómez D, De La Cruz MP (2012) Uncertainty analysis in the sustainable design of concrete structures: a probabilistic method. Construct. Build Mater. https://doi.org/10.1016/j.conbuildmat.2012.04.020.

Diaz-Balteiro, L., González-Pachón, J., Romero, C., 2017. Measuring systems sustainability with multi-criteria methods: a critical review. Eur. J. Oper. Res. 258, 607–616. https://doi.org/10.1016/j.ejor.2016.08.075.

Díaz-Cuevas, P., Camarillo-Naranjo, J.M., Pérez-Alcántara, J.P., 2018. Relational spatial database and multi-criteria decision methods for selecting optimum locations for photovoltaic power plants in the province of Seville (southern Spain). Clean Techn. Environ. Policy. 20, 1889–1902. https://doi.org/10.1007/s10098-018-1587-2.

Dimić, S., Pamučar, D., Ljubojević, S., Dorović, B., 2016. Strategic transport management models—the case study of an oil industry. Sustainability 8. https://doi.org/10.3390/su8090954.

Ekener, E., Hansson, J., Larsson, A., Peck, P., 2018. Developing life cycle sustainability assessment methodology by applying values-based sustainability weighting—tested on biomass based and fossil transportation fuels. J. Clean. Prod. 181, 337–351. https://doi.org/10.1016/j.jclepro.2018.01.211.

Entezaminia, A., Heydari, M., Rahmani, D., 2016. A multi-objective model for multi-product multi-site aggregate production planning in a green supply chain: considering collection and recycling centers. J. Manuf. Syst. 40, 63–75. https://doi.org/10.1016/j.jmsy.2016.06.004.

Fallahpour, A., Udoncy Olugu, E., Nurmaya Musa, S., et al., 2017. A decision support model for sustainable supplier selection in sustainable supply chain management. Comput. Ind. Eng. 105, 391–410. https://doi.org/10.1016/j.cie.2017.01.005.

Formisano, A., Mazzolani, F.M., 2015. On the selection by MCDM methods of the optimal system for seismic retrofitting and vertical addition of existing buildings. Comput. Struct. 159, 1–13. https://doi.org/10.1016/j.compstruc.2015.06.016.

Gao, R., Nam, H.O., Il, K.W., Jang, H., 2017. National options for a sustainable nuclear energy system: MCDM evaluation using an improved integrated weighting approach. Energies 10. https://doi.org/10.3390/en10122017.

Govindan, K., Khodaverdi, R., Jafarian, A., 2013. A fuzzy multi criteria approach for measuring sustainability performance of a supplier based on triple bottom line approach. J. Clean. Prod. 47, 345–354. https://doi.org/10.1016/j.jclepro.2012.04.014.

Govindan, K., Madan Shankar, K., Kannan, D., 2016. Sustainable material selection for construction industry—a hybrid multi criteria decision making approach. Renew. Sustain. Energy Rev. 55, 1274–1288.

Guo, S., Zhao, H., 2015. Optimal site selection of electric vehicle charging station by using fuzzy TOPSIS based on sustainability perspective. Appl. Energy. https://doi.org/10.1016/j.apenergy.2015.08.082.

Halog, A., Manik, Y., 2011. Advancing integrated systems modelling framework for life cycle sustainability assessment. Sustainability. https://doi.org/10.3390/su3020469.

Hayashi, K., Hondo, H., Moriizumi, Y., 2016. Preference construction processes for renewable energies: assessing the influence of sustainability information and decision support methods. Sustainability 8. https://doi.org/10.3390/su8111114.

He, B., Luo, T., Huang, S., 2019. Product sustainability assessment for product life cycle. J. Clean. Prod. 206, 238–250. https://doi.org/10.1016/j.jclepro.2018.09.097.

Ho, W., Xu, X., Dey, P.K., 2010. Multi-criteria decision making approaches for supplier evaluation and selection: a literature review. Eur. J. Oper. Res. https://doi.org/10.1016/j.ejor.2009.05.009.

Hsu, C.W., Kuo, T.C., Shyu, G.S., Chen, P.S., 2014. Low carbon supplier selection in the hotel industry. Sustainability. 6, 2658–2684. https://doi.org/10.3390/su6052658.

Huang, C.Y., Chung, P.H., Shyu, J.Z., et al., 2018. Evaluation and selection of materials for particulate matter MEMS sensors by using hybrid MCDM methods. Sustainability. https://doi.org/10.3390/su10103451.

Inti, S., Tandon, V., 2017. Application of fuzzy preference–analytic hierarchy process logic in evaluating sustainability of transportation infrastructure requiring multicriteria decision making. J. Infrastruct. Syst. https://doi.org/10.1061/(asce)is.1943-555x.0000373.

Ioppolo, G., Traverso, M., Finkbeiner, M., 2019. Preface—a new paradigm for life cycle thinking: Exploring sustainability in urban development scenarios. Int. J. Life Cycle Assess. https://doi.org/10.1007/s11367-019-01586-x.

Ishizaka, A., Siraj, S., 2018. Are multi-criteria decision-making tools useful? An experimental comparative study of three methods. Eur. J. Oper. Res. https://doi.org/10.1016/j.ejor.2017.05.041.

Jeon, C.M., Amekudzi, A.A., Guensler, R.L., 2010. Evaluating plan alternatives for transportation system sustainability: Atlanta metropolitan region. Int. J. Sustain. Transp. 4, 227–247. https://doi.org/10.1080/15568310902940209.

Jia, J., Ibrahim, M., Hadi, M., et al., 2018. Multi-criteria evaluation framework in selection of accelerated bridge construction (ABC) method. Sustainability 10. https://doi.org/10.3390/su10114059.

Jiang, P., Hu, Y.C., Yen, G.F., et al., 2018. Using a novel Grey DANP model to identify interactions between manufacturing and logistics industries in China. Sustainability 10. https://doi.org/10.3390/su10103456.

Jones, S., Tefe, M., Appiah-Opoku, S., 2013. Proposed framework for sustainability screening of urban transport projects in developing countries: a case study of Accra, Ghana. Transp. Res. A Policy Pract. 49, 21–34. https://doi.org/10.1016/j.tra.2013.01.003.

Jovanovic, M., Afgan, N., Bakic, V., 2010. An analytical method for the measurement of energy system sustainability in urban areas. Energy. https://doi.org/10.1016/j.energy.2010.06.010.

Karunathilake, H., Hewage, K., Mérida, W., Sadiq, R., 2019. Renewable energy selection for net-zero energy communities: life cycle based decision making under uncertainty. Renew. Energy. 130, 558–573. https://doi.org/10.1016/j.renene.2018.06.086.

Khalili, N.R., Duecker, S., 2013. Application of multi-criteria decision analysis in design of sustainable environmental management system framework. J. Clean. Prod. 47, 188–198. https://doi.org/10.1016/j.jclepro.2012.10.044.

Khoshnava, S.M., Rostami, R., Valipour, A., et al., 2018. Rank of green building material criteria based on the three pillars of sustainability using the hybrid multi criteria decision making method. J. Clean. Prod. 173, 82–99. https://doi.org/10.1016/j.jclepro.2016.10.066.

Klein, S.J.W., Whalley, S., 2015. Comparing the sustainability of U.S. electricity options through multi-criteria decision analysis. Energy Policy. https://doi.org/10.1016/j.enpol.2015.01.007.

Kucukvar, M., Noori, M., Egilmez, G., Tatari, O., 2014. Stochastic decision modeling for sustainable pavement designs. Int. J. Life Cycle Assess. 19, 1185–1199. https://doi.org/10.1007/s11367-014-0723-4.

Kuo, T.C., Hsu, C.W., Li, J.Y., 2015. Developing a green supplier selection model by using the DANP with VIKOR. Sustainability. https://doi.org/10.3390/su7021661.

Kusi-Sarpong, S., Gupta, H., Sarkis, J., 2018. A supply chain sustainability innovation framework and evaluation methodology. Int. J. Prod. Res.

Liu, A., Xiao, Y., Ji, X., et al., 2018. A novel two-stage integrated model for supplier selection of green fresh product. Sustainability 10. https://doi.org/10.3390/su10072371.

López, E., Monzón, A., 2010. Integration of sustainability issues in strategic transportation planning: a multi-criteria model for the assessment of transport infrastructure plans. Comput. Aided Civ. Inf. Eng. 25, 440–451. https://doi.org/10.1111/j.1467-8667.2010.00652.x.

Luthra, S., Govindan, K., Kannan, D., et al., 2017a. An integrated framework for sustainable supplier selection and evaluation in supply chains. J. Clean. Prod. https://doi.org/10.1016/j.jclepro.2016.09.078.

Luthra, S., Mangla, S.K., Chan, F.T.S., Venkatesh, V.G., 2017b. Evaluating the drivers to information and communication technology for effective sustainability initiatives in supply chains. Int. J. Inf. Technol. Decis. Mak. https://doi.org/10.1142/s0219622017500419.

Mahbub, N., Oyedun, A.O., Zhang, H., et al., 2018. A life cycle sustainability assessment (LCSA) of oxymethylene ether as a diesel additive produced from forest biomass. Int. J. Life Cycle Assess. https://doi.org/10.1007/s11367-018-1529-6.

Manzardo, A., Ren, J., Mazzi, A., Scipioni, A., 2012. A grey-based group decision-making methodology for the selection of hydrogen technologies in life cycle sustainability perspective. Int. J. Hydrogen Energy. 37, 17663–17670. https://doi.org/10.1016/j.ijhydene.2012.08.137.

Marzouk, M., Elmesteckawi, L., 2015. Analyzing procurement route selection for electric power plants projects using Smart. J. Civ. Eng. Manag. 21, 912–922. https://doi.org/10.3846/13923730.2014.971131.

Mavi, R.K., Goh, M., Zarbakhshnia, N., 2017. Sustainable third-party reverse logistic provider selection with fuzzy SWARA and fuzzy MOORA in plastic industry. Int. J. Adv. Manuf. Technol. 91, 2401–2418. https://doi.org/10.1007/s00170-016-9880-x.

Mirjat, N.H., Uqaili, M.A., Harijan, K., et al., 2018. Multi-criteria analysis of electricity generation scenarios for sustainable energy planning in Pakistan. Energies 11. https://doi.org/10.3390/en11040757.

Mitropoulos, L.K., Prevedouros, P.D., 2016. Incorporating sustainability assessment in transportation planning: an urban transportation vehicle-based approach. Transp. Plan. Technol. 39, 439–463. https://doi.org/10.1080/03081060.2016.1174363.

Moslehi, S., Reddy, T.A., 2019. A new quantitative life cycle sustainability assessment framework: application to integrated energy systems. Appl. Energy. https://doi.org/10.1016/j.apenergy.2019.01.237.

Onat, N.C., Gumus, S., Kucukvar, M., Tatari, O., 2016. Application of the TOPSIS and intuitionistic fuzzy set approaches for ranking the life cycle sustainability performance of alternative vehicle technologies. Sustain. Produc. Consum. https://doi.org/10.1016/j.spc.2015.12.003.

Opher, T., Friedler, E., Shapira, A., 2018a. Comparative life cycle sustainability assessment of urban water reuse at various centralization scales. Int. J. Life Cycle Assess., 1–14. https://doi.org/10.1007/s11367-018-1469-1.

Opher, T., Shapira, A., Friedler, E., 2018b. A comparative social life cycle assessment of urban domestic water reuse alternatives. Int. J. Life Cycle Assess. 23, 1315–1330. https://doi.org/10.1007/s11367-017-1356-1.

Osorio-Tejada, J.L., Llera-Sastresa, E., Scarpellini, S., 2017. A multi-criteria sustainability assessment for biodiesel and liquefied natural gas as alternative fuels in transport systems. J. Nat. Gas Sci. Eng. 42, 169–186. https://doi.org/10.1016/j.jngse.2017.02.046.

Ozcan-Deniz, G., Zhu, Y., 2015. A multi-objective decision-support model for selecting environmentally conscious highway construction methods. J. Civ. Eng. Manag. 21, 733–747. https://doi.org/10.3846/13923730.2014.893915.

Padhi, S.S., Pati, R.K., Rajeev, A., 2018. Framework for selecting sustainable supply chain processes and industries using an integrated approach. J. Clean. Prod. 184, 969–984. https://doi.org/10.1016/j.jclepro.2018.02.306.

Palevičius, V., Paliulis, G.M., Venckauskaitė, J., Vengrys, B., 2013. Evaluation of the requirement for passenger car parking spaces using multi-criteria methods. J. Civ. Eng. Manag. 19, 49–58. https://doi.org/10.3846/13923730.2012.727463.

Pamucar, D., Gigovic, L., Bajic, Z., Janoševic, M., 2017. Location selection for wind farms using GIS multi-criteria hybrid model: an approach based on fuzzy and rough numbers. Sustainability 9. https://doi.org/10.3390/su9081315.

Pamucar, D.S., Tarle, S.P., Parezanovic, T., 2018. New hybrid multi-criteria decision-making DEMATEL-MAIRCA model: sustainable selection of a location for the development of multimodal logistics centre. Econ. Res. Istraz. https://doi.org/10.1080/1331677X.2018.1506706.

Papathanasiou, J., Ploskas, N., 2018. Goal programming. In: Springer Optimization and Its Applications.

Pons, O., De La Fuente, A., 2013. Integrated sustainability assessment method applied to structural concrete columns. Construct. Build Mater. 49, 882–893. https://doi.org/10.1016/j.conbuildmat.2013.09.009.

Promentilla, M.A.B., Janairo, J.I.B., Yu, D.E.C., et al., 2018. A stochastic fuzzy multi-criteria decision-making model for optimal selection of clean technologies. J. Clean. Prod. 183, 1289–1299. https://doi.org/10.1016/j.jclepro.2018.02.183.

Pujadas, P., Pardo-Bosch, F., Aguado-Renter, A., Aguado, A., 2017. MIVES multi-criteria approach for the evaluation, prioritization, and selection of public investment projects. A case study in the city of Barcelona. Land Use Policy. 64, 29–37. https://doi.org/10.1016/j.landusepol.2017.02.014.

Rao, C., Goh, M., Zhao, Y., Zheng, J., 2015. Location selection of city logistics centers under sustainability. Transp. Res. Part D: Transp. Environ. 36, 29–44. https://doi.org/10.1016/j.trd.2015.02.008.

Rashid, K., Razzaq, A., Ahmad, M., et al., 2017. Experimental and analytical selection of sustainable recycled concrete with ceramic waste aggregate. Construct. Build Mater. 154, 829–840. https://doi.org/10.1016/j.conbuildmat.2017.07.219.

Rashidi, M., Ghodrat, M., Samali, B., et al., 2017. Remedial modelling of steel bridges through application of analytical hierarchy process (AHP). Appl. Sci. 7, 168. https://doi.org/10.3390/app7020168.

Raslanas, S., Stasiukynas, A., Jurgelaityte, E., 2013. Sustainability assessment studies of recreational buildings. Procedia Eng., 929–937.

Ren, J., 2018. Life cycle aggregated sustainability index for the prioritization of industrial systems under data uncertainties. Comput. Chem. Eng. 113, 253–263. https://doi.org/10.1016/j.compchemeng.2018.03.015.

Ren, J., Liang, H., Chan, F.T.S., 2017a. Urban sewage sludge, sustainability, and transition for Eco-City: multi-criteria sustainability assessment of technologies based on best-worst method. Technol. Forecast. Soc. Chang. 116, 29–39. https://doi.org/10.1016/j.techfore.2016.10.070.

Ren, J., Manzardo, A., Mazzi, A., et al., 2015. Prioritization of bioethanol production pathways in China based on life cycle sustainability assessment and multicriteria decision-making. Int. J. Life Cycle Assess. https://doi.org/10.1007/s11367-015-0877-8.

Ren, J., Ren, X., Dong, L., et al., 2018. Multiactor multicriteria decision making for life cycle sustainability assessment under uncertainties. AICHE J. 64, 2103–2112. https://doi.org/10.1002/aic.16149.

Ren, J., Ren, X., Liang, H., et al., 2017. Multi-actor multi-criteria sustainability assessment framework for energy and industrial systems in life cycle perspective under uncertainties. Part 2: Improved extension theory. Int. J. Life Cycle Assess. 22, 1406–1417. https://doi.org/10.1007/s11367-016-1252-0.

Ren, J., Toniolo, S., 2018. Life cycle sustainability decision-support framework for ranking of hydrogen production pathways under uncertainties: an interval multi-criteria decision making approach. J. Clean. Prod. 175, 222–236. https://doi.org/10.1016/j.jclepro.2017.12.070.

Ren, J., Xu, D., Cao, H., et al., 2016. Sustainability decision support framework for industrial system prioritization. AICHE J. 62, 108–130. https://doi.org/10.1002/aic.15039.

Rezaei, J., 2015. Best-worst multi-criteria decision-making method. Omega (United Kingdom). 53, 49–57. https://doi.org/10.1016/j.omega.2014.11.009.

Rezaei, J., Nispeling, T., Sarkis, J., Tavasszy, L., 2016. A supplier selection life cycle approach integrating traditional and environmental criteria using the best worst method. J. Clean. Prod. https://doi.org/10.1016/j.jclepro.2016.06.125.

Rezaei, N., Diaz-Elsayed, N., Mohebbi, S., et al., 2019. A multi-criteria sustainability assessment of water reuse applications: a case study in Lakeland, Florida. Environ. Sci.: Water Res. Technol. 5, 102–118.

Roinioti, A., Koroneos, C., 2019. Integrated life cycle sustainability assessment of the Greek interconnected electricity system. Sustainable Energy Technol. Assess. 32, 29–46. https://doi.org/10.1016/j.seta.2019.01.003.

Saaty, R.W., 1987. The analytic hierarchy process-what it is and how it is used. Math. Model. https://doi.org/10.1016/0270-0255(87)90473-8.

Safaei Mohamadabadi, H., Tichkowsky, G., Kumar, A., 2009. Development of a multi-criteria assessment model for ranking of renewable and non-renewable transportation fuel vehicles. Energy. 34, 112–125. https://doi.org/10.1016/j.energy.2008.09.004.

San-José Lombera, J.T., Cuadrado Rojo, J., 2010. Industrial building design stage based on a system approach to their environmental sustainability. Construct. Build Mater. 24, 438–447. https://doi.org/10.1016/j.conbuildmat.2009.10.019.

Sánchez-Lozano, J.M., García-Cascales, M.S., Lamata, M.T., 2016. GIS-based onshore wind farm site selection using fuzzy multi-criteria decision making methods. Evaluating the case of Southeastern Spain. Appl. Energy. 171, 86–102. https://doi.org/10.1016/j.apenergy.2016.03.030.

Shahriar, A., Zargar, A., Hewage, K., et al., 2014. Life cycle sustainability assessment (LCSA) for selection of sewer pipe materials. Clean Techn. Environ. Policy. 17, 973–992. https://doi.org/10.1007/s10098-014-0849-x.

Simongáti, G., 2010. Multi-criteria decision making support tool for freight integrators: selecting the most sustainable alternative. Transport. 25, 89–97. https://doi.org/10.3846/transport.2010.12.

Šiožinytė, E., Antuchevičienė, J., Kutut, V., 2014. Upgrading the old vernacular building to contemporary norms: multiple criteria approach. J. Civ. Eng. Manag. 20, 291–298. https://doi.org/10.3846/13923730.2014.904814.

Sivaraja, C.M., Sakthivel, G., 2017. Compression ignition engine performance modelling using hybrid MCDM techniques for the selection of optimum fish oil biodiesel blend at different injection timings. Energy. 139, 118–141. https://doi.org/10.1016/j.energy.2017.07.134.

Škobalj, P., Kijevčanin, M., Afgan, N., et al., 2017. Multi-criteria sustainability analysis of thermal power plant Kolubara-A unit 2. Energy. 125, 837–847. https://doi.org/10.1016/j.energy.2017.02.027.

Stević, Ž., Pamučar, D., Subotić, M., et al., 2018. The location selection for roundabout construction using rough BWM-rough WASPAS approach based on a new rough Hamy aggregator. Sustainability 10. https://doi.org/10.3390/su10082817.

Su, C.M., Horng, D.J., Tseng, M.L., et al., 2016. Improving sustainable supply chain management using a novel hierarchical grey-DEMATEL approach. J. Clean. Prod. 134, 469–481. https://doi.org/10.1016/j.jclepro.2015.05.080.

Tahmasebi Birgani, Y., Yazdandoost, F., 2018. An integrated framework to evaluate resilient-sustainable urban drainage management plans using a combined-adaptive MCDM technique. Water Resour. Manag. 32, 2817–2835. https://doi.org/10.1007/s11269-018-1960-2.

Tang, Y., Ni, M., Zhou, Z., et al., 2018. Model development of sustainability assessment from a life cycle perspective: a case study on waste management systems in China. J. Clean. Prod. 210, 1005–1014. https://doi.org/10.1016/j.jclepro.2018.11.074.

Terracciano, G., Di Lorenzo, G., Formisano, A., Landolfo, R., 2015. Cold-formed thin-walled steel structures as vertical addition and energetic retrofitting systems of existing masonry buildings. Eur. J. Environ. Civ. Eng. 19, 850–866. https://doi.org/10.1080/19648189.2014.974832.

Tsai, W.H., Lin, S.J., Lee, Y.F., et al., 2013. Construction method selection for green building projects to improve environmental sustainability by using an MCDM approach. J. Environ. Plan. Manag. 56, 1487–1510. https://doi.org/10.1080/09640568.2012.731385.

Tsoutsos, T., Drandaki, M., Frantzeskaki, N., et al., 2009. Sustainable energy planning by using multi-criteria analysis application in the island of Crete. Energy Policy. 37, 1587–1600. https://doi.org/10.1016/j.enpol.2008.12.011.

Turskis, Z., Zavadskas, E.K., 2011. A new fuzzy additive ratio assessment method (Aras-F). Case study: the analysis of fuzzy multiple criteria in order to select the logistic centers location. Transport. 25, 423–432. https://doi.org/10.3846/transport.2010.52.

Validi, S., Bhattacharya, A., Byrne, P.J., 2014. A case analysis of a sustainable food supply chain distribution system—a multi-objective approach. Int. J. Prod. Econ. 152, 71–87. https://doi.org/10.1016/j.ijpe.2014.02.003.

van de Kaa, G., Kamp, L., Rezaei, J., 2017. Selection of biomass thermochemical conversion technology in the Netherlands: a best worst method approach. J. Clean. Prod. https://doi.org/10.1016/j.jclepro.2017.07.052.

Vinyes, E., Oliver-Solà, J., Ugaya, C., et al., 2013. Application of LCSA to used cooking oil waste management. Int. J. Life Cycle Assess. 18, 445–455. https://doi.org/10.1007/s11367-012-0482-z.

Vučijak, B., Kupusović, T., MidŽić-Kurtagić, S., Ćerić, A., 2013. Applicability of multicriteria decision aid to sustainable hydropower. Appl. Energy. 101, 261–267. https://doi.org/10.1016/j.apenergy.2012.05.024.

Wang, H., Jiang, Z., Zhang, H., et al., 2019. An integrated MCDM approach considering demands-matching for reverse logistics. J. Clean. Prod. 208, 199–210. https://doi.org/10.1016/j.jclepro.2018.10.131.

Wulf, C., Werker, J., Zapp, P., et al., 2018. Sustainable development goals as a guideline for indicator selection in life cycle sustainability assessment. Procedia CIRP, 59–65.

Xu, D., Lv, L., Ren, J., et al., 2017. Life cycle sustainability assessment of chemical processes: A vector-based three-dimensional algorithm coupled with AHP. Ind. Eng. Chem. Res. https://doi.org/10.1021/acs.iecr.7b02041.

Yazdani, M., Chatterjee, P., Zavadskas, E.K., Streimikiene, D., 2018. A novel integrated decision-making approach for the evaluation and selection of renewable energy technologies. Clean Techn. Environ. Policy. 20, 403–420. https://doi.org/10.1007/s10098-018-1488-4.

Yoon, K.P., Hwang, C.-L., 1995. Multiple Attribute Decision Making: An Introduction. Sage, Thousand Oaks, CA.

Yu, S.M., Wang, J., Qiang, W.J., Li, L., 2018. A multi-criteria decision-making model for hotel selection with linguistic distribution assessments. Appl. Soft Comput. https://doi.org/10.1016/j.asoc.2017.08.009.

Zadeh, L.A., 2011. A note on Z-numbers. Inf. Sci. 181, 2923–2932. https://doi.org/10.1016/j.ins.2011.02.022.

Zavadskas, E.K., Baušys, R., Lazauskas, M., 2015. Sustainable assessment of alternative sites for the construction of a waste incineration plant by applying WASPAS method with single-valued neutrosophic set. Sustainability. 7, 15923–15936. https://doi.org/10.3390/su71215792.

Zavadskas, E.K., Vainiūnas, P., Turskis, Z., Tamošaitienė, J., 2012. Multiple criteria decision support system for assessment of projects managers in construction. Int. J. Inf. Technol. Decis. Mak. 11, 501–520. https://doi.org/10.1142/s0219622012400135.

Zečević, S., Tadić, S., Krstić, M., 2017. Intermodal transport terminal location selection using a novel hybrid MCDM model. Int. J. Uncertainty Fuzziness Knowledge Based Syst. 25, 853–876. https://doi.org/10.1142/s0218488517500362.

Zhang, L., Bai, W., Yu, J., et al., 2018. Critical mineral security in China: an evaluation based on hybrid MCDM methods. Sustainability 10. https://doi.org/10.3390/su10114114.

Zhang, L., Zhou, P., Newton, S., et al., 2015. Evaluating clean energy alternatives for Jiangsu, China: an improved multi-criteria decision making method. Energy. 90, 953–964. https://doi.org/10.1016/j.energy.2015.07.124.

Zhao, H., Guo, S., 2014. Selecting green supplier of thermal power equipment by using a hybrid MCDM method for sustainability. Sustainability. 6, 217–235. https://doi.org/10.3390/su6010217.

Zhao, H., Li, N., 2016. Optimal siting of charging stations for electric vehicles based on fuzzy Delphi and hybrid multi-criteria decision making approaches from an extended sustainability perspective. Energies 9. https://doi.org/10.3390/en9040270.

Zheng, X., Easa, S.M., Yang, Z., et al., 2019. Life-cycle sustainability assessment of pavement maintenance alternatives: methodology and case study. J. Clean. Prod. 213, 659–672. https://doi.org/10.1016/j.jclepro.2018.12.227.

CHAPTER

10

Advancing life cycle sustainability assessment using multiple criteria decision making

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10.1 Introduction

Life cycle sustainability assessment (LCSA) is now a globally adopted framework for assessing the performance of products/services (Valdivia et al., 2013; Zamagni et al., 2013). The framework takes into account all three dimensions of assessment, viz., environmental, economic, and social (Finkbeiner et al., 2010). As proposed by Guinée (2016), apart from broadening the scope of impact, LCSA should also consider expanding the analysis by incorporating issues related to economy scale, and include technological relations in the assessment. LCSA is also expected to consider behavioral aspects, such as rebound effect and cultural aspects (Pizzirani et al., 2014). For example, conventionally economic criteria are used to select the products or alternatives, with increasing environmental pressures in recent times, it has become essential to include environmental impacts and societal concerns in the decision making. Hence, the LCSA framework should aim to choose alternatives that are nearer to low cost and farther from adverse environmental and social impacts. Various methods are typically used for assessing each of such aspects in LCSA, and use of systems approach with life cycle thinking are most essential properties of LCSA (Halog and Manik, 2011). Life cycle analysis (LCA) is most commonly used for assessing the environmental dimension (Azapagic, 2010), whereas life cycle costing (LCC) and social LCA (SLCA) are widely used indicators for evaluating economic and social dimensions, respectively (Naves et al., 2018; Neugebauer et al., 2015).

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The concept of LCA evolved in the last four decades with the evolution in understanding and importance of assessing alternatives' environmental impacts (Ness et al., 2007). LCA considers inputs and outputs throughout the life of an alternative and their combined environmental impacts. Life cycle thinking is also embedded in life cycle costing (LCC) and social impacts through social life cycle assessment (SLCA). Although the LCSA is an ambitious approach to achieve holistic assessments, there are constraints in applying the framework in reality. Fauzi et al. (2019) discuss numerous challenges in enabling LCSA, such as parity in the assessment methods (e.g., different temporal scopes and scales applied between the methods). One of the main challenges identified is integration of indicators across the methods. To be able to achieve the LCSA, it is necessary to integrate the assessment results obtained by methods such as LCA, LCC, and LCSA. One of the most common frameworks used to integrate the results obtained from different tools/methods is multiple criteria decision making (MCDM) (Costa et al., 2019; Hannouf and Assefa, 2018).

The current chapter is focused on discussing the MCDM applications in LCSA. The MCDM methods overview is given in Section 10.2, applications of MCDM methods in combination with LCSA are described in Section 10.3. Section 10.4 provides details on challenges in the application of MCDM while carrying out LCSA. Finally, a framework for MCDM based LCSA is proposed in Section 10.5, and conclusions are provided in Section 10.7.

10.2 MCDM methods overview

Decision-making involves consideration of multiple criteria, which are usually conflicting (for example, efficiency versus cost) with each other. LCSA also consists of evaluating the alternatives based on conflicting criteria. LCSA also strives to include priorities of all stakeholders into decision-making. Stakeholders have wide-ranging preferences, which adds to the complexity of the decision-making process. MCDM methods are developed to counter such complexities embedded in decision-making and provides a strategically suitable decision (Zopounidis and Pardalos, 2010; Hwang and Yoon, 1981). MADM can be diversified into two groups, which are (a) multiattribute decision making (MADM) and (b) multiobjective decision making (MODM) (Figueira et al., 2005; Hwang and Yoon, 1981). MADM is majorly associated with decision-making problems involving a finite number of alternatives (known as discrete variable problems), whereas MODM is concerned with decision-making problems with an infinite number of alternatives (known as continuous variable problems). In MODM, primary objective is to design/formulate an alternative that shows maximum promise or performance corresponding to limited resources.

Literature suggests that many types of MADM methodologies to integrate information processing of attributes with decision-making of humans involving logic and rational thinking have been developed. Asgharizadeh et al. (2017) classified MADM methods into input-oriented and output-oriented. Input-oriented is largely subclassified in two categories: "data available to DMs" and "type of data available." There could be problems where DMs' may have no information on alternatives available to them, whereas, even if data is available then the data could be purely qualitative, purely quantitative, or a mixture of qualitative and quantitative data. Contrary to input-oriented, the process-oriented area could be divided into compensatory and noncompensatory approaches (Hwang and Yoon, 1981). Compensatory approaches allow trade-off within available attributes. In these methods, unfavorable scores or disadvantage of an attribute can be counterbalanced by a favorable score or advantage of another attribute. The compensatory approach can be further segmented into three subcategories, such as:

a. Scoring approaches: In this type of approach, all available attributes are considered at once, and an alternative with maximum utility or score is selected. An example of this approach is simple additive weighting (SAW).
b. Compromising approaches: The approach involves selection of alternative which has the minimum distance from the ideal solution and maximum distance from nonideal solution; for example, technique for order of preference by similarity to ideal solution (TOPSIS).
c. Concordance approaches: Selection of alternative in this type of approaches is based on arranging a set of ranking preferences, which satisfies an adopted concordance measure. An example is *elimination et choixtraduisant la realité* or elimination and choice expressing reality (ELECTRE).

Contrary to compensatory approaches, noncompensatory approaches do not allow trade-off within the attributes and comparison of alternatives is based on considering each attribute individually. A few methods that are based on noncompensatory approaches are lexicographic, elimination by aspects, maximax, maximin, disjunctive constraint, conjunctive constraint, and dominance. The compensatory approaches are cognitively more challenging for decision-makers than noncompensatory approaches; however, the results could be more optimal (Yoon and Hwang, 1995). Segregating MADM methods into two major approach groups suggested by Hwang and Yoon (1981) is one of the ways of classification, whereas, there are multiple ways in which different studies have attempted to classify MADM methods. Readers can refer to Chen and Hwang (1992), where taxonomy of the MADM methods is provided. Triantaphyllou (2000) suggested that MADM methods can also be classified corresponding to decision-makers, such as methods involving single decision-maker and group decision-makers. A detailed understanding of group decision for single decision can be found in Kalbar et al. (2013). Similarly, Kahraman et al. (2015) classified MADM methods into outranking, distance-based, and pairwise-comparison based. Outranking approaches provide outrank relationships but not any value function, whereas, distance-based methods are a development of distance matrixes and pairwise-comparison methods compare a pair of alternatives or indicators at a time in sequence. Liou and Tzeng (2012) presented an overview of MADM method development from 1738 to 2012. The study divided MADM methods into three major categories, which are based on approaches of evaluation, weighting, and normalization.

One of the methods, named TOPSIS, has shown better performance in many applications. TOPSIS has been shown to take into account weights more effectively (Rafiaani et al., 2019; Kalbar et al., 2017a). Another advantage of TOPSIS is that the method takes into account the nature of the indicators (i.e., whether the indicators are "benefit" type or "cost" type) while processing the indicators score by creating sets of a positive ideal solution (PIS) and a negative ideal solution (NIS). Such an approach resembles human thinking and makes it unique among other available methods (Yadav et al., 2019; Kalbar et al., 2012). Considering the benefits and importance of choosing a distance-based method like TOPSIS, Yadav et al. (2019) developed a free and open-source software (FOSS) named PyTOPS, which efficiently supports the use of TOPSIS.

10.3 Generic structure of MADM methods

Understanding of decision-making processes requires clarity over a few terminologies that are commonly used, as explained below (Hwang and Yoon, 1981):

- **Objectives:** Purpose of solving a problem.
- **Characteristics:** This is either distinct or common to other elements and helps understanding an element's character.
- **Attribute or indicator:** This is a distinct element that helps in measuring a characteristic.
- **Criterion:** A criterion is a combination of indicators and helps in understanding the level up to which the set of indicators can achieve an objective.
- **Trade-off:** An exchange of one or more attributes within a criterion to achieve a benefit or advantage.

MADM methods rank or score a finite number of alternatives $A_i = (A_1, A_2, \ldots, A_m)$, based on a set of attributes/criteria/indicators, $X_j = (X_1, X_2, \ldots, X_n)$. The information available from the Decision Makers (DMs) can be represented in the form of a matrix called a decision matrix, which is shown below:

$$\text{Alternatives/Options}\quad \begin{array}{c} \\ \\ A_1 \\ A_2 \\ \vdots \\ A_i \\ \vdots \\ A_m \end{array} \begin{array}{cccccccc} \multicolumn{8}{c}{\text{Criteria/Attributes}} \\ X_1 & X_2 & X_3 & \cdots & X_j & \cdots & X_n \\ x_1(a_1) & x_2(a_1) & x_3(a_1) & \cdots & x_j(a_1) & \cdots & x_n(a_1) \\ x_1(a_2) & x_2(a_2) & x_3(a_2) & \cdots & x_j(a_2) & \cdots & x_n(a_2) \\ \vdots & \vdots & \vdots & \vdots & \vdots & \vdots & \vdots \\ x_1(a_i) & x_2(a_i) & x_3(a_i) & \cdots & x_j(a_i) & \cdots & x_n(a_m) \\ \vdots & \vdots & \vdots & \vdots & \vdots & \vdots & \vdots \\ x_1(a_m) & x_2(a_m) & x_3(a_m) & \cdots & x_j(a_m) & \cdots & x_n(a_m) \end{array}$$

10.3.1 Transformation of attributes

Information on alternatives in MADM can be captured by two kinds of attributes: qualitative and quantitative. For example, in a problem related to selection of a car, cost and mileage can be expressed in quantitative terms (in different units), whereas, reliability of technology would be expressed in qualitative terms.

Transformation of qualitative attributes into ratio scales is arduous; therefore, most of the MADM methods resort to either the ordinal scale or the interval scale (Rafiaani et al., 2019; Hwang and Yoon, 1981). The transformation of the qualitative attribute into ordinal scale is most commonly practiced. To transform the qualitative attribute to an interval scale, a 10-point scale can be chosen and may be calibrated in one of several ways.

Any negative values in the indicator scores also needs to be transformed, as negative values will affect the final outcome of MADM method. Shifting of all indicator score values above zero is commonly used method to handle negative values (Kalbar et al., 2012).

10.3.2 Normalization of attributes

Normalization of the attributes is not required in all of the MADM methods, but many compensatory MADM methods like maximin, simple additive weighting, TOPSIS, ELECTRE, etc., require normalization to perform the further mathematical procedures with comparable scales. Before proceeding towards normalization, it is important to note different types of attributes as given by Yoon and Hwang (1995).

- *Benefit attributes:* Offer an increasing monotonic utility; that is, the higher the attribute value, the more its preference; for example, fuel efficiency.
- *Cost attributes:* Offer a decreasing monotonic utility; that is, the higher the attribute value, the less its preference; for example, production cost.
- *Nonmonotonic attribute:* Offer nonmonotonic utility, such as room temperature in an office, or blood sugar level in human body, where maximum utility is located somewhere in the middle of an attribute range.

Shih et al. (2007) organized a few conventional normalization methods in tabular form based on the works of Milani et al. (2005), Yoon and Hwang (1995), and Hwang and Yoon (1981). Vector normalization and linear normalization are commonly used normalization methods in MADM.

10.3.3 Weighting attributes

It is almost common that DMs may have differences in preferences or importance for various attributes on which alternatives are to be evaluated or ranked. This preference or importance can be taken into consideration using assignments of weights to the attributes. The DM may use a cardinal or ordinal scale to express his or her preference among attributes. MADM methods require cardinal weights, that is $w = (w_1, \ldots, w_j, \ldots W_n)$, where w_j is weight assigned to the jth attribute. Cardinal weights are normalized to sum to 1, that is $\sum W_j = 1$. Hwang and Yoon (1981) reported four methods to assign weights, viz., eigenvector method, weighted least square method, entropy method, and linear programming techniques for multidimensional analysis of preferences (LINMAP).

10.3.4 Ranking of alternatives

Once the data is transformed and normalized, then the next step of MADM methodology is to rank the alternative using the attributes normalized score. Each of the MADM methods has its algorithm or procedure to aggregate and process the data on attributes. The outcome from MADM methods is most of the time ranking on some index, priority, or relative measure. MADM methods have their intrinsic properties and, hence, may generate different ranking for the same decision matrix. Therefore, after ranking of alternatives, sensitivity analysis is recommended to assess the effect of weights, attributes, and MADM methods. Some of the most commonly used MADM methodologies used in decision making have been briefly described in the following section.

10.4 Application of MADM in LCSA

LCSA attempts simultaneous use of different approaches (LCA, LCC, and SLCA) over an alternative and presents results of analysis in an integrated manner (Ness et al., 2007). Coupling of MADM with LCSA makes it possible to unite the results of the assessments obtained using different methods. Additionally, a majority of research on LCSA is focused on one of the three pillars (dimensions) of sustainability (environment, economic, and social) or, in some cases, covering two dimensions. The dimensions can consist of multiple indicators and decision making in LCSA requires judgment considering multiindicator trade-offs (Tarne et al., 2019). The following subsections discuss applications of MADM-based LCSA in various sectors, viz., construction, transport, water, energy, and production and consumption.

10.4.1 Construction

Integration of MADM methods with LCSA has seen a significant number of applications in the field of infrastructure construction. For example, Akhtar et al. (2015) attempt to develop an asset management plan by comparing four types of sewer systems, which are made up of (a) polyvinyl chloride (PVC), (b) ductile iron, (c) concrete, and (d) vitrified clay. For comparison, two LCSA frameworks are developed. The first framework is to integrate LCA and LCC with energy synthesis. Energy is used to convert integrated values from LCA and LCC into equivalent solar power. Whereas, the second framework deals with integrating AHP with LCC. This, therefore, suggests that the study does not consider social aspects during comparison and results suggest PVC pipe performs the best in both economic and environmental aspects. In another study by Dong and Ng (2016), an LCSA framework is developed for a residential housing complex in Hong Kong for an estimation from cradle to end of the construction process. Experts in the study suggest that integration of LCC, LCA, and SLCA must be performed using MADM methods. However, the selection of MADM method and weighting process must be left with stakeholders. Similarly, two six-story buildings, one made of wood and the other with concrete, from Vancouver is investigated using an LCSA framework with assessment from cradle to grave. The methodology utilizes AHP to develop a sustainability index from aggregating the impacts (Hossaini et al., 2015).

10.4.2 Transport

Transport is also a field where integrated MADM and sustainability assessment studies are present. A study was conducted by Sou et al. (2016) on bottom ash management from Macao to China and suggested that transportation was the most sensitive component with impacts including economic, social, and environmental aspects. Transportation impacts have considered additional impacts due to intermediate treatment of bottom ash and associated transport of materials of intermediate treatment. In another study, Steen and Palander (2016) attempt to identify safeguard subjects of critical resources and state indicators for LCSA. The study identifies transport technology as a major safeguard subject and associated transport capacity and transport efficiency as state indicators, therefore, state and central institutions provide incentives to promote costly transport technologies. Onat et al. (2016) conducted a study on assessing sustainable performance of alternative vehicle technologies considering a two-step approach. The first step includes use of microlevel indicators for environmental, economic, and social aspects to develop sustainability assessment model; and in the second step, TOPSIS is used to amalgamate results from the model to develop a final decision.

10.4.3 Water

Another area of infrastructure where studies with integrated application of MADM and LCSA methods are conducted is in the domain of water. Balkema et al. (2002) conducted a study to select wastewater treatment systems and integrated all three aspects of economic, social, and environment to develop a framework highlighting a selected set of sustainability indicators with trade-off between the indicators. On the other hand, Kalbar et al. (2016) attempt to compare technologies of wastewater treatment using a scenario-based decision-making tool. The study developed a tool named "TechSelect 1.0," which uses an LCSA framework in combination with TOPSIS methodology, while Opher et al. (2018) conducted a study to examine potential of reusing domestic wastewater. The study took input from 20 experts on multiple scenarios of domestic wastewater reuse. AHP method is used for estimating weights of sustainability indicators in an LCSA framework. In another study, Godskesen et al. (2018) attempted to identify a suitable technology for water supply in Copenhagen. The study integrated LCC and LCA estimates for multiple scenarios and weights of indicators from the AHP method.

10.4.4 Energy

Energy sectors involve taking decisions in areas such as energy management, selection of energy source, or form of energy output, which can produce either lower environmental impacts, social impacts, and economic viability, or all of them. A study was conducted by Martínez-Blanco et al. (2014) over agricultural energy source with all three components of LCSA (SLCA, LCA, and LCC) taken into consideration. The study suggested that geographical scale variations, assessment method, and indicator selection are major hurdles in the evaluation of SLCA. Similarly, difficulties in evaluation of SLCA are highlighted by Kunifuji et al. (2016), who integrated ELECTRE and LCA to compare power stations that operate on wind and thermal energy from North-East Brazil. In another study, LCSA of the electricity sector in Turkey is introduced by Atilgan and Azapagic (2016), in which three electricity alternatives (geothermal, hydro, and wind) are compared based on six social, eleven environmental, and three economic indicators. MADM method is used to consider all three aspects of sustainability simultaneously, and results indicate hydro-power to be the most sustainable solution. Similarly, Azapagic et al. (2016) attempted to compare electricity generation scenarios considering the United Kingdom's (UK) probable future mix by developing a framework for decision support named "DESIRES." The study utilized AHP for weight estimation for social, economic, and environmental indicators, and integrates the weights for analysis of LCA, LCC, and SLCA for a time horizon of the next 70 years and scope of "cradle to grave." In another type of application in the energy sector, Gumus et al. (2016) attempt to select the best wind turbine for wind energy in the United States (US). TOPSIS is used in combination with environmentally extended input-output based life cycle assessment (EE-IO-LCA) with multiple socio-economic and environmental indicators.

10.4.5 Consumption and production

The context of production and consumption is important in day to day life and overall operation of society. Therefore, this section attempts to focus on some application of integration between sustainability assessment with life cycle thinking and MADM. Foolmaun and Ramjeawon (2013) used AHP for combining LCC, LCA, and SLCA in a study on comparison of different methods of postconsumer polyethylene terephthalate (PET) bottles consumption for Mauritius. SLCA was based on UNEP/SETAC guidelines. Similarly, De Luca et al. (2015) also used AHP to develop a methodology to integrate SLCA with qualitative focus to compare three different crops of citrus from three different production areas of Calabria in Southern Italy. Whereas, Angelo et al. (2017) attempted to understand consumption pattern of food and waste generated from them. The study integrated LCA methodology with a multiattribute method to develop interactive software, which is used to identify preferred environmental options for household food waste. Kalbar et al. (2017a) conducted a study on proposing a method to calculate single scores, which is for environmental decision making and utilized residential consumption data from Denmark. The study suggests that a liner weighted sum method is not capable of providing a perspective of stakeholders realistically, and that TOPSIS, which is a distance-based method was found to be the best MADM method for that application. In another study, Tziolas et al. (2018) developed a tool that can assess production from agriculture in multistages involving multiple frameworks of MADM methods (AHP, VIKOR, ELECTRE, TOPSIS) with life cycle thinking. However, the focus is limited to understand environmental impacts. The above applications of integrated MADM with LCSA highlight that, although LCC and LCA are widely used, still the majority of studies are not focusing on SLCA.

10.5 Challenges in the application of MADM for LCSA

As discussed in previous sections, there are many studies applying MADM for LCSA. However, the detailed analysis of these applications shows that researchers have been facing numerous challenges while using MADM methods for LCSA application owing to the nature of LCSA indicators. Hence, below we have discussed in detail major challenges in application of MADM for LCSA.

10.5.1 Choice of MADM method

A detailed review by Greco et al. (2016) shows that many methods (more than 100) exist for solving discrete decision problems. Choosing an MADM method in itself can be posed as an MADM problem, as suggested by Guitouni and Martel (1998). The first step in using the MADM approach, as explained in Munda (2005), is to take a stand (value choice) whether to adopt compensatory approach or noncompensatory approach. Choosing any of these approaches automatically results in following "weak" sustainability assessments (in the case of MADM methods based on compensation principle) or strong sustainability assessments (in the case of noncompensatory assessments) (Kalbar et al., 2017a; Rowley et al., 2012; Munda, 2005). Kalbar et al. (2017a) specifically demonstrate that linear weighted sum (LWS), a more straightforward form of compensatory approach, favors extreme solutions.

Each of the MADM methods uses different mathematical principles, and hence, it is necessary to test more than one MADM methods in LCSA. For example, Kalbar et al. (2017a) report that the use of distance-based MADM (TOPSIS) is a more suited approach than a relative utility-based approach such as linear weighted sum (LWS) when ranking the individual's environmental footprint.

10.5.2 Rank reversal in MADM

Rank reversal, meaning change in ranks of the alternatives due to change in the MADM methods, or addition, or deletion of criteria, or change of weights, is a well-known and well-discussed phenomenon. Almost all methods of MADM has the rank reversal property (Mousavi-Nasab and Sotoudeh-Anvari, 2018; Mufazzal and Muzakkir, 2018). It can be concluded from the studies dealing with rank reversal that rank reversal is unavoidable and is an underlying property of the MADM approach. However, in some cases (e.g., problems with dominating alternatives) application of different MADM methods can result in selecting the same alternative as the most preferred one.

One of the ways to handle rank reversal is restructuring the decision making, e.g., scenario-based decision making, as demonstrated in Kalbar et al. (2012). The basic approach in scenario-based decision making is defined as the case/situation-specific weights. By applying the case-specific weights, more consistent ranking will be generated by any of the MADM methods.

Hence, it is recommended to structure the decision-problem correctly by articulating scenarios, and more than one method can be used to identify the most preferred alternative. Spearman's rank coefficient can be used for checking the agreement between the ranks generated by two different MADM methods and if there are more than two MADM methods used, Kendall's coefficient of concordance can be used, as demonstrated in Kalbar et al. (2015).

10.5.3 Dominating alternatives

In a decision problem, there could be multiple numbers of alternatives. Alternatives can be divided into two groups, such as alternatives that are dominated and nondominated alternatives (Kalbar et al., 2017a). An alternative can be called dominated if there exists another alternative performing better in at least one of selected attributes and performs equally in other attributes. In the case of a large number of alternatives in decision making, identifying and utilizing only the nondominant alternatives is essential and significantly decreases the efforts required for finding a feasible alternative (Calpine and Golding, 1976). The set of nondominated alternatives is also known as "Pareto-optimal." Removing dominated alternatives from the alternatives is optional; however, it increases unnecessary noise in the overall decision-making process and, considering limited processing capability of decision-makers, develops a condition which is ex ante worse. Literature suggests that presence of dominated alternatives in the decision matrix can cause asymmetric dominance effect or attraction effect (Huber et al., 1982), compromise effect (Simonson, 1989), and similarity effect (Tversky, 1972), thus influencing the decision process. Additionally, according to Montgomery and Willén (1999), the decision maker's tendency of justifying or protecting their selected alternative adds uncertainty. Therefore, initial development of Pareto-optimal set of alternatives is most desirable.

The process of identifying nondominant alternatives as an introductory refining process is a well-utilized concept throughout the literature. Hwang and Yoon (1981) suggested identification of the nondominated alternatives is tricky and, therefore, suggested some methods like "dominance," "permutation method," and "ELECTRE," which could help in the process of identification. Dominance method successively compares two alternatives at a time and deletes the dominated alternative in each step. Whereas, in the permutation method, first, attempts are made to identify the best ordering of ranking and then identify the dominated alternatives. ELECTRE method makes a pairwise comparison of alternatives and the weights help in supporting or denying the dominance relationship among alternatives. But, as mentioned in Section 10.5.1, a distance-based method like TOPSIS is more appropriate than relative utility-based method like LWS; therefore, for distance-based methods with internal reference data, Kalbar et al. (2017b) recommend choosing Hasse diagram technique as the most appropriate technique for identifying dominating alternatives. The method is appropriate due to its simplicity in a pictorial representation of relationships.

10.5.4 Consistency of inputs on indicators

The decision-maker provides information or inputs on indicators related to the decision problem. However, measuring consistency in the provided information is essential as it influences the results. Consistency can be affected by condition of preferential independence and dependency (Waas et al., 2014; Figueira et al., 2005). Preferential independence is the condition in which preference of one indicator over another indicator is not affected by any other existing indicator. However, any MADM problem is rarely free from this condition, as some of the selected indicators may have some level of interaction (Liou and Tzeng, 2012; Tzeng and Huang, 2011; Triantaphyllou, 2000). In MADM, there is a general assumption that indicators are independent of each other. Hence, the results of MADM are not certain, due to lack of understanding in trade-offs among indicators. A detailed approach to check preferential independence using the trade-off method of preference level of indicators is given in Figueira et al. (2005). It is recommended that if interdependence exists among a specific set of indicators, then decision-makers should attempt to group the indicators or break indicators with unique characteristics. Kalbar et al. (2017a) used linear regression across the indicator scores to assess preferential independence. Similarly, inputs of an indicator are consistent if it satisfies the condition of asymmetry, transitivity, and comparability.

A detailed discussion about the above requirements and their significance with examples is provided in French (1986). Seppälä et al. (2001) suggest that if inputs for an indicator have different units then to be consistent during analysis, all the indicators must be transformed into a common dimensionless unit using an appropriate normalization approach. Different methods use different aggregation methods with varied types of inputs and have different procedures for transformation of indicators. Therefore, it is recommended that an aggregation model must be selected before collecting inputs from stakeholders (Seppälä et al., 2001). In specific cases, a set of indicators may have negative values compared to another set of indicators with nonnegative values. During this type of scenario, the indicator scores should be normalized within a value range of 0 to 1 or −1 to 1 (excluding strict negative and positive values) using suitable normalization technique, thus helping in attaining consistency during analysis (Rowley, 2012).

10.5.5 Weighting of the indicators

Weighting is an essential step in MADM, which facilitates the incorporation of stakeholders' preferences. Different weighting sets are subjective, and representing various stakeholder groups, can be formed to observe the change in the results (Kalbar et al., 2017a). Zardari et al. (2015) report that pairwise comparison, point allocation, rating methods, trade-off analysis, and ranking methods are commonly used methods for weighting of indicators. Each of these weighting methods has its extent of disadvantages (inaccuracy, confusions regarding foundation of involved theory, and complexity). For example, Wang et al. (2009) suggest the "equal weights method" is the easiest and most popular method of assigning weights and requires minimal knowledge or input from decision-maker; however, equal weights method does not take into account difference in criteria and their significance in a decision problem. Also, equal weights method does not weight the attribute equally in its absolute sense, in the case where more than one attribute characterizes a criterion.

Ahlroth et al. (2011) provided a taxonomy of all available weighting methods divided into monetary and nonmonetary methods. Kalbar et al. (2017a) report that the most commonly used monetary methods of weighting are converting impacts or damages into monetary valuation using willingness-to-pay, and converting damages into cost incurred and midpoint impacts; and the most commonly used nonmonetary weighting methods are distance-to-target and panel methods. Panel methods can also be known as subjective weighting (SW). There are a number of other methods for eliciting weights. Wang et al. (2009) suggests that other methods of weighting are objective weighting (OW) and combined weighting (CW). In OW, weight is obtained from mathematical models. CW is a combination of both SW and OW. Pair-wise comparison, entropy, and additive synthesis are some of the popular SW, OW, and CW methods, respectively. Zardari et al. (2015) and Eckenrode (1965) suggest that methods that directly take weights may not be accurate; therefore, a method should be selected that derives weights from given information. Additionally, one must also look at some other matters during selection of weighting method such as type of scale use, time required to collect information and analysis, and DM's understanding of the domain. For example, the pair-wise comparison is suitable for a lower number of indicators, but for a significantly higher number of indicators, the ranking method becomes efficient. Hobbs (1980) showcased that different weighting methods lead to different results. Therefore, the decision of an appropriate weighting method is crucial for MADM. Scenario creation in the decision-making problem helps in formulating case-specific weighting set (Kalbar et al., 2012), and conducting sensitivity analysis of the weights is also important (Dhiman et al., 2018).

10.5.6 Uncertainty and sensitivity analysis

To validate results from a selected MCDA method, both uncertainty and sensitivity analysis is essential. Kleijnen (1994) suggests that sensitivity analysis is also known as "what-if" analysis, which is when the model is subjected to extreme values and limited to a set of scenarios in which a real system could be analyzed and experimented (Figueira et al., 2005). For example, what if the queue in a line doubles, or what-if a rule is changed for a service from first-in-first-out (FIFO) to last-in-first-out (LIFO). Sensitivity analysis answers two problems. One is to understand criticality of each indicator in an overall change of results and second is to identify by what extent of alteration could change the overall results (Kalbar et al., 2012; Triantaphyllou, 2000). Whereas, there will always be inherent uncertainty involved in a decision problem, as a DM does not know everything with certainty, and complexity of a decision increases with increased uncertainties, therefore, a DM should always try to minimize uncertainties associated with different areas of a decision problem to find out the best solution possible (Nikolaidis et al., 2004).

Nikolaidis et al. (2004) classify uncertainty into two categories: aleatory and epistemic. Aleatory uncertainty is the uncertainties that are out of the scope of DMs, whereas, epistemic is fully dependent on the set of choices made by DMs. There are majorly four types of epistemic uncertainties, which are related to (i) data uncertainty, (ii) weighting uncertainty, (iii) normalization uncertainty, and (iv) indicator uncertainty (Miller et al., 2017; Beltran et al., 2016; Clavreul et al., 2013). Data uncertainty involves use of inaccurate data or inputs with multiple values for analysis, thus resulting in varied models depicting reallife scenarios. Whereas, there are different approaches to gather weight or assigning weight with varying levels of stakeholder involvement and analysis (Miller et al., 2017; Zardari et al., 2015). Data and weighting uncertainties get effected by stochastic, parameter, heterogeneity, and structural uncertainties, and detail of these uncertainties with examples are provided in Briggs et al. (2012). A detailed methodology and Monte Carlo simulation is suggested by Barfod and Salling (2015) to handle data and weighting uncertainty, respectively. Huppes and van Oers (2011) suggest that weighting is done to compare different types of impacts in LCSA, however, for comparing different types of impacts, there is a need to convert different impacts to a same level or unit.

Normalization helps in the conversion of impacts and is a mandatory step in the integration of MADM with LCSA. However, there are different normalization techniques available in the literature, mentioned in Section 10.3.2, having different procedures to handle low and high values (Miller et al., 2017). A good practice to minimize normalization uncertainty is to use different normalization techniques on the same problem and make a judicious decision on top performers (Kalbar et al., 2017a). Finally, indicator uncertainty involves selection of indicators in a study that are irrelevant or incomplete (Heijungs and Huijbregts, 2004). Additionally, uncertainties could also be associated with the framing of the problem, selection of method for aggregation, and levels of selected attributes (Scholten et al., 2015). There is no clearly documented way to fully remove the uncertainty as the indicator selection is a subjective process and depends on the domain(s) knowledge of involved researcher(s).

Tzeng and Huang (2011) suggest that before proceeding with MADM analysis, data must be put into a histogram and their distribution with standard deviation should be checked. If the distribution is nonnormal and standard deviation is significant, then sensitivity analysis is mandatory during MADM analysis. Sensitivity analysis must precede uncertainty analysis (Kleijnen, 1994).

10.5.7 Interpretation of the results

Results from an analysis in MCDA typically involves ranking of alternatives concerning a specific set of attributes. The results are obtained in the form of an aggregated index and need further interpretation for deriving correct decision support. For example, Figueira et al. (2005) and Munda (2005) conducted studies on four different cities using a distance-based method such as TOPSIS. The studies suggested that results from MADM analysis cannot blindly be relied upon. Even if different types of aggregation schemes are used and still the results are not robust, then reconsideration of areas related to uncertainties mentioned in Section 10.5.6 must be completed. Therefore, it is recommended that in MADM analysis, robustness of the decision process is more critical compared to the final solution.

Similarly, LCSA also has a major challenge in its interpretation of results, where the integration of three different tools (LCA, LCC, and SLCA) is required to produce a collective result (Hannouf and Assefa, 2017). Zhang and Haapala (2015) suggested the use of MADM approaches as an efficient way of developing frameworks to integrate the tools and interpret combined results. Zampori et al. (2016) provided general guidelines to interpret results, in which identification of significant issues can be achieved by the use of MADM methods. Additionally, the study also recommended conducting thorough checks like completeness of inventory data, sensitivity analysis to assess reliability of results, and consistency check of methods and assumptions. There has not been sufficient work on the interpretation of results in MADM integrated with LCSA. One of the efforts for interpretation of MADM results is using radar diagrams, as demonstrated by Kalbar et al. (2012).

The results of ranking in LCSA based on MADM are an aggregated score, i.e., a single value for each indicator. Considering all the methodological choices, data uncertainties, effects of weights, and MADM methods limitations (e.g., rank reversal), unless the topmost ranked alternatives have a significant difference in the score from the second most alternative, that alternative cannot be concluded as the best performing one. For example, Kalbar et al. (2016) implemented an approach wherein such cases, the top two to three alternatives having almost equal scores will be concluded as most-preferred alternatives.

In a reallife situation, as best practice, it is recommended to apply multiple MADM methods for the given problem with different weighting schemes reflecting the priorities of stakeholders. The alternatives that are frequently ranked as topmost can be concluded as the most preferred ones.

10.6 Proposed framework for MADM based LCSA

The challenges discussed above suggest that there is a need for a systematic framework for carrying out LCSA based on MADM. The literature shows that many studies have attempted to use more refined indicators in decision making. For example, end points obtained from LCA are used for decision making, however considering the uncertainties in the impact assessment and damage models, such indicators may not be currently suited for LCSA. Hence, as demonstrated in Niero and Kalbar (2019) and Sohn et al. (2017), midpoint indicators from LCA can be combined with other indicators from material intensity, energy intensity, as well as economic and social indicators using MADM. Here we propose such a framework for MADM based LCSA, as shown in Fig. 10.1.

Step 1: Setting up goals and indicators.
Sustainable development is majorly dependent on three pillars (social, economic, and environmental). Each of the pillars includes multiple indicators, which explains the pillar's major area of concentration. Selection of the indicators for each of the pillars must consider geographic or regional socio-economic-environmental suitability. Therefore, the indicators selected must be dependent on the problem into consideration and available resources in front of DMs without forgetting the core of sustainable development.

The steps which can be followed during selection of indicators are as follows:

i. Firstly, define the problem that needs to be solved.
ii. Break the problem into smaller components or subproblems or scenarios.
iii. Identify methods/indicators from the literature which are suitable to the subproblems.
iv. Filter the methods/indicators according to suitability and overlapping, and if necessary, modify methods/indicators as per regional requirements.
v. Finally, review the methods/indicators for its relevance

Step 2: Divide the selected sustainability assessment methods/indicators into social, environmental, and economic criteria.

Step 3: Perform detailed analysis using methods such as LCA to obtain environmental indicators, LCC to obtain economic indicators, and SLCA to get social indicators.

Step 4: Transformation and normalization: transform the attributes and carry out normalization to obtain indicator scores in commensurate units.

Step 5: MADM method
Choice of MADM method: Necessary to test multiple MADM methods.
Selection of MADM method must consider the type of data available on selected indicators (qualitative and quantitative), representation of results (performance score, distance to target, ranking, visual interpretation or probability), transparency of a method, computational time, and cost of data collection.
Rank reversal: Restructuring of decision problem through scenario-based decision making by applying case-specific weights can address the issue of rank reversal to some extent.
Deletion of dominating alternative: Any of the methods such as dominance method or Jaquet-Lagrkze's successive permutations method or any outranking approach is suitable, and each of these methods has their limitations. However, Hasse diagram

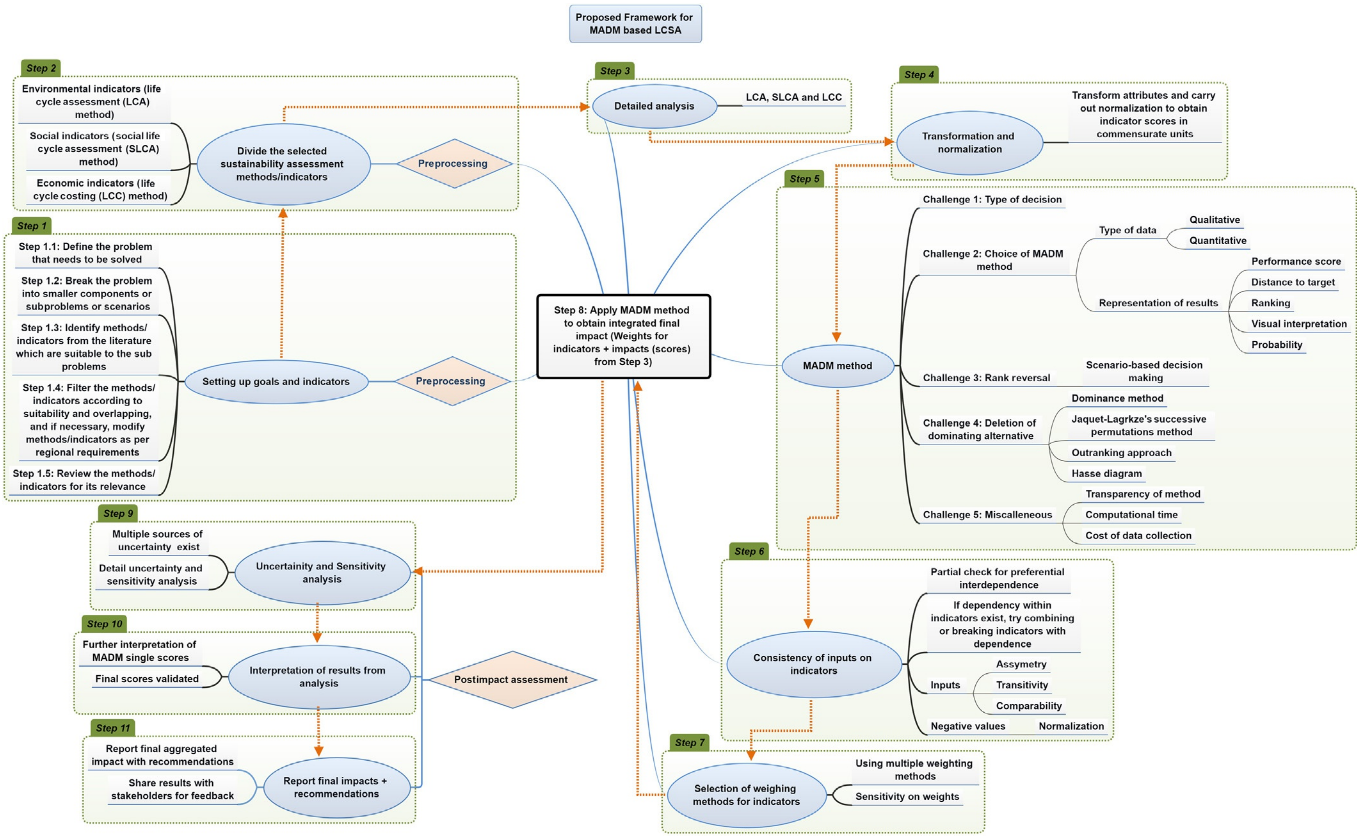

FIG. 10.1 Proposed framework for LCSA using MADM.

method seems to be a more appropriate technique due to its simplicity and pictorial representation of relationships.

Step 6: Consistency of inputs on indicators: Partial check for preferential interdependence must be checked. In case there exists dependency within the indicators, indicators selection can be revisited in Step 1. For example, combine multiple indicators with dependence or break dependent indicators into multiple subindicators. Similarly, inputs must be checked for asymmetry, transitivity, and comparability, and if negative values of indicators exist, then normalization of data is appropriate.

Step 7: Selection of weighting methods for indicators: Different weighting method produce different results. Therefore, use of multiple weighting methods is recommended and carrying out sensitivity on weights.

Step 8: Apply MADM method using the weights for indicators and impacts (Scores) from Step 3 to obtain integrated final impact.

Step 9: Uncertainty and sensitivity analysis: As there are number of sources of uncertainty in LCSA based MADM approach, detailed uncertainty and sensitivity analysis should be performed.

Step 10: Interpretation of results from the analysis: MADM results are usually obtained as single scores and hence need further interpretation. This is an essential step where the final scores should be validated with the data and methods used for LCSA.

Step 11: Report the final aggregated impact with recommendations and share the results with stakeholders for feedback.

10.7 Conclusions

LCSA is a fast-developing field, and numerous efforts are being made to refine the framework and associated methods used for sustainability assessment. In this work, we have taken stock of using different MADM methods for LCSA. The basic structure of LCSA is described in detail and highlighted the suitability of MADM methods in integrating indicators with LCSA.

The review of applications of MADM for LCSA showed that there are numerous challenges of applying MADM to LCSA. The challenges of MADM application are discussed in detail. A framework is proposed for carrying out LCSA using MADM. The framework is also able to highlight tackling of challenges in integrating MCDA with LCSA, such as, dominating alternatives, choice of appropriate MADM method, consistency of inputs on indicators, selection of weighting methods for indicators, and uncertainty and sensitivity analysis.

One of the critical issues identified is the choice of MADM method for LCSA. It is recommended that there is no unique suitable MADM method for LCSA, and hence, it is suggested to define scenarios for the given decision-making situation in LCSA. Once the scenarios are articulated, accordingly, more refined weights can be given to the indicators. Using this set of weights, if more than one MADM method ranks the same alternative as most preferred then such an alternative can be conclusively identified as more sustainable than the other one based on LCSA coupled with MADM approach. In addition, there exist many methodological uncertainties while implementing LCSA (choice of assessment methods, data quality issues, choice of MADM methods), and hence, we recommend that unless the alternative ranked at the top has a significant difference in the score than other alternatives; all top 2–3 alternatives should be chosen as best ones.

References

Ahlroth, S., Nilsson, M., Finnveden, G., Hjelm, O., Hochschorner, E., 2011. Weighting and valuation in selected environmental systems analysis tools–suggestions for further developments. J. Clean. Prod. 19 (2–3), 145–156.

Akhtar, S., Reza, B., Hewage, K., Shahriar, A., Zargar, A., Sadiq, R., 2015. Life cycle sustainability assessment (LCSA) for selection of sewer pipe materials. Clean Techn. Environ. Policy 17 (4), 973–992.

Angelo, A.C.M., Saraiva, A.B., Clímaco, J.C.N., Infante, C.E., Valle, R., 2017. Life cycle assessment and multi-criteria decision analysis: selection of a strategy for domestic food waste management in Rio de Janeiro. J. Clean. Prod. 143, 744–756.

Asgharizadeh, E., Taghizadeh Yazdi, M., Mohammadi Balani, A., 2017. An output-oriented classification of multiple attribute decision-making techniques based on fuzzy c-means clustering method. Int. Trans. Oper. Res 26 (6), 2476–2493.

Atilgan, B., Azapagic, A., 2016. An integrated life cycle sustainability assessment of electricity generation in Turkey. Energy Policy 93, 168–186.

Azapagic, A., 2010. Assessing environmental sustainability: life cycle thinking and life cycle assessment. In: Sustainable Development in Practice: Case Studies for Engineers and Scientists, John Wiley & Sons, Ltd., England, pp. 56–80.

Azapagic, A., Stamford, L., Youds, L., Barteczko-Hibbert, C., 2016. Towards sustainable production and consumption: a novel decision-support framework integrating economic, environmental and social sustainability (DESIRES). Comput. Chem. Eng. 91, 93–103.

Balkema, A.J., Preisig, H.A., Otterpohl, R., Lambert, F.J., 2002. Indicators for the sustainability assessment of wastewater treatment systems. Urban Water 4 (2), 153–161.

Barfod, M.B., Salling, K.B., 2015. A new composite decision support framework for strategic and sustainable transport appraisals. Transp. Res. A Policy Pract. 72, 1–15.

Beltran, A.M., Heijungs, R., Guinée, J., Tukker, A., 2016. A pseudo-statistical approach to treat choice uncertainty: the example of partitioning allocation methods. Int. J. Life Cycle Assess. 21 (2), 252–264.

Briggs, A.H., Weinstein, M.C., Fenwick, E.A., Karnon, J., Sculpher, M.J., Paltiel, A.D., ISPOR-SMDM Modeling Good Research Practices Task Force, 2012. Model parameter estimation and uncertainty: a report of the ISPOR-SMDM Modeling Good Research Practices Task Force-6. Value Health 15 (6), 835–842.

Calpine, H.C., Golding, A., 1976. Some properties of Pareto-optimal choices in decision problems. Omega 4 (2), 141–147.

Chen, S.J., Hwang, C.L., 1992. Fuzzy multiple attribute decision making methods. In: Fuzzy Multiple Attribute Decision Making. Springer, Berlin, Heidelberg, pp. 289–486.

Clavreul, J., Guyonnet, D., Tonini, D., Christensen, T.H., 2013. Stochastic and epistemic uncertainty propagation in LCA. Int. J. Life Cycle Assess. 18 (7), 1393–1403.

Costa, D., Quinteiro, P., Dias, A.C., 2019. A systematic review of life cycle sustainability assessment: current state, methodological challenges, and implementation issues. Sci. Total Environ.

De Luca, A.I., Iofrida, N., Strano, A., Falcone, G., Gulisano, G., 2015. Social life cycle assessment and participatory approaches: a methodological proposal applied to citrus farming in southern Italy. Integr. Environ. Assess. Manag. 11 (3), 383–396.

Dhiman, R., Kalbar, P., Inamdar, A.B., 2018. GIS coupled multiple criteria decision making approach for classifying urban coastal areas in India. Habitat Int. 71, 125–134.

Dong, Y.H., Ng, S.T., 2016. A modeling framework to evaluate sustainability of building construction based on LCSA. Int. J. Life Cycle Assess. 21 (4), 555–568.

Eckenrode, R.T., 1965. Weighting multiple criteria. Manag. Sci. 12 (3), 180–192.

Fauzi, R.T., Lavoie, P., Sorelli, L., Heidari, M.D., Amor, B., 2019. Exploring the current challenges and opportunities of life cycle sustainability assessment. Sustainability 11 (3), 636.

Figueira, J., Greco, S., Ehrgott, M. (Eds.), 2005. Multiple Criteria Decision Analysis: State of the Art Surveys. vol. 78. Springer Science & Business Media.

Finkbeiner, M., Schau, E.M., Lehmann, A., Traverso, M., 2010. Towards life cycle sustainability assessment. Sustainability 2 (10), 3309–3322.

Foolmaun, R.K., Ramjeawon, T., 2013. Life cycle sustainability assessments (LCSA) of four disposal scenarios for used polyethylene terephthalate (PET) bottles in Mauritius. Environ. Dev. Sustain. 15 (3), 783–806.

French, S., 1986. Decision Theory: An Introduction to the Mathematics of Rationality. Halsted Press.

Godskesen, B., Hauschild, M., Albrechtsen, H.J., Rygaard, M., 2018. ASTA—a method for multi-criteria evaluation of water supply technologies to assess the most sustainable alternative for Copenhagen. Sci. Total Environ. 618, 399–408.

Greco, S., Figueira, J., Ehrgott, M., 2016. Multiple Criteria Decision Analysis. Springer, New York.

Guinée, J., 2016. Life cycle sustainability assessment: what is it and what are its challenges? In: Taking stock of industrial ecology. Springer, Cham, pp. 45–68.

Guitouni, A., Martel, J.M., 1998. Tentative guidelines to help choosing an appropriate MCDA method. Eur. J. Oper. Res. 109 (2), 501–521.

Gumus, S., Kucukvar, M., Tatari, O., 2016. Intuitionistic fuzzy multi-criteria decision making framework based on life cycle environmental, economic and social impacts: the case of US wind energy. Sustain. Prod. Consump. 8, 78–92.

Halog, A., Manik, Y., 2011. Advancing integrated systems modelling framework for life cycle sustainability assessment. Sustainability 3 (2), 469–499.

Hannouf, M., Assefa, G., 2017. Life cycle sustainability assessment for sustainability improvements: a case study of high-density polyethylene production in Alberta, Canada. Sustainability 9 (12), 2332.

Hannouf, M., Assefa, G., 2018. A life cycle sustainability assessment-based decision-analysis framework. Sustainability 10 (11), 3863.

Heijungs, R., Huijbregts, M.A., 2004. A Review of Approaches to Treat Uncertainty in LCA. Elsevier, Osnabrück, Germany.

Hobbs, B.F., 1980. A comparison of weighting methods in power plant siting. Decis. Sci. 11 (4), 725–737.

Hossaini, N., Reza, B., Akhtar, S., Sadiq, R., Hewage, K., 2015. AHP based life cycle sustainability assessment (LCSA) framework: a case study of six storey wood frame and concrete frame buildings in Vancouver. J. Environ. Plan. Manag. 58 (7), 1217–1241.

Huber, J., Payne, J.W., Puto, C., 1982. Adding asymmetrically dominated alternatives: violations of regularity and the similarity hypothesis. J. Consum. Res. 9 (1), 90–98.

Huppes, G., van Oers, L., 2011. Background Review of Existing Weighting Approaches in Life Cycle Impact Assessment (LCIA). Luxembourg, European Union.

Hwang, C.L., Yoon, K., 1981. Multiple Attribute Decision Making: A State of the Art Survey. Lecture Notes in Economics and Mathematical Systems, vol. 186. Palgrave Macmillan, United Kingdom

Kahraman, C., Onar, S.C., Oztaysi, B., 2015. Fuzzy multicriteria decision-making: a literature review. Int. J. Comput. Intell. Syst. 8 (4), 637–666.

Kalbar, P.P., Karmakar, S., Asolekar, S.R., 2012. Selection of an appropriate wastewater treatment technology: a scenario-based multiple-attribute decision-making approach. J. Environ. Manag. 113, 158–169.

Kalbar, P.P., Karmakar, S., Asolekar, S.R., 2013. The influence of expert opinions on the selection of wastewater treatment alternatives: a group decision-making approach. J. Environ. Manag. 128, 844–851.

Kalbar, P.P., Karmakar, S., Asolekar, S.R., 2015. Selection of wastewater treatment alternative: significance of choosing MADM method. Environ. Eng. Manage. J. 14(5).

Kalbar, P.P., Karmakar, S., Asolekar, S.R., 2016. Life cycle-based decision support tool for selection of wastewater treatment alternatives. J. Clean. Prod. 117, 64–72.

Kalbar, P.P., Birkved, M., Nygaard, S.E., Hauschild, M., 2017a. Weighting and aggregation in life cycle assessment: do present aggregated single scores provide correct decision support? J. Ind. Ecol. 21 (6), 1591–1600.

Kalbar, P., Birkved, M., Elsborg Nygaard, S., Hauschild, M., 2017b. Response to comment on "weighting and aggregation in life cycle assessment: do present aggregated single scores provide correct decision support?". J. Ind. Ecol. 21 (6), 1603–1605.

Kleijnen, J.P., 1994. Sensitivity analysis versus uncertainty analysis: when to use what? In: Predictability and Nonlinear Modelling in Natural Sciences and Economics. Springer, Dordrecht, pp. 322–333.

Kunifuji, S., Papadopoulos, G.A., Skulimowski, A.M., Kacprzyk, J., 2016. Knowledge, Information and Creativity Support Systems. Springer-Verlag, Berlin, Heidelberg

Liou, J.J., Tzeng, G.H., 2012. Comments on "multiple criteria decision making (MCDM) methods in economics: an overview". Technol. Econ. Dev. Econ. 18 (4), 672–695.

Martínez-Blanco, J., Lehmann, A., Muñoz, P., Antón, A., Traverso, M., Rieradevall, J., Finkbeiner, M., 2014. Application challenges for the social life cycle assessment of fertilizers within life cycle sustainability assessment. J. Clean. Prod. 69, 34–48.

Milani, A.S., Shanian, A., Madoliat, R., Nemes, J.A., 2005. The effect of normalization norms in multiple attribute decision making models: a case study in gear material selection. Struct. Multidiscip. Optim. 29 (4), 312–318.

Miller, P., de Barros, A.G., Kattan, L., Wirasinghe, S.C., 2017. Managing uncertainty in the application of composite sustainability indicators to transit analysis. Transport. Res. Procedia 25, 4003–4018.

Montgomery, H., Willén, H., 1999. Decision making and action: the search for a good structure. In: Judgment and decision making. Lawrence Erlbaum Associates, New Jersey, pp. 155–182.

Mousavi-Nasab, S.H., Sotoudeh-Anvari, A., 2018. A new multi-criteria decision making approach for sustainable material selection problem: a critical study on rank reversal problem. J. Clean. Prod. 182, 466–484.

Mufazzal, S., Muzakkir, S.M., 2018. A new multi-criterion decision making (MCDM) method based on proximity indexed value for minimizing rank reversals. Comput. Ind. Eng. 119, 427–438.

Munda, G., 2005. "Measuring sustainability": a multi-criterion framework. Environ. Dev. Sustain. 7 (1), 117–134.

Naves, A.X., Barreneche, C., Fernández, A.I., Cabeza, L.F., Haddad, A.N., Boer, D., 2018. Life cycle costing as a bottom line for the life cycle sustainability assessment in the solar energy sector: a review. Sol. Energy.

Ness, B., Urbel-Piirsalu, E., Anderberg, S., Olsson, L., 2007. Categorising tools for sustainability assessment. Ecol. Econ. 60 (3), 498–508.

Neugebauer, S., Martinez-Blanco, J., Scheumann, R., Finkbeiner, M., 2015. Enhancing the practical implementation of life cycle sustainability assessment–proposal of a tiered approach. J. Clean. Prod. 102, 165–176.

Niero, M., Kalbar, P.P., 2019. Coupling material circularity indicators and life cycle based indicators: a proposal to advance the assessment of circular economy strategies at the product level. Resour. Conserv. Recycl. 140, 305–312.

Nikolaidis, E., Ghiocel, D.M., Singhal, S., 2004. Types of uncertainty in design decision making. In: Engineering Design Reliability Handbook. CRC Press, pp. 137–156.

Onat, N.C., Gumus, S., Kucukvar, M., Tatari, O., 2016. Application of the TOPSIS and intuitionistic fuzzy set approaches for ranking the life cycle sustainability performance of alternative vehicle technologies. Sustain. Prod. Consump. 6, 12–25.

Opher, T., Friedler, E., Shapira, A., 2018. Comparative life cycle sustainability assessment of urban water reuse at various centralization scales. Int. J. Life Cycle Assess., 1–14.

Pizzirani, S., McLaren, S.J., Seadon, J.K., 2014. Is there a place for culture in life cycle sustainability assessment? Int. J. Life Cycle Assess. 19 (6), 1316–1330.

Rafiaani, P., Dikopoulou, Z., Van Dael, M., Kuppens, T., Azadi, H., Lebailly, P., Van Passel, S., 2019. Identifying social indicators for sustainability assessment of CCU technologies: a modified multi-criteria decision making. Soc. Indic. Res., 1–30.

Rowley, H.V., Peters, G.M., Lundie, S., Moore, S.J., 2012. Aggregating sustainability indicators: beyond the weighted sum. J. Environ. Manag. 111, 24–33.

Scholten, L., Schuwirth, N., Reichert, P., Lienert, J., 2015. Tackling uncertainty in multi-criteria decision analysis–an application to water supply infrastructure planning. Eur. J. Oper. Res. 242 (1), 243–260.

Seppälä, J., Basson, L., Norris, G.A., 2001. Decision analysis frameworks for life-cycle impact assessment. J. Ind. Ecol. 5 (4), 45–68.

Shih, H.S., Shyur, H.J., Lee, E.S., 2007. An extension of TOPSIS for group decision making. Math. Comput. Model. 45 (7–8), 801–813.

Simonson, I., 1989. Choice based on reasons: the case of attraction and compromise effects. J. Consum. Res. 16 (2), 158–174.

Sohn, J.L., Kalbar, P.P., Birkved, M., 2017. Life cycle based dynamic assessment coupled with multiple criteria decision analysis: a case study of determining an optimal building insulation level. J. Clean. Prod. 162, 449–457.

Sou, W.I., Chu, A., Chiueh, P.T., 2016. Sustainability assessment and prioritisation of bottom ash management in Macao. Waste Manag. Res. 34 (12), 1275–1282.

Steen, B., Palander, S., 2016. A selection of safeguard subjects and state indicators for sustainability assessments. Int. J. Life Cycle Assess. 21 (6), 861–874.

Tarne, P., Lehmann, A., Finkbeiner, M., 2019. Introducing weights to life cycle sustainability assessment—how do decision-makers weight sustainability dimensions? Int. J. Life Cycle Assess. 24 (3), 530–542.

Triantaphyllou, E., 2000. Multi-criteria decision making methods. In: Multi-Criteria Decision Making Methods: A Comparative Study. Springer, Boston, MA, pp. 5–21.

Tversky, A., 1972. Elimination by aspects: a theory of choice. Psychol. Rev. 79, 281–299.

Tzeng, G.H., Huang, J.J., 2011. Multiple Attribute Decision Making: Methods and Applications. Chapman and Hall/CRC.

Tziolas, E., Bournaris, T., Manos, B., Nastis, S., 2018. Life cycle assessment and multi-criteria analysis in agriculture: synergies and insights. In: Multicriteria analysis in agriculture. Springer, Cham, pp. 289–321.

Valdivia, S., Ugaya, C.M., Hildenbrand, J., Traverso, M., Mazijn, B., Sonnemann, G., 2013. A UNEP/SETAC approach towards a life cycle sustainability assessment—our contribution to Rio+ 20. Int. J. Life Cycle Assess. 18 (9), 1673–1685.

Waas, T., Hugé, J., Block, T., Wright, T., Benitez-Capistros, F., Verbruggen, A., 2014. Sustainability assessment and indicators: tools in a decision-making strategy for sustainable development. Sustainability 6 (9), 5512–5534.

Wang, J.J., Jing, Y.Y., Zhang, C.F., Zhao, J.H., 2009. Review on multi-criteria decision analysis aid in sustainable energy decision-making. Renew. Sust. Energ. Rev. 13 (9), 2263–2278.

Yadav, V., Karmakar, S., Kalbar, P.P., Dikshit, A.K., 2019. PyTOPS: a Python based tool for TOPSIS. SoftwareX 9, 217–222.

Yoon, K.P., Hwang, C.L., 1995. Multiple Attribute Decision Making: An Introduction. vol. 104. Sage publications.

Zamagni, A., Pesonen, H.L., Swarr, T., 2013. From LCA to life cycle sustainability assessment: concept, practice and future directions. Int. J. Life Cycle Assess. 18 (9), 1637–1641.

Zampori, L., Saouter, E., Schau, E., Cristobal Garcia, J., Castellani, V., Sala, S., 2016. Guide for Interpreting Life Cycle Assessment Result. Publications Office of the European Union, Luxembourg.

Zardari, N.H., Ahmed, K., Shirazi, S.M., Yusop, Z.B., 2015. Weighting Methods and Their Effects on Multi-Criteria Decision-Making Model Outcomes in Water Resources Management. Springer.

Zhang, H., Haapala, K.R., 2015. Integrating sustainable manufacturing assessment into decision making for a production work cell. J. Clean. Prod. 105, 52–63.

Zopounidis, C., Pardalos, P.M. (Eds.), 2010. Handbook of Multicriteria Analysis. In: vol. 103. Springer Science & Business Media.

CHAPTER

11

A composite life cycle sustainability index for sustainability prioritization of industrial systems

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11.1 Introduction

Although industrial systems play a critical role in the global economy, they also cause a variety of environmental burdens and social issues. Therefore, the concept of sustainability or sustainable development has been a hot topic for the industrial systems, which aims to improve the performances of environmental impacts, economic prosperity, and social responsibilities, simultaneously (Heijungs et al., 2010; Xu et al., 2018a). However, the traditional sustainability assessment only focuses on the manufacturing stage of the industrial system, failing to track the comprehensive performance/impacts with respect to the stages of construction, production, transportation, utilization, and disposal of the investigated system. For addressing this issue, a life cycle perspective can bring powerful insights into the sustainability assessment of industrial systems, by providing extended information on the traditional triple-bottom-line (TBL) sustainability, as depicted in Fig. 11.1, where environmental, economic, and social concerns can be fully collected and then evaluated along the whole supply chain, from extraction of raw material to its end of life (Heijungs et al., 2010; Xu et al., 2017). As an emerging method for assessing the sustainability of industrial systems with life cycle thinking, the life cycle sustainability assessment (LCSA) could be essentially denoted as the combination of three life cycle-based assessment tools, i.e., the environmental life cycle assessment (E-LCA or LCA) for environmental impacts, life cycle costing (LCC) for

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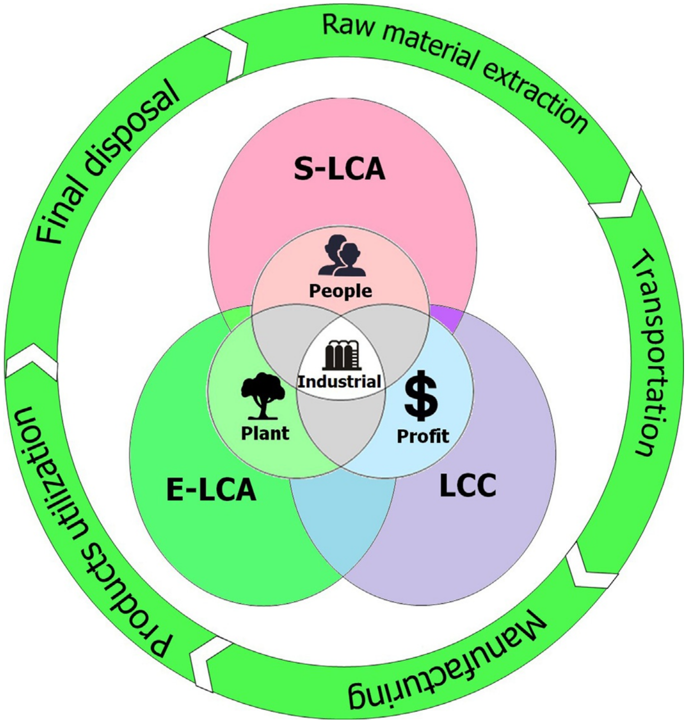

FIG. 11.1 From sustainability assessment to life cycle sustainability assessment.

economic prosperity, and social life cycle assessment (S-LCA) for social responsibility, which is preferred to be denoted as LCSA = E-LCA + LCC + S-LCA (Heijungs et al., 2010).

However, assessing the sustainability of industrial systems from a life cycle perspective is still at its development stage. In order to make a contribution to this issue, this chapter offers a review regarding the life cycle sustainability criteria for the prioritization of industrial systems; in which, typical environmental, economic, and social indicators from the well-known tools of E-LCA, LCC, and S-LCA are summarized, while some other typical indicators that can be employed for evaluating the life cycle-based sustainability of the industrial systems are also introduced. Subsequently, a composite life cycle sustainability index for the prioritization of industrial systems is developed, which can effectively aggregate multiple criteria from the environmental, economic, and social concerns into a composite index. The developed composite index is characterized by integrating the absolute score and relative balance of the multi-criteria in a compromise way for offering a rigorous ranking result, while its feasibility and robustness are also confirmed by implementing a case study and sensitivity analysis.

11.2 Life cycle environmental indicators

With an increasing prominence of environmental problems from the middle of the twentieth century, multiple indicators are available for evaluating the environmental performance of the industrial systems. In this section, the indicators from the life cycle assessment, from the footprint assessment, and some typical indicators that can be employed for evaluating the life cycle-based environmental performance of industrial systems are introduced.

11.2.1 Introduction of the environmental-life cycle assessment

Environmental life cycle assessment (E-LCA), also known as life cycle assessment (LCA), is an approach that covers a wide range of environmental concerns regarding a product throughout its entire lifetime, from raw material acquisition to the disposal of the product at the end. According to the literature (De Menna et al., 2018; UNEP/SETAC, 2009; Verma and Kumar, 2015), there are four stages involved in the E-LCA:

1. Determination of purpose and scope: this stage describes the functional units, system boundaries, data distribution process, and data quality requirements of an investigated system.
2. Inventory analysis: this stage establishes a list of inputs and outputs regarding the energy and material data, where the demand data for calculating the assessment indicators can be collected.
3. Impact assessment: this stage converts the collected demand data into specific impact types and indicator parameters, for facilitating the understanding of the environmental impact of the system from the life cycle perspective.
4. Interpretation of the results: this stage offers further interpretations by testing the completeness, sensitivity, and consistency. The final conclusions, suggestions, and limitations of the system can be given in this stage.

11.2.2 Environmental indicators from E-LCA

With an increasing interest in sustainability, E-LCA has become a very popular tool for representing the environmental concerns of the industrial systems from the life cycle perspective. Naturally, the indicators within the E-LCA have been frequently employed for sustainability prioritization. In the life cycle sustainability assessment, E-LCA is considered as a systematic tool that evaluates the environmental impacts occurring throughout the entire life cycle of an industrial product, process, or activity. In E-LCA, various indicators can be used for the environmental assessment, which are usually the manifestation of some environmental problems (Hermann et al., 2007). Among the popular E-LCA tools, methods like CML2001, EDIP97, and Eco-indicator 99 can be used for developing the environmental indicators (Dreyer et al., 2003). Generally, there would be eleven indicators that are suitable to be used for evaluating the environmental performance of the industrial systems, which can be classified into three main categories including resources, ecosystems, and human health (Van Hoof et al., 2013). As can be observed in Fig. 11.2, the indicators of depletion of abiotic resources, depletion of biotic resources, and land use represent the concerns of the resources; land use, ecotoxicity, eutrophication, acidification, and climate change stand for the concerns of the ecosystems; while the indicators of climate change, photo-oxidant formation, stratospheric ozone depletion, human toxicity, and carcinogenic substances belong to the category of human health. Notably, the indicators of land use and climate change can be used for representing different concerns according to the actual conditions of the investigated system.

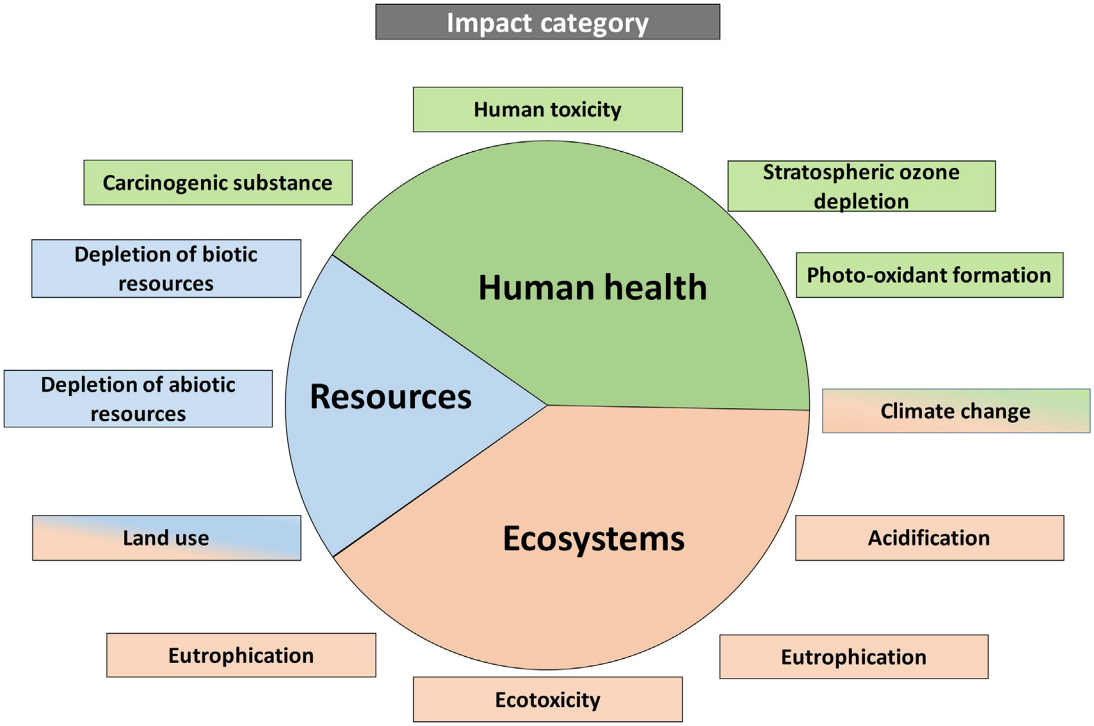

FIG. 11.2 Classification of the environmental assessment indicators in E-LCA. *Adapted from Van Hoof, G., Vieira, M., Gausman, M., Weisbrod, A., 2013. Indicator selection in life cycle assessment to enable decision making: issues and solutions. Int. J. Life Cycle Assess. 18(8), 1568-1580.*

As concluded in Table 11.1, the eleven indicators can be calculated by the E-LCA tools, i.e., CML2001, EDIP97, and Eco-indicator 99, while the basic definition of each indicator is also offered in Table 11.1. Although the three E-LCA methods can be used for developing the environmental indicators, differences can be found among them, i.e., CML2001 is an adapted version of an integrated multimedia model (USES) that focuses on evaluating the risk substances, while EDIP97 employs a simple modular fate model for the evaluation, which focuses on identifying the environmental key properties of chemicals; while between EDIP97 and Eco-indicator 99, the manners to assign the weight as well as to aggregate the index are different, which may cause different results in the assessment (Dreyer et al., 2003). Accordingly, choosing a proper E-LCA tool according to a certain research focus is an important step for creating a rational environmental index system.

Considering the E-LCA indicators are the most frequently adopted ones for evaluating the environmental performance of industrial systems, an example regarding the hydrogen production system is offered here for illustrating the procedures of E-LCA, as depicted in Fig. 11.3, where the research boundary should be defined first. Subsequently, the raw materials and energy that are consumed in the system within the boundary, and the outputs of the

TABLE 11.1 Environmental indicators from E-LCA (Bruijn et al., 2002).

Indicator (Unit)	Description	Method: CML 2001	Method: EDIP 97	Method: Eco-indicator 99
Depletion of abiotic resources (kg)	It represents the non-living resource consumption, like iron ore and crude oil	√		√
Depletion of biotic resources (kg)	It represents the consumption of biological resources, like rainforests and animal resources	√		
Land use (m^2)	It represents the land used by the system, which covers a range of consequences of human land use	√	√	√
Climate change (kg CO_2)	It represents the impact of human emissions on the earth's environment and atmosphere	√		√
Stratospheric ozone depletion (kg CFC-11)	It represents the thinning of the stratospheric ozone layer caused by anthropogenic emissions	√	√	√
Ecotoxicity (kg 1,4-DCB)	It represents the impacts of toxic substances on aquatic, terrestrial, and sediment ecosystems	√	√	√
Photo-oxidant formation (kg ethylene)	It represents the photochemical smog caused by the oxidation of some major atmospheric pollutants	√	√	
Eutrophication (kg PO_4)	It represents the excessive nutrient levels in the environment, especially nitrogen and phosphorus	√	√	√
Acidification (kg SO_2)	It represents the acidifying pollutants that have impacts on soil, water, biological organisms, ecosystems, and materials	√	√	√
Human toxicity (kg1,4-dichlorobenzene)	It represents the impact of toxic substances discharged into the environment, which have impacts on human health	√	√	
Carcinogenic substance (kg)	It represents the level of toxic substances that pose a fatal threat to the human body			√

system should be collected. After selecting the E-LCA methodology (like CML 2 baseline 2000 V2.04), the environmental index can be created according to the actual conditions of the investigated case, while for a hydrogen production system, the indicators of global warming, depletion of ozone layer, photochemical oxidant formation, acidification, human toxicity, non-biological resource consumption, eutrophication, human toxicity, and land use are frequently considered (Dufour et al., 2009; Jolliet et al., 2018). Based on the established index system, software like GaBi or SimaPro can be employed for calculating the data of each indicator of the hydrogen production system; while the E-LCA results can then be analyzed according to the software evaluation results.

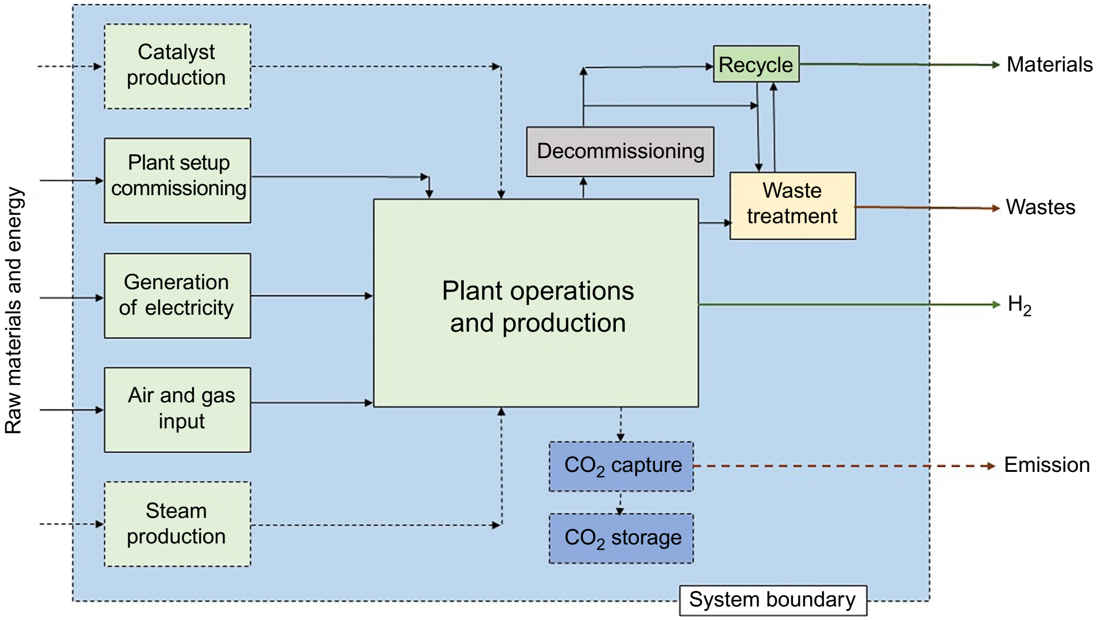

FIG. 11.3 Hydrogen production system framework. *Adapted from Dufour, J., Serrano, D., Galvez, J., Moreno, J., Garcia, C., 2009. Life cycle assessment of processes for hydrogen production. Environmental feasibility and reduction of greenhouse gases emissions. Int. J. Hydrog. Energy 34(3), 1370–1376*

11.2.3 Environmental indicators from the footprint assessment

For determining the environmental performance of the industrial systems from the life cycle perspective, calculation method environmental "footprints" can also be employed (Burman et al., 2018), where the different "footprints" measure the consumption of natural resources (Hoekstra, 2009) and describe the human activities that impact on sustainable development (UNEP, 2009). Considering that several footprint evaluation methods can be used in the life cycle sustainability assessment of industrial systems, corresponding indicators are offered here for representing the environmental performance, as depicted in Fig. 11.4, including ecological footprint, water footprint, carbon footprint, energy footprint, emission footprint, nitrogen footprint, land footprint, and biodiversity footprint (Alvarez et al., 2016). For more detailed information regarding footprint-based indicators, see Table 11.2.

Among the multiple footprints, the carbon footprint, water footprint, and ecological footprint are the most commonly used ones in the environmental assessment of industrial systems; which are correspondingly related to the hot issues of global warming, depletion of water resources, and ecosystem destruction. Although the other footprints are not as popular as the above-mentioned ones, they are still effective indicators in specific situations and can be employed for particular goals, such as addressing the concerns regarding the energy utilization, the emission reduction, and the land occupation, etc.

FIG. 11.4 Multiple "footprints" for life cycle-based environmental evaluation.

11.2.4 Other typical environmental indicators

As a research focus during the past decades, the environmental assessment of the industrial systems has been well-developed, offering a great number of indicators that can be adopted for creating a comprehensive environmental index system. In this sub-section, some other typical indicators are summarized. Consequently, users can select proper indicators according to their preferences and the actual conditions of the investigated system.

Except for the environmental indicators summarized in Table 11.3, some other criteria, such as the energy balance (consumptions and/or savings), use of materials (mass flows), (Ibáñez-Forés et al., 2014), the degree of clean production (Cobuloglu and Büyüktahtakın, 2015), etc., could also be used as alternative indicators for assessing the environmental performance of an industrial systems with life cycle thinking.

TABLE 11.2 Environmental indicators from the "footprint" (Čuček et al., 2012; Rees, 2016; Hoekstra, 2008; Chen and Lin, 2008; Sandholzer and Narodoslawsky, 2007; Leach et al., 2012; Wiedmann and Lenzen, 2007).

Indicator	Content	Description	Tool
Ecological footprint (m^2)	1. Arable land 2. Pasture land 3. Forest/woodland 4. Built-up land 5. Productive sea space 6. Forest land to absorb CO_2	It measures the amount of "biologically productive" land or water that enables sustainable development, including sub-indicators like the use of arable land, pasture land, forest/woodland, built-up land, productive sea space, and forest land to absorb CO_2	RegiOpt, Bottomline
Water footprint (L)	1. Blue water 2. Green water 3. Grey water	It refers to the total amount of fresh water used, consumed, or polluted, directly or indirectly. In which, blue is the consumption of surface and groundwater, green is the total consumption of rainwater resources, grey is the amount of water needed to be treated for satisfying the water quality	Mathematical programming tools
Carbon footprint (kg)	/	It represents the total amount of carbon dioxide and other greenhouse gases (GHGs) emitted during the entire life cycle of a process or product	Carbon footprint calculators
Energy footprint (J)	1. Renewable energy footprint (wind, solar, etc.) 2. Fossil or fossil energy footprint	It refers to the total energy consumption of the evaluated object in a certain period (except for the food consumption), which is used to indicate the energy dependence of a system, service, or product	Mathematical programming tools
Emission footprint (kg)	/	It is the total amount of emissions that a system or product releases into the air (SO_2, particles, CO, CO_2, etc.), water (COD, etc.), and soil (waste residue)	Mathematical programming tools
Nitrogen footprint (kg)	/	It refers to the total amount of nitrogen compounds emitted by the system, product, or human activities (all of the nitrogen species except for N_2)	Mathematical programming tools
Land footprint (m^2)	/	It refers to the actual land area needed to produce a product, or establish a system, or implement human activities	Mathematical programming tools
Biodiversity footprint	/	It represents the loss of biodiversity or excessive depletion of biological resources caused by a system or product	Mathematical programming tools

TABLE 11.3 Other typical indicators for the environmental assessment of industrial systems (Acar and Dincer, 2014; Kaya and Kahraman, 2010; García-Gusano et al., 2016; Aranda Usón et al., 2013).

Indicator	Description	Method
Energy efficiency	The ratio of energy output to total energy input that can be effectively utilized in the energy conversion process	$\eta = (m \times LHV)/E_{in}$ where m is the mass flow rate of the investigated system, LHV is its lower calorific value, and E_{in} is the energy input rate of the process
Exergy efficiency	It is also an efficiency, which is defined as useful output by consumed input, which is a measure of the thermodynamic perfection of the system	$\psi = (m \times ex^{ch})/Ex_{in}$ where m is the mass flow rate of the investigated system, ex^{ch} is its chemical exergy, and EX_{in} is the exergy input of the process
Noise	It refers to the environmental impacts regarding sound, which would be harmful to human health	The impact of noise on the environmental sustainability depends partly on the decibel level of the sound and partly on people's level of acceptance
Technical maturity	It refers to the state-of-the-art of the adopted technologies in the system	It is a subjective indicator, which relies on the experts' experience and judgment
Waste management	It refers to the activities and measures for reducing and managing the waste generated by the system	It includes the waste generation, collection, transportation, storage, and disposal, which should be determined by the life cycle inventory analysis

11.3 Life cycle economic indicators

In the triple-bottom-line-based sustainability assessment, the economic prosperity always plays a critical role for determining the overall sustainability of the industrial system, resulting in the development of economic indicators, such as the capital cost, production cost, operating and maintenance cost, feedstock cost, and replacement cost, etc. However, these indicators only focus on the costs or economic benefits of a sole stage (especially the manufacturing stage) of the industrial system, failing to measure the "cradle to grave" cost. Therefore, some useful economic assessment tools have been developed for evaluating the economic performance from the life cycle perspective, which can take into account the costs of designing, developing, running, and disposing of the industrial system.

11.3.1 Introduction of life cycle costing

Life cycle costing (LCC) is a methodology that can measure all costs related to an industrial system over its entire life cycle, which is preferred to evaluate the system that has a long lifetime and/or high maintenance, use, or disposal costs. Typically, the whole life cost of an industrial system could embrace the costs regarding purchase, installation, operating and maintenance, financing, and depreciation. Accordingly, the life cycle costs could be generically presented as: LCC = initial capital costs + lifetime operating costs + lifetime maintenance costs + rehabilitation costs + disposal costs − residual value (Li et al., 2017). To some degree, life cycle costing is similar to environmental-life cycle assessment, where the goal and scope (i.e., system boundaries) and other aspects need to be defined with the decisions conducted for the E-LCA in order to obtain an overall consistent analysis.

11.3.2 Economic indicator from the LCC

For conducting life cycle costing of an industrial system, several ways, such as analogy model, parametric model, and engineering cost model, can be employed, in which the costs for investment, operation and maintenance, and feedstock are usually accounted into the total cost. In addition, the recycling and disposing costs are also frequently integrated. Recently, the cost from the environmental impact (like carbon emission cost) has been suggested as worthy of consideration, as it is likely that emission and pollution of an industrial system will be charged in the near future. As can be observed in Table 11.4, some typical items are summarized, which could be selected and then combined for calculating the indicator of LCC. For fully understanding the economic indicator from the LCC, the detailed description regarding each categorized cost can be found in Table 11.4, while a well-established example regarding the LCC performance of the hydrogen production system is offered below (Fig. 11.5).

As can be observed in Fig. 11.5, multiple stages are involved in the water electrolysis-based hydrogen production system. Accordingly, several categorized costs should be accounted into the total cost, including the capital cost (C_C), operating and maintenance cost ($C_{O\&M}$), feedstock cost (C_F), replacement cost (C_R), salvage value (C_{SV}), and carbon emission cost (C_{CE}).

TABLE 11.4 Typical items for calculating the LCC of an industrial system (Ibáñez-Forés et al., 2014; Li et al., 2017; Wood and Hertwich, 2012; Ren et al., 2018).

Item (Unit)	Description
Capital cost ($, ¥, …)	It refers to the fixed one-time cost of purchasing land, buildings, construction, and equipment, which is the total cost of bringing the investigated system to a commercially viable state
Operating & maintenance cost ($, ¥, …)	It refers to all the costs required in the operation or sales of the investigated system; the costs of labor, laboratory services, utilities, administration, and all energy and material flows are also included
Feedstock cost ($, ¥, …)	It refers to the cost of raw materials required by the system to produce or start operation, which may be purchased, homemade, or commissioned external processing
Salvage value ($, ¥, …)	It refers to the residual value of the dismantled or cleaned fixed assets in the system, representing the part that can be reused or sold as useful materials, which can be obtained from the resale or recycling of equipment after the dismantling of an industrial system
Replacement value ($, ¥, …)	It refers to the amount of cash or cash equivalent required to pay for the same asset in accordance with the current market conditions. In practice, replacement cost is mostly used in the measurement of fixed assets
Emission cost ($, ¥, …)	It refers to the cost regarding the cost for environmental impacts of an industrial system, such as the carbon emissions, eutrophication effect, acidification effect, and winter smog effect. Among them, the cost of carbon emissions would be one of the most important factors that need to be integrated into the LCC

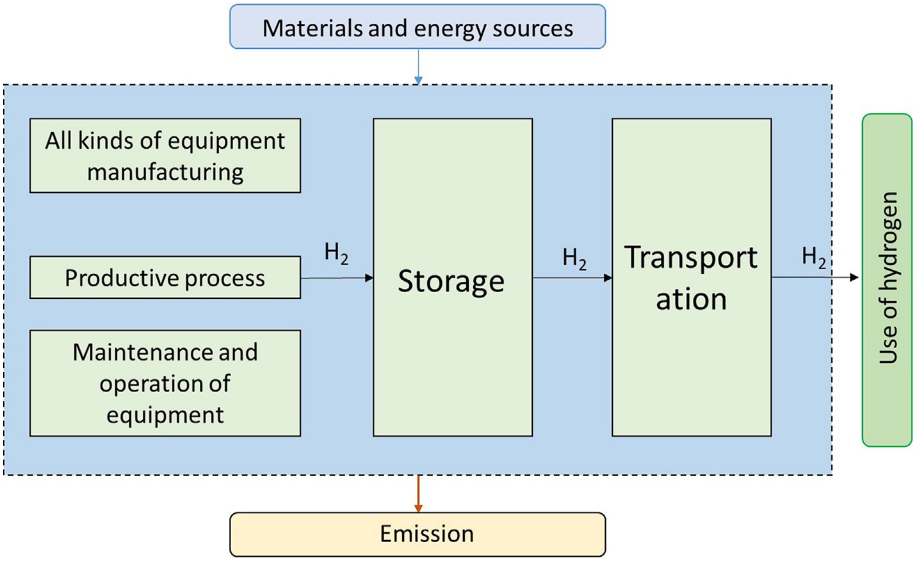

FIG. 11.5 An illustrative example of the electrolysis water-based hydrogen production system.

By referring to the work of Li et al. (2017), the indicator of LCC regarding the hydrogen production system can be mathematically determined as in Eq. (11.1).

$$LCC = C_C + C_{O\&M} + C_F + C_R + C_{CE} - C_{SV} \tag{11.1}$$

Since analyzing the cash flows at different times for an industrial system is quite important for the stakeholders (Zakeri and Syri, 2015), Eq. (11.1) is suggested to be transformed into Eq. (11.2) for better capturing the characteristic of cash flows of the investigated system.

$$LCC = C_C + \sum_{i=1}^{n} \frac{C_{O\&M,i}}{(1+r)^i} + \sum_{i=1}^{n} \frac{C_{F,i}}{(1+r)^i} + C_R + C_{CE} - \frac{C_{SV}}{(1+r)^n} \tag{11.2}$$

where n represents the life of the investigated system in years, and r is the discount rate, which is the interest rate used to determine the present value of future cash flow.

In one step forward, the decision-makers/stakeholders would adopt the payback period to evaluate the economic performance of the hydrogen production system, which represents the time required for the net income generated by the system to equal the initial investment. Based on the literature (Li et al., 2017), this indicator can also be employed as a life cycle-based economic criterion, which is determined by solving Eq. (11.3).

$$\sum_{i=1}^{k} \frac{C_{net-income} - C_{produc-\cos t}}{(1+r)^i} + C_{SV} - C_C - C_{CE} = 0 \tag{11.3}$$

11.3.3 Economic indicators from other economic assessment tools

Except for the traditional Life Cycle Costing for measuring the "cradle to grave" cost for an industrial system, some other methods can also be extended into the life cycle perspective for the economic evaluation. Here, two economic assessment tools, as well as their corresponding

TABLE 11.5 Economic indicators from economic parameters and cash flows (Ibáñez-Forés et al., 2014; López-Maldonado et al., 2011).

Method	Indicator (Unit)	Description
Economic parameters	Total annual cost ($, ¥, …)	It represents the annual cost of operating assets spent by the investigated system, which embraces the expense ratio, front-end load, back-end load, redemption fee, transaction costs, and opportunity costs of all those costs
	Net present value ($, ¥, …)	It is the difference between the present value of future cash inflows (income) and the present value of future cash outflows (expenditure), which accounts for the time value of money
	Economic potential	It refers to the potential of a region, country, or company in terms of economic development, growth, and creation of surplus value
Cash flows	Incoming cash ($, ¥, …)	It refers to the total capital income generated by the system, which may include revenue, the sales of a product or service, salvage income, and variable value income, current assets recovered, etc., while the implementation of a decision for reducing the costs also can be integrated
	Outgoing cash ($, ¥, …)	It refers to the total capital expenditure of the system, which may include the capital expenditures for the acquisition or construction of fixed assets, operating costs, production costs, administrative expenses, and sales expenses incurred in the operation, etc.

indicators, are introduced by referring to the work of Ibáñez-Forés et al. (2014). In this, the tool of economic parameters can be adopted for analyzing the cost-effectiveness of an industrial system, which typically embraces three indicators including the total annual cost, the net present value, and the economic potential; while the tool of cash flows can be utilized for measuring the economic balance of all the incoming and outgoing cash flows regarding the investment of an industrial system, where both the expenditure and profits should be taken into account. Detailed descriptions of the involved indicators can be found in Table 11.5.

11.3.4 Other typical economic indicators

Among the existing researches regarding life cycle sustainability assessment, some other economic indicators have been developed for representing the economic performance/impact/potential of an industrial system. In this sub-section, other typical economic indicators are summarized in Table 11.6. Consequently, users can select proper indicators according to their preferences and the actual conditions of the investigated system.

For measuring the economic performance of an industrial system with life cycle thinking, the LCC method is strongly suggested due to its high flexibility and easy operation, as well as its good connection with the environmental and social assessment (De Menna et al., 2018). However, in order to offer a more comprehensive economic evaluation result, the indicators from the tools of economic parameters and cash flows, as well as other typical criteria like economic benefit, and economic risk can also be selected according to the actual conditions of the investigated system.

TABLE 11.6 Other typical indicators for economic assessment of the industrial systems.

Indicator (Unit)	Description
Economic benefit	It measures the economic benefits contributed by starting the system, such as improving the GDP
Total cost ($, ¥, …)	It combines privately borne costs of a certain activity with those that are external to that activity, where all costs should be evaluated in a given base year (Bachmann, 2012)
Income ($, ¥, …)	It refers to money earned from the sale of products produced by the system, which is generally divided into gross income and net income
Taxes ($, ¥, …)	It refers to the compulsory financial expenses that the assessed system needs to pay in the process of production or operation
Economic risk	It can be described as the likelihood that an investment will be affected by macroeconomic conditions such as government regulation, exchange rates, or political stability

11.4 Life cycle social indicators

Compared to the life cycle-based environmental and economic assessments, life cycle-based social assessment targets only social and sociological impacts through a range of categories, in which, the social life cycle assessment (S-LCA) is usually employed to measure the impacts pertaining to the social concerns of an industrial system in its entire lifetime. However, the approach for measuring social performance is still in the development stage, while the corresponding indicators are typically unquantifiable, which rely heavily on experts' experiences and evaluations. In this sub-section, a mini review on some well-known social assessment methodologies developed so far is given, while the typical social indicators within these methods are introduced.

11.4.1 Introduction of the social life cycle assessment

Social life cycle assessment (S-LCA) is recently emerging as a useful approach in sustainability science, which can be employed for evaluating the social impacts of an industrial system. Among the available evaluation frameworks regarding the social concerns, the following five alternatives, including the GRI sustainability framework, UN millennium and sustainable development goals, SA 8000, ISO 26000, and UNEP and SETAC S-LCA guidelines would be selected to be incorporated into the life cycle sustainability assessment of the industrial systems (Kühnen and Hahn, 2017). In which, the GRI sustainability framework can be used for identifying the social sustainability-related information, which is suitable for offering information regarding an organization's positive or negative impacts on sustainable development; however, it fails to offer performance measurement to support decision-making.

UN millennium and sustainable development goals aim at offering a potential normative foundation and reference to indicate a positive contribution to sustainable development, but they may not be suitable for evaluating the contributions at organizational or product level. SA 8000 could offer a "cradle-to-gate" assessment regarding the social performance of a system, which focuses on the protection of human rights of employees by setting requirements for working conditions in internal and upstream supplier operations. ISO 26000 (IOS, 2017) covers seven main categories, including organizational governance, human rights, labor practices, the environment, fair operating practices, consumer issues, and community involvement and development, which can be utilized for better understanding social responsibility of organizations. Compared to the former four frameworks regarding the social-based assessment, the framework of UNEP and SETAC S-LCA guidelines is preferable to be adopted in the field of social life cycle assessment, which includes 31 social indicators related to five divergent stakeholder groups (i.e., workers, consumers, local community, society, and value chain actors). The reason for the popularity of the UNEP S-LCA can be attributed to the fact that S-LCA is similar to E-LCA, where the same procedures, i.e., definition of goal and scope of the study, inventory analysis, and impact assessment, need to be implemented (UNEP/SETAC, 2009; Wu et al., 2015; Papong et al., 2015).

By reviewing the literature, the most important social concerns/responsibilities could be classified into six main categories:

(1) concerning the safety and health of the workers (like injuries of the employees);
(2) concerning the safety and health of the local communities (like potential of accident risks);
(3) contributing to development of the society (like job creation);
(4) promoting social responsibility among the value chain actors (like working conditions within the whole value chain);
(5) concerning the safety and health of the consumers (like product safety); and
(6) other concerns (like stakeholder satisfaction) (Kühnen and Hahn, 2017).

Therefore, the indicators for evaluating the social performance of an industrial system should be selected rationally according to which stakeholder will be involved in a certain stage among the whole life span. For instance, workers and local communities would be more engaged into the stage of raw material extraction and treatment, while consumers and value chain actors would be highly involved in the stage of utilization of product.

11.4.2 Social Indicators from the UNEP S-LCA guidelines

Among the available social evaluation systems, the guidelines for S-LCA of product published by the UNEP is the most commonly practiced one, which embraces five categorized stakeholders including the worker, local community, society, consumer, and value chain actors (UNEP/SETAC, 2009), as given in Table 11.7. The reason for the classification is to support the identification of different stakeholders with divergent concerns, to classify multiple indicators within groups that have the same impacts, and to implement the corresponding assessment and interpretation. Here, Table 11.7 summarizes the stakeholder categories and the corresponding social indicators, where 26 criteria (among a total number of 31 indicators) are suggested in this chapter for representing the social concerns of the industrial systems.

For conducting the S-LCA, the procedures of the implementation of E-LCA can be referred, where the items of goal and scope definition, life cycle inventory analysis, life cycle impact assessment, and interpretation should also be conducted in an orderly manner.

TABLE 11.7 Major categories, indicators, and corresponding requirements regarding the UNEP S-LCA (Arcese et al., 2013; Aparcana and Salhofer, 2013; UNEP/SETAC, 2009; Mattioda et al., 2017).

Stakeholder	Indicator	Requirement
Workers (Those who work in the investigated system)	Child labor	The absence of children working in the system
	Fair salary	The salary should be no less than the minimum wage
	Working hours	The average number of working hour should be limited to 8h/day and 48h/week
	Forced labor	The abolition of forced labor
	Discrimination/ equal opportunity	The prevention of discrimination and the promotion of equal opportunities
	Health and safety	The guarantee of works' health and safety
	Social benefits/ social security	The suggestion of more than two social benefits provided by the organization
Consumers (Those who use the system's products or are affected by the system)	Health and safety	The guarantee of consumers' health and safety.
	Consumer privacy	The protection of consumers' right to privacy
	Feedback mechanism	The presence of consumers' feedback mechanism
	End of life responsibility	Information on end-of-life options or recalls policy for consumers
Local community (Those who live in the area where the system locates in)	Local employment	The minimum percentage of local labor should no less than 50%
	Access to material resources	The sustainable utilization of nature resources and the recycling of used material
	Access to immaterial resources	The promoting of community service
	Delocalization and Migration	The absence of forced resettlement caused by the system
	Safe & healthy living conditions	The guarantee of safe and healthy surrounding communities
	Respect of indigenous rights	The protection of indigenous rights
	Community engagement	The consideration of the environment, health, or welfare of a community

Continued

TABLE 11.7 Major categories, indicators, and corresponding requirements regarding the UNEP S-LCA (Arcese et al., 2013; Aparcana and Salhofer, 2013; UNEP/SETAC, 2009; Mattioda et al., 2017)—cont'd

Stakeholder	Indicator	Requirement
Society (Both the national and global society affected by the system)	Contribution to economic development	The promotion of economic contribution to the society
	Public commitments to sustainability issues	The promise or agreement related to the sustainable development of the system
	Technology development	The development of efficient and environmentally friendly technologies
	Corruption	The prevention of corruption of the system
Value chain actors (Those who are involved in the sale/production of products, or in the operation of the system, except for the consumers)	Fair competition	The grantee of fair competition and the prevention of antitrust legislation or monopoly practices
	Promoting social responsibility	The improvement of social responsibility contributed by the whole value chain of the investigated system
	Supplier relationships	The cooperation between the supplies and the investigated system should be facilitated stably
	Respect of intellectual property rights	The protection of the intellectual property rights by all the involved actors within the value chain

However, the S-LCA focuses on the evaluation of social impacts rather than the environmental performance, and it collects additional information on organization-related aspects with the life cycle perspective. Therefore, the utilization of the S-LCA should be implemented according to the specific characteristics and necessities of the system, by considering its physical, environmental, social, and economic limitations (Mattioda et al., 2017).

11.4.3 Other typical social indicators

With an increasing tendency in considering social concerns/responsibilities into the sustainability assessment of the industrial systems, some other social indicators such as social acceptability and social benefit have also been suggested frequently for being employed in the life cycle-based assessment system. Besides, by viewing the policy-maker as another kind of categorized stakeholder, some criteria from the political aspect, like government support and political applicability, also can be deemed as social-political indicators. Some typical social (or social-political) indicators are briefly introduced in Table 11.8.

TABLE 11.8 Other typical indicators for social assessment of the industrial systems (Ren et al., 2016; Xu et al., 2018b).

Indicator	Description
Social acceptability	It represents the degree of the acceptability of the investigated system, which is characterized as an indicator of individual opinions
Social benefit	It is a generic indicator that can be defined specifically according to a certain concern with respect to the investigated system, for instance, the benefit of job creation
Inherent safety index	It measures the inherent hazard of the investigated system, which could be extended into the life cycle perspective
Government support	It refers to the government support regarding establishing and/or running a certain industrial system by setting corresponding policies

From the above-mentioned frameworks and indicators, the evaluation of social concerns regarding the industrial systems relies heavily on people's subjective judgements according to the specific characteristics and necessities of the investigated system, failing to offer a generic social index that embraces both quantifiable and unquantifiable indicators. Therefore, more efforts should be put into the development of life cycle-based social indicators for improving the usability of the S-LCA tools.

11.5 A composite life cycle sustainability index

As stated before, dozens of criteria from the environmental, economic, and social dimensions can be selected for assessing the industrial systems, resulting in difficulty aggregating multiple criteria for representing the overall sustainability. Therefore, this subsection aims at developing a novel composite life cycle sustainability index that can integrate multiple criteria from the triple-bottom-line concerns for the prioritization of the industrial systems. In this subsection, a composite index is proposed, based on the work of Xu et al. (2017, 2018b). Subsequently, a case of the prioritization of five low-carbon ammonia production systems by using the composite index is studied. Finally, sensitivity analysis is conducted for demonstrating the feasibility of the composite life cycle sustainability index.

11.5.1 Development of a composite life cycle sustainability index

Because of the absence of general standardized indicators, the overall life cycle sustainability of the industrial systems is always hard to measure, especially when dozens of environmental, economic, and social indicators could be selected. Therefore, this subsection focuses on the development of a composite life cycle sustainability index for the prioritization of industrial systems.

Since life cycle-based sustainability can be presented by the TBL-based three-dimension (3D) cube, as shown in Fig. 11.6 (Moradi-Aliabadi and Huang, 2016), the overall sustainability of an industrial system has recently been depicted by a vector function. In the 3D cube, x, y, and z, respectively, stand for the environmental, economic, and social pillars, α, β, and γ are

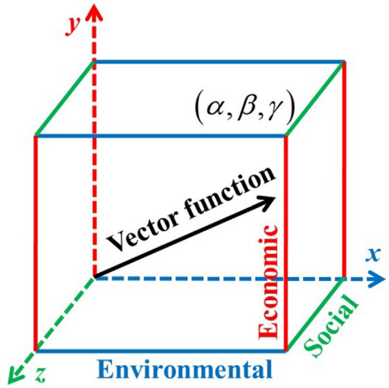

FIG. 11.6 The triple-bottom-line 3D sustainability cube.

correspondingly the weights of the three pillars for indicating their relative importance in the overall sustainability, and a 3D vector $\vec{S}_i = \langle \hat{x}_i, \hat{y}_i, \hat{z}_i \rangle$ could be used for denoting the sustainability of the ith system (i=1, 2, …) (Eq. 11.4).

$$\vec{S}_i = \langle \hat{x}_i, \hat{y}_i, \hat{z}_i \rangle = \langle \alpha En_i, \beta Ec_i, \gamma So_i \rangle = \alpha En_i \hat{x} + \beta Ec_i \hat{y} + \gamma So_i \hat{z} \tag{11.4}$$

In Eq. (11.4), En_i, Ec_i, and So_i are the quantified composite performances of the ith system with respect to the three pillars, which could be calculated by using Eq. (11.5) (Moradi-Aliabadi and Huang, 2016; Xu et al., 2017); while the pillars' weights (α, β and γ) could be determined by using the subjective weighting method (such as AHP and BWM), the objective weighting method (such as the entropy and CRITIC), or the combined ones.

$$\begin{cases} En_i = \sum_{k=1}^{u} a_k en_{k-i} \\ Ec_i = \sum_{k=1}^{v} b_k ec_{k-i} \\ So_i = \sum_{k=1}^{w} c_k so_{k-i} \end{cases} \tag{11.5}$$

where a_k, b_k, and c_k represent the local weights assigned to the kth indicator in the corresponding pillar, en_{k-i}, ec_{k-i}, and so_{k-i} are the normalized data of the ith alternative regarding to the kth criterion, and u, v, and w represent the indicators number in each pillar. Noting that there are several weighting and normalization methods that could be utilized for supporting the development of the composite life cycle sustainability index, the users can select proper ones for assigning the weights and processing the data according to the actual conditions of the investigated systems. For more detailed information regarding the weighting method and the normalization technique, the reader is referred to the literature (Xu et al., 2018b).

Based on our previous works (Xu et al., 2017, 2018b), two referring 3D sustainability cubes, i.e., the ideal and nadir cubes (which, respectively, represent the highest and the lowest sustainability that an alternative system can ideally achieve), are used for supporting the development of a composite life cycle sustainability index. As depicted in Fig. 11.7, the left cube

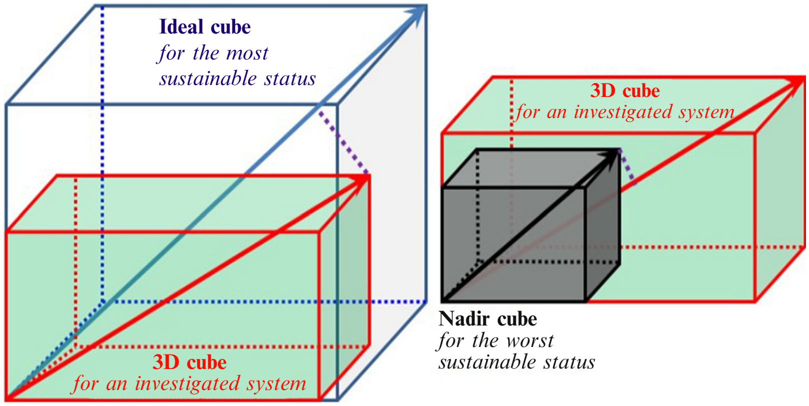

FIG. 11.7 The ideal and nadir 3D sustainability cubes.

implies the most sustainable status of an industrial system, which is denoted as $\vec{S}^* = \langle \hat{x}^*, \hat{y}^*, \hat{z}^* \rangle$, while the right one indicates the worst status of the system, which is denoted as $\vec{S}^- = \langle \hat{x}^-, \hat{y}^-, \hat{z}^- \rangle$. Herein, the vector formats with respect to both the ideal and nadir cubes could be employed for representing the highest and the lowest sustainability status, as given in Eqs. (11.6), (11.7), respectively, by modifying the work of Xu et al. (2017, 2018b).

$$\vec{S}^* = \langle \hat{x}^*, \hat{y}^*, \hat{z}^* \rangle = \langle \alpha En^*, \beta Ec^*, \gamma So^* \rangle = \alpha En^* \hat{x} + \beta Ec^* \hat{y} + \gamma Ec^* \hat{z} \tag{11.6}$$

$$\vec{S}^- = \langle \hat{x}^-, \hat{y}^-, \hat{z}^- \rangle = \langle \alpha En^-, \beta Ec^-, \gamma So^- \rangle = \alpha En^- \hat{x} + \beta Ec^- \hat{y} + \gamma So^- \hat{z} \tag{11.7}$$

Similarly, Eq. (11.8) and Eq. (11.9), respectively, should be employed for obtaining the categorized composite performances of the ideal system and the nadir one.

$$\begin{cases} En^* = \sum_{k=1}^{u} a_k \max_{i=1,2,\cdots} en_{k-i} \\ Ec^* = \sum_{k=1}^{v} b_k \max_{i=1,2,\cdots} ec_{k-i} \\ So^* = \sum_{k=1}^{w} c_k \max_{i=1,2,\cdots} so_{k-i} \end{cases} \tag{11.8}$$

$$\begin{cases} En^- = \sum_{k=1}^{u} a_k \min_{i=1,2,\cdots} en_{k-i} \\ Ec^- = \sum_{k=1}^{v} b_k \min_{i=1,2,\cdots} ec_{k-i} \\ So^- = \sum_{k=1}^{w} c_k \min_{i=1,2,\cdots} so_{k-i} \end{cases} \tag{11.9}$$

Based on the characteristics of the vector function, the sustainability of an industrial system can be judged by considering two parameters, i.e., the magnitude of the vector (Eqs. 11.10–11.12) for measuring the absolute sustainability score, and the cosine angle of the vector from the ideal (or nadir) one (Eqs. 11.13, 11.14) for quantifying the relative sustainability balance. Apparently, a large value in the vector's magnitude stands for a good sustainability performance in an absolute way; while a large value in cos^* and a small value in cos^- is preferable from the viewpoint of relative balance.

$$\left\|\vec{S}_i\right\| = \sqrt{\alpha^2 En_i^2 + \beta^2 Ec_i^2 + \gamma^2 So_i^2} \tag{11.10}$$

$$\left\|\vec{S}^*\right\| = \sqrt{(\alpha En^*)^2 + (\beta Ec^*)^2 + (\gamma So^*)^2} \tag{11.11}$$

$$\left\|\vec{S}^-\right\| = \sqrt{(\alpha En^-)^2 + (\beta Ec^-)^2 + (\gamma So^-)^2} \tag{11.12}$$

$$\cos^* = \cos\left(\vec{S}_i, \vec{S}^*\right) = \left(\frac{\vec{S}_i \cdot \vec{S}^*}{\left\|\vec{S}_i\right\|\left\|\vec{S}^*\right\|}\right) = \frac{\alpha^2 En_i En^* + \beta^2 Ec_i Ec^* + \gamma^2 So_i So^*}{\sqrt{\alpha^2 En_i^2 + \beta^2 Ec_i^2 + \gamma^2 So_i^2} \times \sqrt{(\alpha En^*)^2 + (\beta Ec^*)^2 + (\gamma So^*)^2}} \tag{11.13}$$

$$\cos^- = \cos\left(\vec{S}_i, \vec{S}^-\right) = \left(\frac{\vec{S}_i \cdot \vec{S}^-}{\left\|\vec{S}_i\right\|\left\|\vec{S}^-\right\|}\right) = \frac{\alpha^2 En_i En^- + \beta^2 Ec_i Ec^- + \gamma^2 So_i So^-}{\sqrt{\alpha^2 En_i^2 + \beta^2 Ec_i^2 + \gamma^2 So_i^2} \times \sqrt{(\alpha En^-)^2 + (\beta Ec^-)^2 + (\gamma So^-)^2}} \tag{11.14}$$

In order to use a comprehensive way to measure the overall sustainability, the absolute and relative sustainability performances of the industrial system should be integrated by employing the vector projection function, as given in Eqs. (11.15), (11.16), where Pr_i^* is the projection of the ith alternative on the ideal system, while Pr_i^- is the projection of the nadir system on the ith alternative.

$$\mathrm{Pr}_i^* = \mathrm{Pr}\left(\vec{S}_i, \vec{S}^*\right) = \left\|\vec{S}_i\right\| \cos^* = \frac{\alpha^2 En_i En^* + \beta^2 Ec_i Ec^* + \gamma^2 So_i So^*}{\sqrt{(\alpha En^*)^2 + (\beta Ec^*)^2 + (\gamma So^*)^2}} \tag{11.15}$$

$$\mathrm{Pr}_i^- = \mathrm{Pr}\left(\vec{S}^-, \vec{S}_i\right) = \left\|\vec{S}^-\right\| \cos^- = \frac{\alpha^2 En_i En^- + \beta^2 Ec_i Ec^- + \gamma^2 So_i So^-}{\sqrt{\alpha^2 En_i^2 + \beta^2 Ec_i^2 + \gamma^2 So_i^2}} \tag{11.16}$$

Subsequently, the two projection values should be normalized into a uniform distribution for making better comparisons (Xu et al., 2017, 2018b), where Eq. (11.17) reflects the normalized similarity regarding the vector-pairs of $\vec{S}_i \sim \vec{S}^*$, while Eq. (11.18) shows that of $\vec{S}_i \sim \vec{S}^-$, and both of them with uniform distribution [0, 1].

$$NPr_i^* = \frac{\Pr_i^*}{\left\|\vec{S}^*\right\|} = \frac{\alpha^2 En_i En^* + \beta^2 Ec_i Ec^* + \gamma^2 So_i So^*}{\sqrt{(\alpha En^*)^2 + (\beta Ec^*)^2 + (\gamma So^*)^2}} \Bigg/ \sqrt{(\alpha En^*)^2 + (\beta Ec^*)^2 + (\gamma So^*)^2}$$
$$= \frac{\alpha^2 En_i En^* + \beta^2 Ec_i Ec^* + \gamma^2 So_i So^*}{(\alpha En^*)^2 + (\beta Ec^*)^2 + (\gamma So^*)^2} \tag{11.17}$$

$$NPr_i^- = \frac{\Pr_i^-}{\left\|\vec{S_i}\right\|} = \frac{\alpha^2 En_i En^- + \beta^2 Ec_i Ec^- + \gamma^2 So_i So^-}{\sqrt{\alpha^2 En_i^2 + \beta^2 Ec_i^2 + \gamma^2 So_i^2}} \Bigg/ \sqrt{\alpha^2 En_i^2 + \beta^2 Ec_i^2 + \gamma^2 So_i^2}$$
$$= \frac{\alpha^2 En_i En^- + \beta^2 Ec_i Ec^- + \gamma^2 So_i So^-}{(\alpha En_i)^2 + (\beta Ec_i)^2 + (\gamma So_i)^2} \tag{11.18}$$

Apparently, a real sustainable system should simultaneously have higher similarity degree with the ideal vector but lower similarity degree with the nadir one, resulting in the development of a composite life cycle sustainability index (*CI*) as given in Eq. (11.19) (Xu et al., 2018b).

$$CI_i = \frac{NPr_i^*}{NPr_i^* + NPr_i^-} = \frac{\dfrac{\alpha^2 En_i En^* + \beta^2 Ec_i Ec^* + \gamma^2 So_i So^*}{(\alpha En^*)^2 + (\beta Ec^*)^2 + (\gamma So^*)^2}}{\dfrac{\alpha^2 En_i En^* + \beta^2 Ec_i Ec^* + \gamma^2 So_i So^*}{(\alpha En^*)^2 + (\beta Ec^*)^2 + (\gamma So^*)^2} + \dfrac{\alpha^2 En_i En^- + \beta^2 Ec_i Ec^- + \gamma^2 So_i So^-}{(\alpha En_i)^2 + (\beta Ec_i)^2 + (\gamma So_i)^2}} \tag{11.19}$$

Here, *CI* is the composite life cycle sustainability index that can integrate the triple-bottom-line concerns, while multiple life cycle-based criteria can be used in the equation according to the actual conditions of the investigated systems and the preference of the decision-makers. Different from the previous sustainability prioritization frameworks, which rely on the multi-criteria decision making methods for ranking the alternative industrial systems, the proposed index (*CI*) can directly prioritize the sustainability sequence of the alternatives by aggregating the TBL-based concerns from the life cycle perspective.

11.5.2 Case study of the composite life cycle sustainability index

In order to demonstrate the developed composite index for supporting the life cycle sustainability prioritization of industrial systems, five low-carbon ammonia production routes proposed by Xu et al. (2018b) have been adapted here for the case study; namely, wind power-based electrolysis (A_1), solar power-based electrolysis (A_2), hydropower-based electrolysis (A_3), biomass gasification-based electrolysis (A_4), and nuclear power-based electrolysis (A_5). For more detailed information regarding the five alternative systems, the reader is referred to the literature (Xu et al., 2018b).

From the life cycle perspective, human toxicity, global warming, and abiotic depletion are selected as the environmental indicators (en_{1-3}), the life cycle costs, market potential, and economic contribution are identified as the economic indicators (ec_{1-3}), while the inherent safety, social acceptance, and policy applicability are taken as the social indicators (so_{1-3}). For developing these life cycle-based indicators, the life cycle assessment tool CML 2001 should be employed for the three environmental indicators, the life cycle costing is suggested to collect the indicator of ec_1, the inherent safety proposed by Heikkilä (1999) should be extended into

TABLE 11.9 Decision matrix for implementing the composite life cycle sustainability index (Xu et al., 2018b).

	en_1	en_2	en_3	ec_1	ec_2	ec_3	so_1	so_2	so_3
A_1	0.052	0.257	0.219	0.151	0.231	0.273	0.262	0.267	0.247
A_2	0.049	0.140	0.122	0.110	0.279	0.140	0.262	0.267	0.211
A_3	0.326	0.317	0.265	0.139	0.165	0.479	0.262	0.234	0.289
A_4	0.529	0.142	0.274	0.374	0.173	0.019	0.127	0.149	0.126
A_5	0.045	0.144	0.120	0.225	0.151	0.090	0.086	0.084	0.126
Local weight	0.333	0.333	0.333	0.333	0.333	0.333	0.333	0.333	0.333

the same life cycle span to develop the indicator of so_1, while the other four indicators, i.e., ec_2, ec_3, so_2, and so_3, are developed by using life cycle thinking, given the subjective nature of these indicators. Based on the work of Xu et al. (2018b), the data of the alternative's performance regarding each indicator of the low-carbon ammonia production systems can be created as shown in Table 11.9, denoted as a decision matrix. It is worth pointing out that the data shown in Table 11.9 have already been normalized, while the weights were equally assigned to the categorized indicators in the corresponding dimension for simply illustrating how to use the composite life cycle sustainability index for the prioritization. For more detailed information regarding the data normalization, the work of Xu et al. (2018b) can be referred to.

Based on the decision matrix, the quantified composite performances of a system can be obtained, taking the alternative system A_1, the ideal system, and the nadir one as examples; they are calculated as below:

$$\begin{cases} En_1 = 0.333(en_{1-1} + en_{2-1} + en_{3-1}) = 0.333(0.052 + 0.257 + 0.219) = 0.176 \\ Ec_1 = 0.333(ec_{1-1} + ec_{2-1} + ec_{3-1}) = 0.333(0.151 + 0.231 + 0.273) = 0.218 \\ So_1 = 0.333(so_{1-1} + so_{2-1} + so_{3-1}) = 0.333(0.262 + 0.267 + 0.247) = 0.259 \end{cases}$$

$$\begin{cases} En^* = 0.333\left(\max\limits_{i=1,2,\cdots,5} en_{1-i} + \max\limits_{i=1,2,\cdots,5} en_{1-i} + \max\limits_{i=1,2,\cdots,5} en_{1-i}\right) = 0.333(0.529 + 0.317 + 0.274) = 0.373 \\ Ec^* = 0.333\left(\max\limits_{i=1,2,\cdots,5} ec_{1-i} + \max\limits_{i=1,2,\cdots,5} ec_{1-i} + \max\limits_{i=1,2,\cdots,5} ec_{1-i}\right) = 0.333(0.374 + 0.279 + 0.479) = 0.377 \\ So^* = 0.333\left(\max\limits_{i=1,2,\cdots,5} so_{1-i} + \max\limits_{i=1,2,\cdots,5} so_{1-i} + \max\limits_{i=1,2,\cdots,5} so_{1-i}\right) = 0.333(0.262 + 0.267 + 0.289) = 0.273 \end{cases}$$

$$\begin{cases} En^- = 0.333\left(\min\limits_{i=1,2,\cdots,5} en_{1-i} + \min\limits_{i=1,2,\cdots,5} en_{1-i} + \min\limits_{i=1,2,\cdots,5} en_{1-i}\right) = 0.333(0.045 + 0.140 + 0.120) = 0.102 \\ Ec^- = 0.333\left(\min\limits_{i=1,2,\cdots,5} ec_{1-i} + \min\limits_{i=1,2,\cdots,5} ec_{1-i} + \min\limits_{i=1,2,\cdots,5} ec_{1-i}\right) = 0.333(0.110 + 0.151 + 0.019) = 0.093 \\ So^- = 0.333\left(\min\limits_{i=1,2,\cdots,5} so_{1-i} + \min\limits_{i=1,2,\cdots,5} so_{1-i} + \min\limits_{i=1,2,\cdots,5} so_{1-i}\right) = 0.333(0.086 + 0.084 + 0.126) = 0.099 \end{cases}$$

TABLE 11.10 Quantified composite performances of each system with respect to the three pillars.

	A_1	A_2	A_3	A_4	A_5	Ideal	Nadir	Weight
En	0.176	0.104	0.303	0.315	0.103	0.373	0.102	0.333
Ec	0.218	0.176	0.261	0.189	0.155	0.377	0.093	0.333
So	0.259	0.247	0.262	0.134	0.099	0.273	0.099	0.333

Similarly, the quantified composite performances of each system with respect to the three pillars can be determined as summarized in Table 11.10.

Based on the data in Table 11.10, the composite life cycle sustainability index in Eq. (11.19) should be employed for ranking the five alternative systems; taking the system of A_1 as an example, the value of CI_1 was calculated as follows:

$$CI_1 = \left[\frac{\alpha^2 En_1 En^* + \beta^2 Ec_1 Ec^* + \gamma^2 So_1 So^*}{(\alpha En^*)^2 + (\beta Ec^*)^2 + (\gamma So^*)^2}\right] \Big/ \left[\frac{\alpha^2 En_1 En^* + \beta^2 Ec_1 Ec^* + \gamma^2 So_1 So^*}{(\alpha En^*)^2 + (\beta Ec^*)^2 + (\gamma So^*)^2}\right.$$

$$\left. + \frac{\alpha^2 En_1 En^- + \beta^2 Ec_1 Ec^- + \gamma^2 So_1 So^-}{(\alpha En_1)^2 + (\beta Ec_1)^2 + (\gamma So_1)^2}\right]$$

$$= \left[\frac{0.333^2(0.176 \times 0.373 + 0.218 \times 0.377 + 0.259 \times 0.333)}{0.333^2(0.373^2 + 0.377^2 + 0.333^2)}\right]$$

$$\div \quad \left[\frac{0.333^2(0.176 \times 0.373 + 0.218 \times 0.377 + 0.259 \times 0.333)}{0.333^2(0.373^2 + 0.377^2 + 0.333^2)}\right.$$

$$\left. + \frac{0.333^2(0.176 \times 0.102 + 0.218 \times 0.093 + 0.259 \times 0.099)}{0.333^2(0.176^2 + 0.218^2 + 0.259^2)}\right]$$

$$= 0.614/[0.614 + 0.438] = 0.583$$

Similarly, the composite life cycle sustainability index, with respect to each alternative system, can be obtained by running Eq. (11.19). The obtained results are depicted in Fig. 11.8, demonstrating that the overall sustainability sequence regarding the five low-carbon ammonia production routes is $A_3 > A_4 > A_1 > A_2 > A_5$.

From the case study, it can be concluded that by proposing the composite life cycle sustainability index, all the criteria from the environmental, economic, and social concerns could be aggregated into a composite index, which is characterized by integrating the absolute score and relative balance of the multi-criteria in a compromise way for offering a rigorous ranking result. Different from the existing works that employ the multi-criteria decision making approaches to rank the alternative industrial systems, the proposed index (*CI*) can simplify the ranking procedures by using a single yet reliable equation, as given in Eq. (11.19).

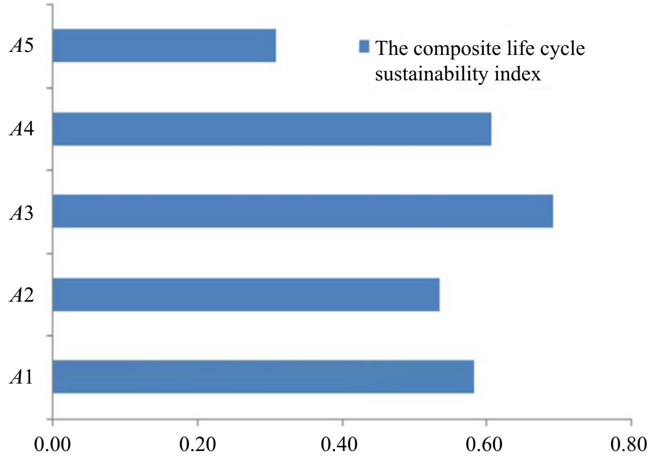

FIG. 11.8 The composite life cycle sustainability index for each low-carbon ammonia production system.

11.5.3 Sensitivity analysis of the composite life cycle sustainability index

It is worth pointing out that the local weight with respect to each criterion, and the relative importance regarding each pillar for implementing the composite life cycle sustainability index, should be determined according to the actual conditions of the investigated systems and the preferences of the decision-makers. Accordingly, the ranking result derived from the composite sustainability index would be influenced by the weighting information fed into it. Therefore, for testing the effects of the weights on the composite index for the prioritization, the following scenarios were studied by changing the weights of the involved criteria, as well as the relative importance of the three pillars.

Case en_k: a dominant local weight (0.60) was assigned to the kth criterion in the environmental pillar, and an equal weight of 0.20 was assigned to the other two criteria in the same pillar; in addition, an equal local weight of 0.333 was given to the criteria in the economic and social pillars. As for the relative importance of the three pillars, $\alpha=0.4545$, $\beta=\gamma=0.2727$ was used. Therefore, the global weights with respect to the eight non-dominant criteria can be set as the same value, equaling 0.091.
Case ec_k: a dominant local weight (0.60) was assigned to the kth criterion in the economic pillar, and an equal weight of 0.20 was assigned to the other two criteria in the same pillar; similarly, an equal local weight of 0.333 was given to the criteria in the environmental and social pillars; and $\beta=0.4545$, $\alpha=\gamma=0.2727$was used.
Case so_k: a dominant local weight (0.60) was assigned to the kth criterion in the social pillar, and an equal weight of 0.20 was assigned to the other two criteria in the same pillar; similarly, an equal local weight of 0.333 was given to the criteria in the environmental and economic pillars; and $\gamma=0.4545$, $\alpha=\beta=0.2727$was used.

By running the same equation of the composite life cycle sustainability index (Eq. 11.19), the rankings with respect to the above-mentioned cases can be determined, as depicted in Fig. 11.9; in which, the system A_3 remains the best choice for almost all the cases, except for the weights-change in case en_1 and case ec_1, while A_5 is always the most undesirable

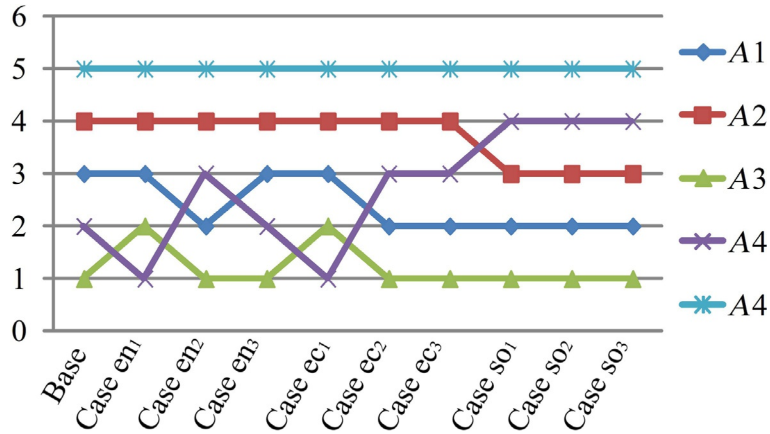

FIG. 11.9 The result of sensitivity analysis of the composite life cycle sustainability index.

option, indicating that the developed composite index is effective to identify the most sustainable industrial system as well as the worst one among various alternatives. However, the sequences with respect to the other alternatives are sensitive to the weights changing, implying that assigning the weights to the criteria and pillars accurately is a critical action for offering a reliable prioritization result. Noting that this work aims at proposing a generic composite index for life cycle sustainability prioritization of industrial systems without the consideration of specific weighting method, the users can select the subjective, the objective, or the combined methods for the determination of weights according to the actual conditions of the investigated systems.

11.6 Conclusions

For boarding the scope of LCSA regarding the industrial systems, the life cycle-based triple bottom line should be employed as an accounting framework, which covers the divergent concerns with respect to the environmental, economic, and social performance with a life cycle perspective. Till now, the three life cycle assessment tools, i.e., E-LCA, LCC, and S-LCA, are still preferred by users for creating a comprehensive assessment system. However, it can be noted that the economic performances are limited to a few indicators that derived from LCC, while the social concerns rely heavily on subjective judgements by using traditional S-LCA. To be specific, the life cycle costs (from the LCC) cannot provide a comprehensive evaluation system of economic sustainability; while the application of the social indicators (from the S-LCA) has not been investigated sufficiently, where the limitations of data acquisition, quantification, and the subjective nature of these indicators need to be addressed. Therefore, in addition to the three popular life cycle assessment tools, some other promising assessment approaches along with the indicators, that could be adopted or adapted into the life cycle sustainability assessment of the industrial systems, were also summarized in this work; in which, the assessment frameworks of multiple "footprints," the economic parameters and cash flows, as well as the guidelines like SA 8000 and ISO 26000 could, respectively, act as substitutes for developing the environmental, economic, and social indicators. In addition, the most frequently employed criteria, like energy and exergy efficiencies for the environmental impacts, economic benefit and risk for the economic prosperity, as well as social acceptability and benefit for the social concerns were suggested to be considered for offering a well-rounded assessment system.

Dozens of criteria that relate to the environmental-economic-social concerns could be used for representing the sustainability of industrial systems, resulting in difficulty aggregating multiple criteria for the prioritization. Therefore, a composite life cycle sustainability index was proposed in this chapter by referring to the work of Xu et al. (2018b), where the life cycle environmental, economic, and social criteria can be integrated for measuring the overall sustainability of industrial systems by employing a vector-projection theory. The developed composite index is characterized by combining the absolute performances and relative balance of the multi-criteria in a compromise way for ranking alternative systems, which is favored by the nature of the sustainability. In addition, an industrial case regarding five low-carbon ammonia production systems was investigated by the composite life cycle sustainability index, while the results of the case study and the corresponding sensitivity analysis reveal that the developed composite index is feasible and valid for ranking industrial systems.

In general, a composite life cycle sustainability index plays a significant role in the sustainability prioritization of industrial systems, while creating a comprehensive and rational indicator system is not an easy task, especially when dozens of environmental, economic, and social criteria could be selected. Because of the absence of general standardized indicators, the overall life cycle sustainability of the industrial systems is always hard to measure. Therefore, further studies should focus on the selection and integration of the most important indicators for laying a strong foundation for the establishment of the final composite index of life cycle sustainability assessment.

Acknowledgments

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References

Acar, C., Dincer, I., 2014. Comparative assessment of hydrogen production methods from renewable and non-renewable sources. Int. J. Hydrog. Energy 39 (1), 1–12.

Alvarez, S., Carballo-Penela, A., Mateo-Mantecón, I., Rubio, A., 2016. Strengths-weaknesses-opportunities-threats analysis of carbon footprint indicator and derived recommendations. J. Clean. Prod. 121, 238–247.

Aparcana, S., Salhofer, S., 2013. Application of a methodology for the social life cycle assessment of recycling systems in low income countries: three Peruvian case studies. Int. J. Life Cycle Assess. 18 (5), 1116–1128.

Aranda Usón, A., López-Sabirón, A.M., Ferreira, G., Llera Sastresa, E., 2013. Uses of alternative fuels and raw materials in the cement industry as sustainable waste management options. Renew. Sust. Energ. Rev. 23, 242–260.

Arcese, G., Lucchetti, M., Merli, R., 2013. Social life cycle assessment as a management tool: methodology for application in tourism. Sustainability 5 (8), 3275–3287.

Bachmann, T.M., 2012. Towards life cycle sustainability assessment: drawing on the NEEDS project's total cost and multi-criteria decision analysis ranking methods. Int. J. Life Cycle Assess. 18 (9), 1698–1709.

Bruijn, H., van Duin, R., Huijbregts, M., 2002. Handbook on Life Cycle Assessment: Operational Guide to the ISO Standards. Kluwer Academic Publishers, Dordrecht.

Burman, N.W., Croft, J., Engelbrecht, S., Ladenika, A.O., MacGregor, O.S., Maepa, M., Bodunrin, M.O., Harding, K.G., 2018. Review: life-cycle assessment, water footprinting, and carbon footprinting in Portugal. Int. J. Life Cycle Assess. 23 (8), 1693–1700.

Chen, C.-Z., Lin, Z.-S., 2008. Multiple timescale analysis and factor analysis of energy ecological footprint growth in China 1953–2006. Energy Policy 36 (5), 1666–1678.

Cobuloglu, H.I., Büyüktahtakın, İ.E., 2015. A stochastic multi-criteria decision analysis for sustainable biomass crop selection. Expert Syst. Appl. 42 (15-16), 6065–6074.

Čuček, L., Klemeš, J.J., Kravanja, Z., 2012. A review of footprint analysis tools for monitoring impacts on sustainability. J. Clean. Prod. 34, 9–20.

De Menna, F., Dietershagen, J., Loubiere, M., Vittuari, M., 2018. Life cycle costing of food waste: a review of methodological approaches. Waste Manag. 73, 1–13.

Dreyer, L.C., Niemann, A.L., Hauschild, M.Z., 2003. Comparison of three different LCIA methods: EDIP97, CML2001 and Eco-indicator 99. Int. J. Life Cycle Assess. 8 (4), 191–200.

Dufour, J., Serrano, D., Galvez, J., Moreno, J., Garcia, C., 2009. Life cycle assessment of processes for hydrogen production. Environmental feasibility and reduction of greenhouse gases emissions. Int. J. Hydrog. Energy 34 (3), 1370–1376.

García-Gusano, D., Iribarren, D., Martín-Gamboa, M., Dufour, J., Espegren, K., Lind, A., 2016. Integration of life-cycle indicators into energy optimisation models: the case study of power generation in Norway. J. Clean. Prod. 112, 2693–2696.

Heijungs, R., Huppes, G., Guinée, J.B., 2010. Life cycle assessment and sustainability analysis of products, materials and technologies. Toward a scientific framework for sustainability life cycle analysis. Polym. Degrad. Stab. 95 (3), 422–428.

Heikkilä, A.M., 1999. Inherent Safety in Process Plant Design: An Index Based Approach. Ph.D. Dissertation. Helsinki University of Technology, Espoo, Finland.

Hermann, B.G., Kroeze, C., Jawjit, W., 2007. Assessing environmental performance by combining life cycle assessment, multi-criteria analysis and environmental performance indicators. J. Clean. Prod. 15 (18), 1787–1796.

Hoekstra, A., 2008. Water Neutral: Reducing and Offsetting the Impacts of Water Footprints. Value of Water Research Report Series No. 28, UNESCO-IHE, Delft, The Netherlands.

Hoekstra, A.Y., 2009. Human appropriation of natural capital: a comparison of ecological footprint and water footprint analysis. Ecol. Econ. 68 (7), 1963–1974.

Ibáñez-Forés, V., Bovea, M.D., Pérez-Belis, V., 2014. A holistic review of applied methodologies for assessing and selecting the optimal technological alternative from a sustainability perspective. J. Clean. Prod. 70, 259–281.

IOS (International Organization for Standardization), 2017. ISO 26000 and OECD Guildlines. Practical Overview of the Linkagess, availible at: https://www.iso.org/files/live/sites/isoorg/files/store/en/PUB100418.pdf.

Jolliet, O., Antón, A., Boulay, A.M., Cherubini, F., Fantke, P., Levasseur, A., McKone, T.E., Michelsen, O., Milà, i., Canals, L., Motoshita, M., Pfister, S., Verones, F., Vigon, B., Frischknecht, R., 2018. Global guidance on environmental life cycle impact assessment indicators: impacts of climate change, fine particulate matter formation, water consumption and land use. Int. J. Life Cycle Assess. 23 (11), 2189–2207.

Kaya, T., Kahraman, C., 2010. Multicriteria renewable energy planning using an integrated fuzzy VIKOR & AHP methodology: the case of Istanbul. Energy 35 (6), 2517–2527.

Kühnen, M., Hahn, R., 2017. Indicators in social life cycle assessment: a review of frameworks, theories, and empirical experience. J. Ind. Ecol. 21, 1547–1565.

Leach, A.M., Galloway, J.N., Bleeker, A., Erisman, J.W., Kohn, R., Kitzes, J., 2012. A nitrogen footprint model to help consumers understand their role in nitrogen losses to the environment. Environ. Dev. 1 (1), 40–66.

Li, Y., Chen, D.W., Liu, M., Wang, R.Z., 2017. Life cycle cost and sensitivity analysis of a hydrogen system using low-price electricity in China. Int. J. Hydrog. Energy 42 (4), 1899–1911.

López-Maldonado, L.A., Ponce-Ortega, J.M., Segovia-Hernández, J.G., 2011. Multiobjective synthesis of heat exchanger networks minimizing the total annual cost and the environmental impact. Appl. Therm. Eng. 31 (6-7), 1099–1113.

Mattioda, R., Teixeira Fernandes, P., Casela, J., Canciglieri Jr., O., 2017. Social life cycle assessment of hydrogen energy technologies. In: Hydrogen Economy. Academic Press, pp. 171–188.
Moradi-Aliabadi, M., Huang, Y.L., 2016. Vector-based sustainability analytics: a methodological study on system transition toward sustainability. Ind. Eng. Chem. Res. 55, 3239–3252.
Papong, S., Itsubo, N., Malakul, P., Shukuya, M., 2015. Development of the social inventory database in Thailand using input–output analysis. Sustainability 7 (6), 7684–7713.
Rees, W.E., 2016. Ecological footprints and appropriated carrying capacity: what urban economics leaves out. Environ. Urban. 4 (2), 121–130.
Ren, J., Xu, D., Cao, H., Wei, S.A., Goodsite, M., Dong, L., 2016. Sustainability decision support framework for industrial system prioritization. AICHE J. 62, 108–130.
Ren, J., Ren, X., Dong, L., Manzardo, A., He, C., Pan, M., 2018. Multiactor multicriteria decision making for life cycle sustainability assessment under uncertainties. AICHE J. 64 (6), 2103–2112.
Sandholzer, D., Narodoslawsky, M., 2007. SPIonExcel—fast and easy calculation of the sustainable process index via computer. Resour. Conserv. Recycl. 50 (2), 130–142.
UNEP (United Nations Environment Program), SETAC (Society of Environmental Toxicology and Chemistry), 2009. Guidelines for Social Life Cycle Assessment of Products.
UNEP (United Nations Environment Programme), 2009. Life Cycle Management: How Business Uses It to Drecrease Footprint, Create Opportunities and Make Value Chains More Sustainable. pp. 181–202.
Van Hoof, G., Vieira, M., Gausman, M., Weisbrod, A., 2013. Indicator selection in life cycle assessment to enable decision making: issues and solutions. Int. J. Life Cycle Assess. 18 (8), 1568–1580.
Verma, A., Kumar, A., 2015. Life cycle assessment of hydrogen production from underground coal gasification. Appl. Energy 147, 556–568.
Wiedmann, T., Lenzen, M., 2007. On the conversion between local and global hectares in Ecological Footprint analysis. Ecol. Econ. 60 (4), 673–677.
Wood, R., Hertwich, E.G., 2012. Economic modelling and indicators in life cycle sustainability assessment. Int. J. Life Cycle Assess. 18 (9), 1710–1721.
Wu, S.R., Chen, J., Apul, D., Fan, P., Yan, Y., Fan, Y., Zhou, P., 2015. Causality in social life cycle impact assessment (SLCIA). Int. J. Life Cycle Assess. 20 (9), 1312–1323.
Xu, D., Lv, L., Ren, J., Shen, W., Wei, S.a., Dong, L., 2017. Life cycle sustainability assessment of chemical processes: a vector-based three-dimensional algorithm coupled with AHP. Ind. Eng. Chem. Res. 56 (39), 11216–11227.
Xu, D., Lv, L., Dong, L., Ren, J., He, C., Manzardo, A., 2018a. Sustainability assessment framework for chemical processes selection under uncertainties: a vector-based algorithm coupled with multicriteria decision-making approaches. Ind. Eng. Chem. Res. 57 (23), 7999–8010.
Xu, D., Lv, L., Ren, X., Ren, J., Dong, L., 2018b. Route selection for low-carbon ammonia production: a sustainability prioritization framework based-on the combined weights and projection ranking by similarity to referencing vector method. J. Clean. Prod. 193, 263–276.
Zakeri, B., Syri, S., 2015. Electrical energy storage systems: a comparative life cycle cost analysis. Renew. Sust. Energ. Rev. 42, 569–596.

CHAPTER

12

Life cycle sustainability assessment and decision-making under uncertainties

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12.1 Introduction

Decision-making can be a complex process aiming to establish satisfying solutions or possible compromises submitted to the judgment of a decision-maker or a group of decision-makers under a scientific base. Even when only one decision-maker is involved in the decision-making process, rarely does the decision-maker have in mind only one criterion. This means that the decision-making more often involves multi-criteria than a monocriterion. In this sense, the multi-criteria approaches have been playing an important role for analyzing and structuring any decision-making process (Greco et al., 2016).

Decision-making based on life cycle sustainability assessment (LCSA) clearly represents a multi-criteria decision approach. LCSA adopts the widespread three-dimensional view of sustainability (environment, economic, and social). It is considered a promising methodology for developing a transparent, robust, and comprehensive approach towards sustainability (Sala et al., 2012), and has been largely discussed in the literature (e.g., Halog and Manik, 2011; Santoyo-Castelazo and Azapagic, 2014; Akhtar et al., 2015; Ren et al., 2015).

Decision-making using LCSA is not an easy task since it requires taking into account incommensurable dimensions and many uncertainties due to the model parameters and input data. As uncertainty refers to lacking complete knowledge or confidence regarding some situation, it is per se a complex issue to be handled in any decision-making. However, decision problems involving LCSA adds a multifaceted meaning to the uncertainty, making the decision much more complex. In addition, one of the main challenges of LCSA is in how to build a comprehensive judgment of the sustainability performance of products, services, and processes, taking into account several and distinct indicators, and avoiding reductionist approaches (Sala et al., 2012). This is precisely where multi-criteria decision analysis (MCDA)

Life Cycle Sustainability Assessment for Decision-Making
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can be useful, which is applied to support decision-making in problems where several criteria are taken into consideration to evaluate alternatives (solutions).

MCDA can be useful as a tool for conflict management, since it allows consideration by various decision-makers and stakeholders, who often have conflicting interests. It aims to organize the mixed available information and to help decision-makers to aggregate the criteria and identify the pros and cons of each alternative, by enabling them to have a diversity and a large number of indicators being analyzed in the same framework, independently, if they are expressed in a quantitative or qualitative manner (Matteson, 2014; Clímaco and Valle, 2014; Recchia, 2011).

This chapter aims to contribute to the discussion on how to consider the uncertainties inherent in LCSA in decision-making through an integrated LCSA and MCDA approach. It is organized in four sections, including this introduction. Section 12.2 presents the meaning of uncertainty and its types, as well as a brief description of the decision-making under uncertainties. Section 12.3 focuses on the decision-making of LCSA taking into account uncertainties, mainly regarding MCDA integrated with LCSA. Some examples of the literature are presented. Finally, Section 12.4 gives final remarks and points out the relevance of robustness in decision-making on LCSA.

12.2 The decision-making under uncertainties

12.2.1 What is uncertainty?

There are different meanings to the term uncertainty. Roughly speaking, uncertainty refers to something lacking complete knowledge or confidence. In a decision-making context, specially that dealing with features of the real-world, uncertainty refers to the inability of the decision-maker to describe, prescribe, or predict deterministically and numerical a system and its behavior due to the lack of quantitative or qualitative information about the decision problem (Zimmermann, 2001).

Uncertainty can be treated in different forms; for example, Zimmermann (2001) differentiates between three forms: stochastic, linguistic, and informational uncertainty. The first one can be handled based on theory of probability and statistics, in which the decision-making depends on events or statements that are well defined (e.g., the *n*% of risk/probability a nuclear accident to occur). In contrast, both linguistic and informational uncertainties are related to the vagueness concerning the establishment of the meaning of events, statements, or situation problems. Therefore, they are considered as fuzziness due to the lack of precision and the indefinite nature of human language (linguistic uncertainty) and the high quantity of information required to describe a situation (informational uncertainty).

Uncertainty can also be treated as internal or external uncertainty. The former is related to the decision-maker's values and judgments that will influence the decision-making, and the latter refers to the imperfect knowledge concerning the situation and its behavior or consequences. In other words, internal uncertainty can arise from the process of decision problem structuring and analysis, and external uncertainty from the nature of the environment from where the decision-making comes, which may be out of the control of the decision-maker (Levary and Wan, 1998). In summary, it is possible to make some associations taking into account the different kinds of uncertainty, thus leading to a multifaceted meaning (Fig. 12.1).

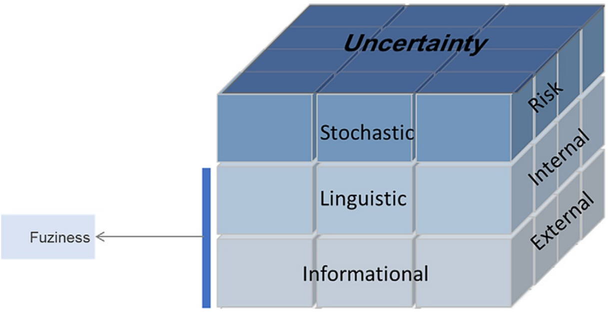

FIG. 12.1 Different types of uncertainty.

As risk usually is related to situations where the probabilities of consequences/outcomes are objectively known (Millet and Wedley, 2002), the stochastic uncertainty can be associated with the term risk. As linguistic uncertainty is related to the imprecision or ambiguity in human judgments, it can be associated with internal uncertainty. Finally, as informational uncertainty refers to incomplete, uncertain, or even ambiguous information, it can be associated with external uncertainty. In addition, besides the external uncertainty about the environment, it is also possible to have uncertainties related to the interconnections between decisions, i.e., how the decision and its outcomes influence another decision. All of these issues must be properly treated in the decision-making.

12.2.2 Decision-making under uncertainties

Decision-making is usually associated with a problem-solving process, of which an alternative (or action) must be chosen. In general, the decision-making starts with the identification and definition of the problem situation, and ends with the choosing of a compromise alternative (solution) (Fig. 12.2). The term compromise solution is adopted, especially in

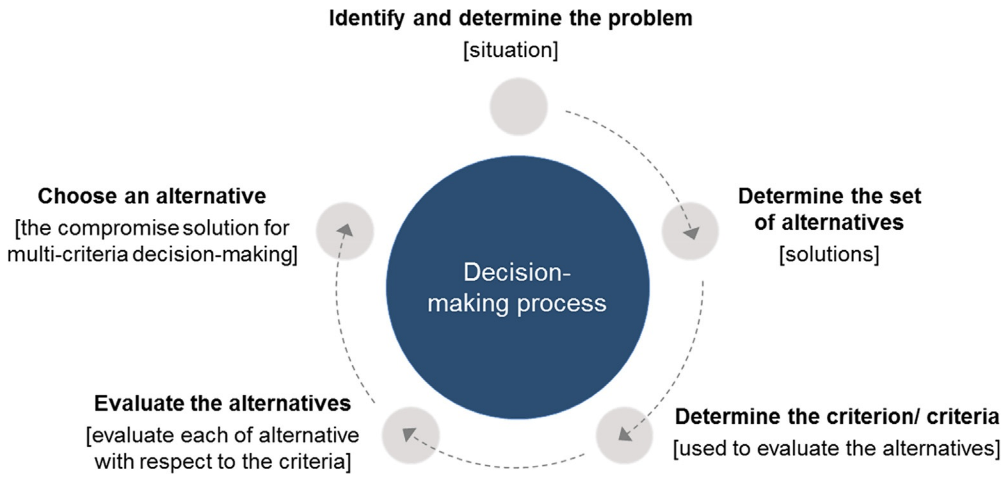

FIG. 12.2 The five steps of a typical decision-making process.

multi-criteria decision-making problems, where there is no solution capable of satisfying all criteria at the same time (this should be an ideal solution).

The alternatives can be evaluated with respect to one criterion (monocriterion approach) or to multiple criteria (multi-criteria approach). Therefore, a multi-criteria decision problem is one in which more than one criterion is considered in the assessment. For both approaches, uncertainties may appear, due to the simple fact that behind any decision-making process there is at least one decision-maker, or even due to the nature of the decision-problem.

The uncertainty can be handled in different ways. For instance, internal uncertainties can be resolved through a better structuring of the decision problem, or if they are not resolvable, by carrying out an appropriate sensitivity and robustness analysis (Belton and Stewart, 2002). External uncertainties can be handled by a consistent understanding of the environment from which the decision problem arises, as well as by an expansion of the decision area in order to incorporate interconnected decisions and consequences (Greco et al., 2016).

In practice, these uncertainties are handled by an appropriate sensitivity analysis of the results, i.e., after the application of a deterministic multi-criteria method, for example, to identify the compromise solution. Because of this, a sensitivity analysis is usually carried out in the decision-making process dealing with uncertainties, as presented in Fig. 12.3.

It is important to note that stochastic uncertainties can be handled through risk analysis, which is the process of predicting a decision's outcome in face of uncertainties, and can be conducted with or without simulation. Even in this case, a consistent and better problem structuring is needed.

12.3 LCSA and decision-making under uncertainties

12.3.1 Uncertainties inherent to LCSA

Despite uncertainties being inherent to decision problems involving sustainability assessment, a review carried out by Thies et al. (2019) pointed out that the majority of LCSA studies

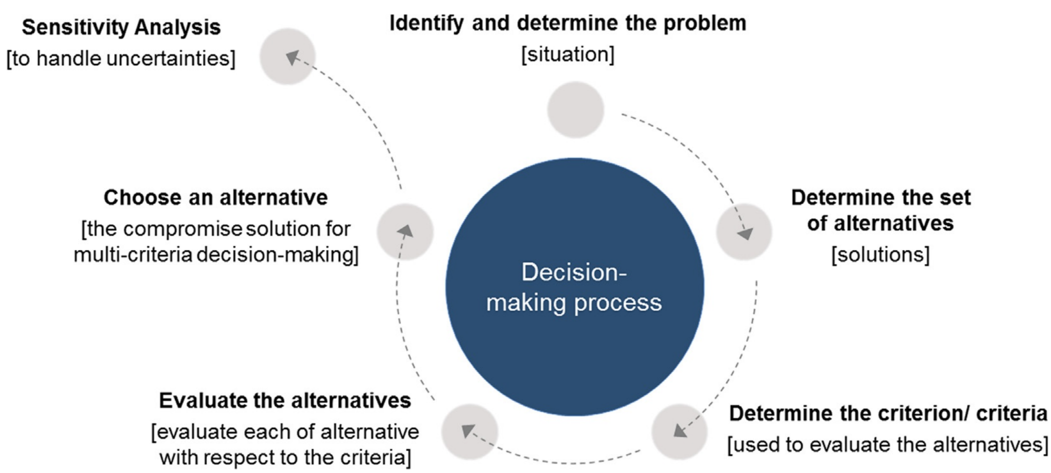

FIG. 12.3 A six-step decision-making process dealing with uncertainties.

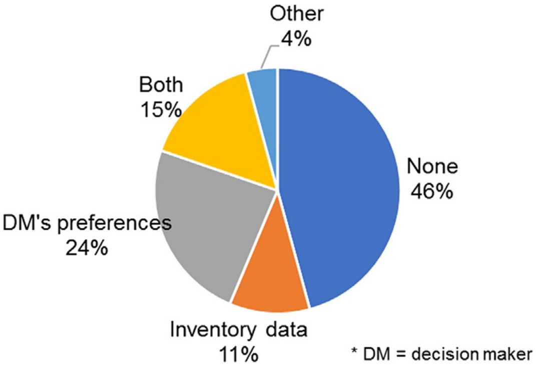

FIG. 12.4 Types of uncertainty in LCSA studies. *Based on data from Thies, C., et al., 2019. Operations research for sustainability assessment of products: a review. Eur. J. Oper. Res. 274 (1) 1–21.*

do not take into account any kind of uncertainty. From those studies that consider uncertainties in the analysis, most of them are related to decision makers' preference, which refers to the vagueness and ambiguity of them regarding the priority of each criterion. Furthermore, uncertainties related to inventory data are also treated, typically those associated with the variability in input and output flows of the product system modeled (Fig. 12.4). Therefore, decision-maker preferences are related to internal uncertainties, while inventory data is related to external uncertainties.

The authors also identified the methods used to handle these uncertainties. Sensitivity analysis is the main procedure adopted in LCSA studies, since it is easy to conduct and allows investigation of the stability of the LCSA results under some conditions of uncertainty, i.e., to see how changes in critical parameters could affect the LCSA outcomes.

The sensitivity analysis can comprise several strategies, such as: changes in weight factor analysis (Dong et al., 2014); different weight combinations (Milani, 2011; Manzardo et al., 2014; Akhtar et al., 2015; Ren et al., 2015); maximizing the main criteria (von Doderer and Kleynhans, 2014), using minimum, nominal, and maximum values for each criteria (Klein and Whalley, 2015); random techniques (Basson and Petrie, 2007; Hanandeh and El-zein, 2009); statistical tools (Halog and Manik, 2011); Monte-Carlo simulation (Sparrevik et al., 2012; Basson and Petrie, 2007; Hanandeh and El-zein, 2010); and fuzzy theory (Liu et al., 2012; Pires and Chang, 2011).

Despite being uncommon in practice, some multi-criteria approaches can handle LCSA uncertainties. For instance, internal uncertainties can be resolved through fuzzy set approaches, rough set approaches, and identifying potentially optimal solutions amongst uncertainty ranges; while external uncertainties can be handled by stochastic dominance concepts, the use of surrogate risk measures as additional decision criteria, and the integration of MCDA and scenario planning (Greco et al., 2016).

12.3.2 The integration of LCSA and MCDA

As mentioned before, MCDA is a powerful tool to be integrated with LCSA because it is capable of handling several issues at the same time, for instance: different stakeholders involved, who often have conflict interests; or a large number of indicators to be addressed, which it may not be easy to express quantitatively in a consistent manner, apart from some trade-offs presented between them.

12.3.2.1 A brief presentation of MCDA methods

There are several MCDA methods that can be classified according to the aggregation procedure adopted to take into account all criteria analyzed. The most traditional approach is the one based on utility or value-function by single synthesizing criterion, in which the criteria multiplicity is reduced to a unique criterion by using formal rules mathematically structured, e.g., the multi-attribute utility theory (MAUT). Several commonly used methods belong to this group, such as: technique for order preference by similarity to ideal solution (TOPSIS); analytic hierarchy process (AHP); and analytic network process (ANP).

TOPSIS, also known as a reference point approach, is based on the concept that the alternative chosen should be the nearest to the ideal solution and farthest from the negative-ideal solution (Greco et al., 2016). Developed by Saaty (1980), AHP is based on the creation of a hierarchical structure of criteria and alternatives, which are pairwise compared to assess the relative preference among each other according to decision-maker preferences. ANP is a derived form of AHP that comprises the generalization of hierarchies to networks with dependence and feedback (Greco et al., 2016).

These methods assume some compensability among criteria, i.e., trade-offs where a disadvantage on a criterion can be compensated by a sufficient advantage on another criterion (Rowley et al., 2012; Benoit and Rousseaux, 2003; Guitouni and Martel, 1998). However, TOPSIS and AHP have been largely used in sustainable related decisions (e.g., AHP: Myllyviita et al., 2013; von Doderer and Kleynhans, 2014, Akhtar et al., 2015; TOPSIS: Su et al., 2010; Dong et al., 2014; Sedláková et al., 2014, 2015). This compensability may be very problematic in decisions involving sustainability (Munda, 2008).

The other MCDA approach is based on a synthesizing preference relational system, which involves pairwise comparison of the alternatives on each criterion supported by well-structured mathematical rules based on discrimination thresholds and veto threshold; examples of this group include the outranking methods such as the preference ranking organization method for enrichment evaluation (PROMETHEE) family and elimination and choice expressing reality (ELECTRE) family.

PROMETHEE is composed of six methods, including:

- partial ranking (PROMETHEE I);
- complete ranking (PROMETHEE II);
- ranking based on intervals (PROMETHEE III);
- ranking based on continuous case (PROMETHEE IV);
- with constraints segmentation (PROMETHEE V); and
- with representation of the human brain (PROMETHEE VI).

All of these methods are based on positive and negative preference flows for each alternative, according to the selected criteria preferences (weights) (Greco et al., 2016).

ELECTRE comprises six different methods:

- ELECTRE I is dedicated to choice problems, with the aim of reducing the size of a non-dominated set of alternatives.

- ELECTRE IS, an improved form of ELECTRE I, uses an indifference threshold.
- ELECTRE II ranks alternatives from the best to worst option, using either strong or weak relations.
- ELECTRE III allows the use of pseudo-criteria and fuzzy outranking relations.
- ELECTRE IV is similar to ELECTRE III, but without the use of criteria weights.
- ELECTRE TRI is used for dealing with ordinal classification problems (Roy and Bouyssou, 1993).

Four of these ELECTRE methods have fuzzy outranking relationship of alternatives (Fig. 12.5), which means they are able to handle uncertain and ambiguous information (Guitouni and Martel, 1998; Rogers and Bruen, 1998). Therefore, for decision-making where uncertainties inherent to criterion estimates can be significant, such as sustainable problems, the choice of a fuzzy decision model such as ELECTRE III seems more appropriate (Rogers et al., 2000). In fact, ELECTRE III is the most popular of the ELECTRE family methods, with strong application in environmental problems, especially those involving complex decision-making such as energy management, chemical and biochemical engineering, policy, social, and education (Govindan and Jepsen, 2016).

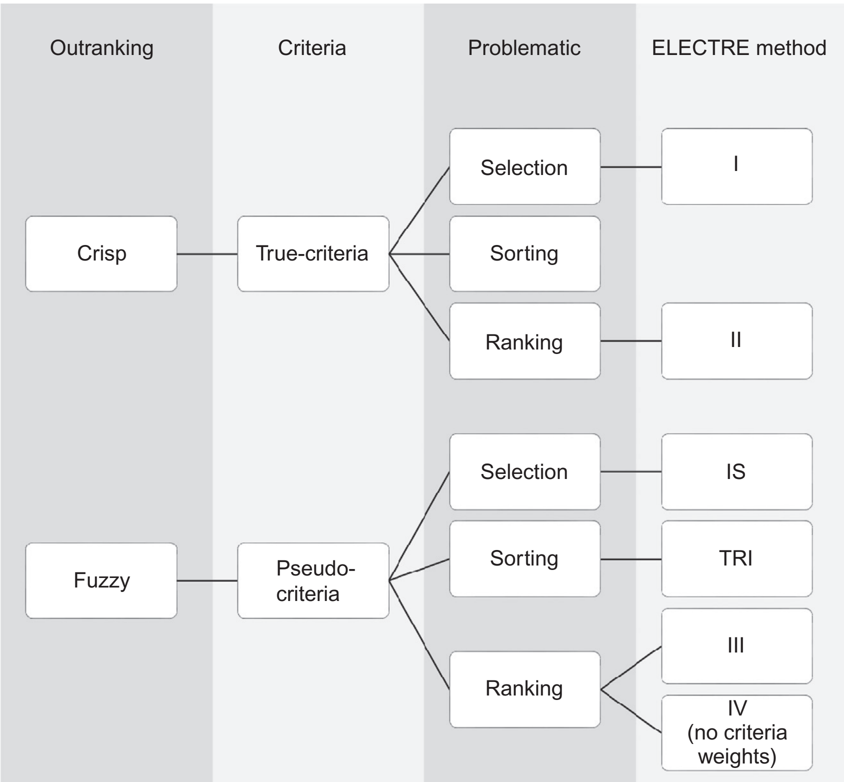

FIG. 12.5 The ELECTRE family of methods.

There are other methods that are not in accordance with these approaches due to their interactive nature. An example could be found in Angelo et al. (2017), where an interactive learning-oriented multi-attribute additive model using imprecise information, called VIP-Analysis, was applied in order to identify the most preferable organic waste treatment in terms of life cycle assessment (LCA) results, taking the domestic solid waste management in the city of Rio de Janeiro as a case study.

Moreover, there is another approach used when the decision problem involves an infinite or a very large number of alternatives, known as multi-objective decision-making. It comprises programming methods such as multi-objective optimization and goal programming, and is, in general, restricted to operational decisions (Azapagic and Perdan, 2005). However, it must be remarked that facing the multiplicity of MCDA methods, none can be considered as the best method appropriated to all decision-making situations (Guitouni and Martel, 1998).

12.3.2.2 Main MCDA methods applied in LCSA studies

As mentioned before, MCDA methods can be applied in an integrated manner to LCSA in order to aggregate its results, providing a better understanding of them, thus potentializing the interpretation phase, and aiding the decision. Most LCSA studies adopt MCDA methods to support the decision-making; of these, AHP, PROMETHEE, ELECTRE, and TOPSIS comprise the majority (Fig. 12.6) (Thies et al., 2019). Unsurprisingly, almost half of the LCSA studies reviewed by Thies and co-authors have adopted the AHP method. It is the most widely applied MCDA method in decision-making (Vaidya and Kumar, 2006).

12.3.2.3 Examples of MCDA integrated with LCSA under uncertainties

LCSA integrated with ELECTRE III

This section is based on a sustainable life cycle analysis developed by Angelo et al. (2019), where two urban transport systems (bus rapid transit and metro) of the city of Rio de Janeiro are evaluated by considering environmental performance resulted from a LCA study, economic, and social indicators in the same framework through ELECTRE III.

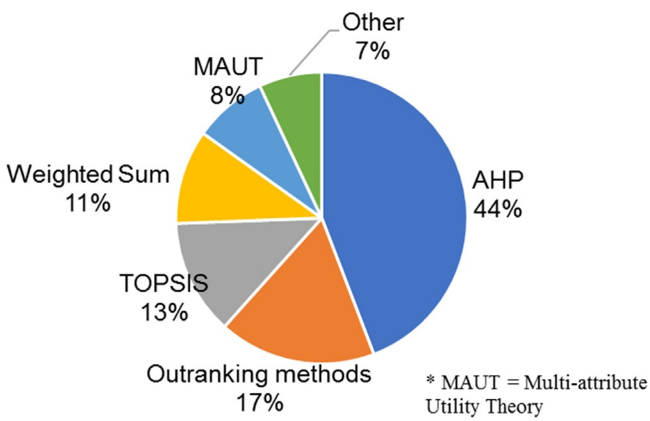

FIG. 12.6 Most frequent MCDA methods in LCSA studies. *Based on data from Thies, C., et al., 2019. Operations research for sustainability assessment of products: a review. Eur. J. Oper. Res. 274 (1) 1–21.*

A brief description of ELECTRE III

Vagueness and uncertainty are treated by ELECTRE III by introducing the indifference, preference, and veto thresholds to establish a pseudo-criterion, that allows creation of an intermediary zone in which decision-makers' information is contradictory or indeterminate (Rogers et al., 2000). The outranking relationship of each pair of alternatives is evaluated by assuming these thresholds, making it possible to determine if two alternatives are indifferently, weakly, or strongly preferable to each other, and even if they are incomparable, if there is no sufficient information to distinguish the preference between them.

For instance, let's take two alternatives, a and b, to be compared on one criterion, g. They are indifferent (aIb) if the difference between the performances of these two alternatives is smaller than the indifference threshold (q). The alternative a is weak preferred to b of the difference of their performances is between the thresholds of indifference (q) and preference (p). The strong preference occurs when the difference of their performances is greater than the preference threshold (p). Finally, the incomparability of alternatives occurs when this difference surpasses the veto threshold (v) or if there is no sufficient information to compare the alternatives (Fig. 12.7).

After the pairwise comparison of alternatives considering all criteria defined in the decision problem structuring, concordance and discordance indices are created. The former is a fuzzy index indicating the truth of the assertion "alternative a is at least as good as b on such criterion g." The discordance index indicates if some criterion is more or less discordant with the previous assertion. By taking into account both indices, a credibility index is calculated; then, the ranking of alternatives can be built through distillations procedures. It must be noted that the way the credibility index is constructed excludes the possibility of compensation between criteria (Dias et al., 2006).

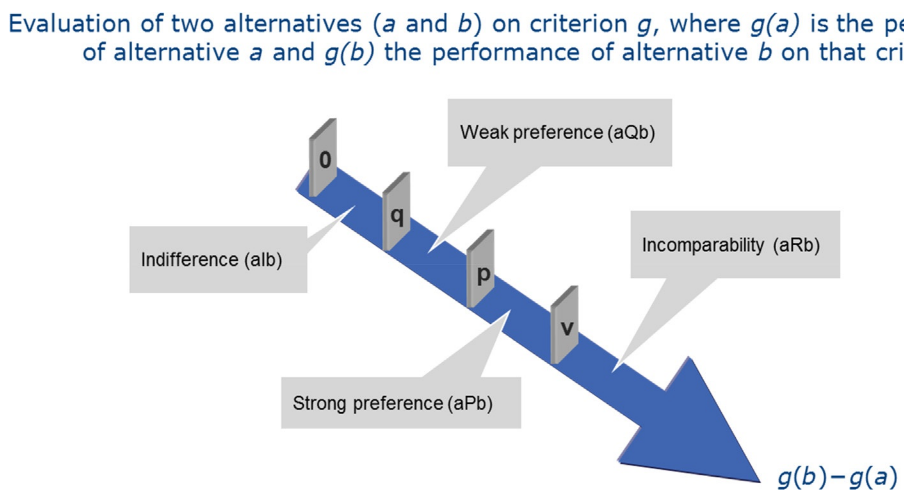

FIG. 12.7 The outranking relationship in ELECTRE III.

The case study presentation

In order to host the Olympic and Paralympic Games 2016, the city of Rio de Janeiro massively invested in expanding collective transportation networks, more especially extending and improving the quality of metro services and expanding the use of bus rapid transit (BRT) systems. It is well known that an efficient public transport system goes beyond the simple improvement of population mobility. The increased use of non-motorized transport and public transport are directly associated with environmental benefits such as reducing greenhouse gas (GHG) emissions, which influences reducing respiratory diseases, improving human health (Banister, 2008). Moreover, GHG mitigation measures applied in the transport sector can enhance positive impacts for all three sustainability dimensions (IPCC, 2014).

A sustainable lifecycle analysis was carried out comparing BRT and metro, focusing on BRT Transcarioca and Metro Line 4. With 39 km of extension, the BRT Transcarioca serves 27 neighborhoods, has 47 stations, and allows the integration with rail systems, metro, other BRT lines and connects with the International Airport. The majority of its users are middle and low-income people (BRT Rio, 2019). Metro Line 4 extends for 16 km underground, connecting the south region to the west side of the city, which is the region with the largest population growth in recent years. Travel between these regions can require up to 2 h by car or bus when traffic is heavy, while by metro this time can be reduced to 30 min (Nobrega, 2012).

The sustainable life cycle analysis comprised an integrated assessment of nine criteria: three criteria of each sustainable dimension—environmental, social, and economic (Table 12.1). ELECTRE III was applied to facilitate the interpretation of these indicators and indicate the most sustainable option of public transport. Equal weights were assumed

TABLE 12.1 Criteria used in the assessment.

Sustainable dimension	Criterion	Unit	Reference
Environment	Cr1: Climate change	kg CO_2e	Martins and Angelo (2018)
	Cr2: Particulate matter formation	10^{-3} kg PM10e	
	Cr3: Photochemical oxidant formation	10^{-2} kg NMVOC	
Social	Cr4: Perception of quality of service	% of users that considered good/very good	ITDP (2018)
	Cr5: Travel time reduction	Minute	
	Cr6: Perception of transport expenditures reduction	% of population that earns up to 1 minimum wage	
Economic	Cr7: Demand	Users per day (Thousand)	Deng and Nelson (2011)
	Cr8: Investment on infrastructure	Brazilian currency (Billion)	Restum (2018), Castro et al. (2015)
	Cr9: Operational and maintenance costs and expenditures	Brazilian currency (Thousand)	Baker Tilly Brazil (2017), Invepar (2017)

for the criteria, and the thresholds adopted were 0.1 for indifference, 0.2 for preference, and no veto threshold.

The three environmental criteria come from the LCA study carried out by Martins and Angelo (2018). Climate change, particulate matter formation, and photochemical oxidant formation were considered as most relevant for Rio de Janeiro city due to the existing reduction goals for anthropic GHG emissions established by a municipal law of 2011 and the constant monitoring of the air quality in terms of CO, SO_2, O_3, NO_x, and PM10 emissions (SMAC, 2012). The attributional comparative LCA of BRT Transcarioca and Metro Line 4 adopted 1 passenger kilometer travelled (pkm) as functional unit. Vehicle manufacture, infrastructure construction, maintenance, and operation phase were assumed as system boundaries. The end of life was not included in the analysis due to the high complexity of evaluating final destination routes as well as the lack of data available in Brazil of the waste sector.

The social criteria chosen were obtained from a field survey (ITDP, 2018) considering passengers point-of-view as well as the institutions involved in the planning, management, and operation process of these transportation systems. The criterion travel time refers to the average time gained in the trip with the implementation of the system, the quality of service is related to comfort and safety, and the perception about the expenses is correlated to the passenger's monthly income.

The economic criteria were chosen as they reflect the most relevant indicators of an economic analysis. The number of users per day is an acknowledged criterion for operational and financial performance of public transport systems. The infrastructure investment is crucial in the comparison, since the investment required is significant and distinct for each system. The annual operational and maintenance costs and expenditures are crucial information that must be managed (Table 12.2).

Due to the uncertainties and vagueness of real-world decision-problems involving sustainability, mainly regarding lower accuracy of inventory data and of decision makers' preference, the indifference and preference thresholds are defined to illustrate the preference or indifference of one alternative compared to another. For instance, the indifference threshold 0.1 means for such criterion, the acceptance threshold of an alternative to another is 10%. According to Rogers and Bruen (1998), the thresholds are linked to the margin of uncertainty/error associated with the criterion in question. Therefore, the fuzzy relationships

TABLE 12.2 Performance matrix.

Alternative/Criteria	Cr1	Cr2	Cr3	Cr4	Cr5	Cr6	Cr7	Cr8	Cr9
BRT	0.040	0.059	0.021	0.660	38	0.420	234	2	766,099
Metro	0.035	0.087	0.024	0.780	27	0.150	160	10.4	85,100
Direction	Min	Min	Min	Max	Max	Max	Max	Min	Min
Indifference (q)	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Preference (p)	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
Veto (v)	NA	NA	NA	NA	NA	NA	NA	NA	NA
Weights	1	1	1	1	1	1	1	1	1

NA, *not applied.*

between alternatives built on each criterion serve as a basis to determine the concordance and discordance indices, which leads to the credibility index, then the ranking the alternatives through distillations procedures. The results of ELECTRE III have shown BRT Transcarioca as the most sustainable option compared to Metro Line 4. In fact, from a social and economic-perspective, BRT has favorable performance for the majority of criteria, while in the environment dimension they are similar.

A sensitivity analysis was carried out in order to investigate the stability of the results by varying some modeling parameters. The results have shown an outranking relationship of indifference between BRT and Metro Line 4 when the criteria related to investment and costs were removed from the analysis, reflecting the fragility of the results, which could be handled by taking into account more criteria in the assessment; likewise annual revenues, utility factor, jobs created, social acceptability, social benefit and security.

LCSA integrated with non-classical approaches

The uncertainties of LCSA studies were treated by Ren and Toniolo (2018) through the application of a novel MCDA method, which allows the use of interval numbers in the performance matrix, thus leading to incorporate the uncertainties of LCSA by taking a hydrogen production as a case study. Moreover, the authors have treated the uncertainties of the weighting process, since defining weights unambiguously may not be an easy task in sustainable problems.

The authors built the performance matrix by taking into account the LCSA results as interval numbers, established the weights by carrying out a novel fuzzy weighting method (an improved version of the decision-making trial and evaluation laboratory (DEMATEL)) to address interdependences and interactions between the criteria assessed. Thus, they ranked the alternatives by applying an improved version of the distance from average solution (EDAS) to take into account the interval evaluation. This application do not only indicated the feasibility of this novel approach but also its accuracy for sustainable problems.

The internal uncertainties of LCSA are also treated by Tarne et al. (2019). Fifty-four decision makers from different areas of a German automotive company were asked through limit conjoint analysis to rank the economic, environment, and social performance of a vehicle component. The results were evaluated by functional clusters and for the entire sample, and the authors observed a large spread in weighting without clear clustering, and, on average, all the three sustainability dimensions were almost equally important.

Uncertainties concerning the weighting process in decision-making involving sustainability are also clear in their study, since the analysis has pointed out different points of view depending on the area of acting of the decision-maker, even at the same company. Those people working in sustainable areas gave more importance to the social dimension, followed by environment and economic. While non-sustainability people put economic at first position. The approach proposed by Tarne and co-authors enabled the decision-making within LCSA by treating the uncertainties of the weighting process.

External uncertainties were treated by Do Carmo et al. (2018) through applying a three-step methodology: (1) assessment of LCSA uncertainties; (2) extending LCSA performances uncertainty to MCDA methods (eg. weighted sum, PROMETHEE and TOPSIS); and (3) interpreting the stochastic rankings resulted from the MCDA methods; by taking the life cycle of truck tires in Brazil as a case study.

The authors considered the reference flows of the system product analyzed, the use of data from Ecoinvent database, transport distances, the end-of-life benefits, land use change, etc. as sources of uncertainties of the environmental dimension in the LCSA study. They performed a Monte Carlo simulation in order to represent these uncertainties, then applied the MCDA methods to propagate the uncertainties and analyzed the probabilities to establish a general ranking. Their results demonstrated the feasibility to account LCSA uncertainties in a decision-making through MCDA, hence determining a compromise solution in a more robust way.

12.4 Final remarks

The multi-dimensional nature of LCSA requires a multi-criteria approach on decision-making. It is consensual that it brings more complexity to the decision process, since different points of view of stakeholders and diverse indicators must be considered. In this sense, MCDA methods have been recognized as a powerful tool, not only to aggregate LCSA results, but also to allow taking into account trade-offs, and quantitative and qualitative indicators.

Besides these issues, the assessment of the sustainability performance of products, services, and processes has plenty of external uncertainties, related to modeling parameters and inventory data. Another uncertainty typically critical in decision-making based on LCSA refers to introducing weights to the sustainability dimensions, which can be considered an internal uncertainty. Therefore, the multifaceted uncertainties rooted in LCSA require a multi-criteria approach capable of handling them and providing robust and accurate decision-making, going beyond a simple sensitivity analysis.

In practice, there are several LCSA studies integrating MCDA methods to support decision-making, but very few studies have adopted an approach dealing with LCSA uncertainties. A traditional MCDA outranking method from the European School can treat uncertainties and imprecise information by integrating fuzziness into the outranking relationships of alternatives. ELECTRE III provides the possibility for a decision-maker to analyze both the qualitative and quantitative indicators at different levels of ambiguity. Moreover, as a partially or non-compensatory method, ELECTRE III is a powerful tool for decision-making involving sustainability.

The application of non-traditional multi-criteria approaches in LCSA to aid decision-making treating its uncertainties is expected to increase, as it is feasible to provide robust decision-making through these approaches. However, they are not so widely used, perhaps because they are often unworkable or difficult to put in practice in real-world problems. Despite this, the application of these methods has to be encouraged and, from that, noted in the literature as a reference for the LCSA and decision-making under uncertainties.

References

Akhtar, S., et al., 2015. Life cycle sustainability assessment (LCSA) for selection of sewer pipe materials. Clean Techn. Environ. Policy 17 (4), 973–992.

Angelo, A.C.M., et al., 2017. Life cycle assessment and multi-criteria decision analysis: selection of a strategy for domestic food waste management in Rio de Janeiro. J. Clean. Prod. 143, 744–756.

Angelo, A.C.M., et al., 2019. Comparative Sustainable Lifecycle Analysis of Bus Rapid Transit (BRT) and Metro: a case study of Rio de Janeiro City. In: 4th International Conference on Energy and Environment: Bringing Together Engineering and Economics, Guimarães, Portugal, 16-17 May.

Azapagic, A., Perdan, S., 2005. An integrated sustainability decision-support framework Part II: Problem analysis. Int. J. Sus. Dev. World Ecol. 12, 112–131.

Baker Tilly Brazil, 2017. Consórcio Transcarioca Transportes. Demonstrações Contábeis e Relatório dos Auditores Independentes. Rio de Janeiro, https://cpidosonibus.com.br/files/drf/2016_RTR4_Transcarioca.pdf.

Banister, D., 2008. The sustainable mobility paradigm. Transp. Policy 15, 73–80.

Basson, L., Petrie, J., 2007. A critical systems approach to decision support for process engineering. Comput. Chem. Eng. 31, 876–888.

Belton, V., Stewart, T., 2002. Multiple Criteria Decision Analysis: An Integrated Approach. Springer, Boston, MA. https://doi.org/10.1007/978-1-4615-1495-4_8.

Benoit, V., Rousseaux, P., 2003. Aid for aggregating the impacts in life cycle assessment. Int. J. Life Cycle Assess. 8 (2), 74–82.

BRT Rio, 2019. Conheça o BRT. http://brtrio.com/.

Castro, D.G., et al., 2015. Rio de Janeiro: Os impactos da copa do mundo 2014 e das Olimpíadas 2016, first ed. Observatório das Metrópoles, Rio de Janeiro.

Clímaco, J.C.N., Valle, R.A.B., 2014. MCDA and LCSA—a note on the aggregation of the preferences. In: Proceedings of KICSS 2014, pp. 93–99.

Deng, T., Nelson, J.D., 2011. Recent developments in bus rapid transit: a review of the literature. J. Transp. Rev. 31 (1), 69–96.

Dias, J.A., Figueira, J.R. and Roy, B. 2006. The Software ELECTRE III-IV Methodology and User Manual (Version 3.X). Lisboa: Centro de Estudos de Gestão (CEG-IST). ISSN 1646-2955.

Do Carmo, B.B.T., Margini, M., Baptiste, P., 2018. Propagating uncertainty in life cycle sustainability assessment into decision-making problems: a multiple criteria decision aid approach. In: Benetto, E., Gericke, K., Guiton, M. (Eds.), Designing Sustainable Technologies, Products and Policies—From Science to Innovation. Springer Open. https://doi.org/10.1007/978-3-319-66981-6.

Dong, J., et al., 2014. Energy–environment–economy assessment of waste management systems from a life cycle perspective: model development and case study. Appl. Energy 114, 400–408.

Govindan, K., Jepsen, M.B., 2016. ELECTRE: a comprehensive literature review on methodologies and applications. Eur. J. Oper. Res. 250 (1), 1–29.

Greco, S., Ehrgott, M., Figueira, J.R., 2016. In: Greco, S., Ehrgott, M., Figueira, J.R. (Eds.), Multiple Criteria Decision Analysis—State of the Art Surveys, second ed. Springer-Verlag, New York. https://doi.org/10.1007/978-1-4939-3094-4.

Guitouni, A., Martel, J., 1998. Tentative guidelines to help choosing an appropriate MCDA method. Eur. J. Oper. Res. 109 (2), 501–521.

Halog, A., Manik, Y., 2011. Advancing integrated systems modelling framework for life cycle sustainability assessment. Sustainability 3 (2), 469–499.

Hanandeh, A., El-zein, A., 2009. Strategies for the municipal waste management system to take advantage of carbon trading under competing policies: the role of energy from waste in Sydney. Waste Manag. 29 (7), 2188–2194.

Hanandeh, A., El-zein, A., 2010. The development and application of multi-criteria decision-making tool with consideration of uncertainty: the selection of a management strategy for the bio-degradable fraction in the municipal solid waste. Bioresour. Technol. 101 (2), 555–561.

Invepar, 2017. Relatório anual 2017. http://www.invepar.com.br.

IPCC, 2014. In: Core Writing Team, Pachauri, R.K., Meyer, L.A. (Eds.), Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. IPCC, Geneva, Switzerland. https://doi.org/10.1017/CBO9781107415324.

ITDP, 2018. Linha 4 do Metrô do Rio de Janeiro Avaliação de resultados e recomendações de melhoria. https://itdpbrasil.org.

Klein, S.J.W., Whalley, S., 2015. Comparing the sustainability of U.S. electricity options through multi-criteria decision analysis. Energy Policy 79, 127–149.

Levary, R.R., Wan, K., 1998. A simulation approach for handling uncertainty in the analytic hierarchy process. Eur. J. Oper. Res. 106 (1), 116–122.

Liu, K.F., et al., 2012. Combining risk assessment, life cycle assessment, and multi-criteria decision analysis to estimate environmental aspects in environmental management system. Int. J. Life Cycle Assess. 17 (7), 845–862.

Manzardo, A., et al., 2014. Integration of water footprint accounting and costs for optimal chemical pulp supply mix in paper industry. J. Clean. Prod. 72, 167–173.

Martins, I.M., Angelo, A.C.M., 2018. Avaliação Comparativa do Ciclo de Vida do Bus Rapid Transite (BRT) e Metrô: um estudo de caso do BRT Transcarioca e da Linha 4 do metrô do Rio de Janeiro. In: VI Congresso Brasileiro de Gestão do Ciclo de Vida, Brasilia, Brasil.

Matteson, S., 2014. Methods for multi-criteria sustainability and reliability assessments of power systems. Energy 71, 130–136.

Milani, A.S., 2011. Multiple criteria decision making with life cycle assessment for material selection of composites. Express Polym Lett 5 (12), 1062–1074.

Millet, I., Wedley, W.C., 2002. Modelling risk and uncertainty with the analytic hierarchy process. J. Multi-Criteria Decis. Anal. 11 (2), 97–107.

Munda, G., 2008. Social Multi-Criteria Evaluation for a Sustainable Economy. Springer.

Myllyviita, T., et al., 2013. Sustainability assessment of wood-based bioenergy—a methodological framework and a case-study. Biomass Bioenergy 59, 293–299.

Nobrega, P., 2012. Metro Line 4 in Rio de Janeiro. https://www.tunnel-online.info/download/484280/2012_06_Metro_Line_4_in_Rio_de_Janeiro.pdf.

Pires, A., Chang, N., 2011. An AHP-based fuzzy interval TOPSIS assessment for sustainable expansion of the solid waste management system in Setúbal Peninsula, Portugal. Resour. Conserv. Recycl. 56, 7–21.

Recchia, L. (Ed.), 2011. Multi-criteria analysis and LCA techniques: with applications to agro-engineering problems. In: Green Energy and Technology. Springer-Verlag, London; New York.

Ren, J., Toniolo, S., 2018. Life cycle sustainability decision-support framework for ranking of hydrogen production pathways under uncertainties: an interval multi-criteria decision making approach. J. Clean. Prod. 175, 222–236.

Ren, J., et al., 2015. Prioritization of bioethanol production pathways in china based on life cycle sustainability assessment and multicriteria decision-making. Int. J. Life Cycle Assess. 20 (6), 842–853.

Restum, Y., 2018. Obra para a Copa de 2014, Transcarioca sofre com depredação e insegurança. G1 Rio. https://g1.globo.com/rj/rio-de-janeiro/noticia/obra-para-a-copa-de-2014-transcarioca-sofre-com-depredacao-e-inseguranca.ghtml.

Rogers, M., Bruen, M., 1998. Choosing realistic values of indifference, preference and veto thresholds for use with environmental criteria within ELECTRE. Theory Methodol. 107 (3), 542–551.

Rogers, M., et al., 2000. ELECTRE and Decision Support: Methods and Applications in Engineering and Infrastructure Investment. Springer Science+Business Media, New York. https://doi.org/10.1007/978-1-4757-5057-7.

Rowley, H.V., et al., 2012. Aggregating sustainability indicators: beyond the weighted sum. J. Environ. Manag. 111, 24–33.

Roy, B., Bouyssou, D., 1993. Aide Multicritère à La Décision: Méthodes et Cas. Economica, Paris.

Saaty, T.L., 1980. The Analytic Hierarchy Process. McGraw-Hill, New York.

Sala, S., Farioli, F., Zamagni, A., 2012. Progress in sustainability science: lessons learnt from current methodologies for sustainability assessment: part 1. Int. J. Life Cycle Assess. 18 (9), 1653–1672.

Santoyo-Castelazo, E., Azapagic, A., 2014. Sustainability assessment of energy systems: integrating environmental, economic and social aspects. J. Clean. Prod. 80, 119–138.

Sedláková, A., Vilcekova, S., Burdova, E.K., 2014. Evaluation of structures design concept of lower structure from embodied energy and emissions. Chem. Eng. Trans, 139–144.

Sedláková, A., Vilčeková, S., Burdová, E.K., 2015. Analysis of material solutions for design of construction details of foundation, wall and floor for energy and environmental impacts. Clean Techn. Environ. Policy 17 (5), 1323–1332.

SMAC, 2012. Municipal Plan of Solid Waste Management of Rio de Janeiro City. Municipal Secretariat of Environment, Rio de Janeiro, RJ, Brazil. http://www.rio.rj.gov.br/web/smac/residuos-solidos.

Sparrevik, M., et al., 2012. Use of stochastic multi-criteria decision analysis to support sustainable management of contaminated sediments. Environ. Sci. Technol. 46 (3), 1326–1334.

Su, J.-P., Hung, M.-L., Chao, C.-W., Ma, H.-w., 2010. Applying multi-criteria decision-making to improve the waste reduction policy in Taiwan. Waste Manag. Res. 28 (1), 20–28. https://doi.org/10.1177/0734242X09103839.

Tarne, P., Lehmann, A., Finkbeiner, M., 2019. Introducing weights to life cycle sustainability assessment—how do decision-makers weight sustainability dimensions? Int. J. Life Cycle Assess. 24 (3), 530–542.

Thies, C., et al., 2019. Operations research for sustainability assessment of products: a review. Eur. J. Oper. Res. 274 (1), 1–21.
Vaidya, O.S., Kumar, S., 2006. Analytic hierarchy process: an overview of applications. Eur. J. Oper. Res. 169 (1), 1–29.
von Doderer, C., Kleynhans, T.E., 2014. Determining the most sustainable lignocellulosic bioenergy system following a case study approach. Biomass Bioenergy 70, 273–286.
Zimmermann, H., 2001. Fuzzy Set Theory and Its Applications, fourth ed. Springer Science+Business Media, LLC. https://cours.etsmtl.ca/sys843/REFS/Books/ZimmermannFuzzySetTheory2001.pdf.

Further reading

Azapagic, A., et al., 2013. An integrated approach to assessing the environmental and health impacts of pollution in the urban environment: methodology and a case study. Process Saf. Environ. Prot. 91 (6), 508–520.

CHAPTER

13

Multi-criteria decision-making after life cycle sustainability assessment under hybrid information

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13.1 Introduction

Life cycle assessment (LCA), also called "environmental life cycle assessment" regulated by ISO 14040 standard has been recognized as one of the most powerful tools for assessing the environmental performance of a product or process in life cycle perspective from the extraction of raw materials to the end of the product (Saad et al., 2011). However, as a tool, LCA which is environmental-centric cannot incorporate the economic and social performances. Life cycle costing (LCC), which refers to an economic performance technique, can encompass all associated costs of a product in its whole life cycle (Sherif and Kolarik, 1981). Social life cycle assessment (SLCA), is a social assessment technique to assess the social performance of products and the potential impacts, including both positive and negative, in their life cycle (UNEP, 2009). Similarly, life cycle costing (LCC) and social life cycle assessment (SLCA) can only investigate the economic and the social pillar of sustainability, respectively. Life cycle sustainability assessment (LCSA), which combines LCA, LCC, and SCLA, can assess the economic, environmental, and social aspects of products or processes (Guinée, 2016). Therefore, life cycle sustainability assessment has been widely used for sustainability assessment of energy and industrial systems recently for its advantage of incorporating economic, environmental, and social dimensions of sustainability simultaneously.

LCSA can be employed to compare the relative performances of different energy and industrial systems with respect to the indicators in economic, environmental, and social

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aspects. However, the results of LCSA cannot answer one of the most common questions of the users: which is the most sustainable scenario among these alternatives? This is because the users must face a set of conflict criteria when selecting the most sustainable energy and industrial system among various alternatives. Accordingly, LCSA is combined with multi-criteria decision analysis, also called "multi-criteria decision-making," for ranking the alternative energy and industrial systems according to their sustainability performances. For instance, Ren et al. (2015b) combined LCSA with AHP (analytic hierarchy process) and VIšekriterijumsko KOmpromisno Rangiranje (VIKOR) for sustainability prioritization of three bioethanol production pathways (corn-based, wheat-based, and cassava-based). Xu et al. (2017) combined LCSA with the vector-based three-dimensional algorithm and AHP for ranking three alternative ammonia production processes.

All these studies ranked the energy and industrial systems based on the condition that all the data with respect to the evaluation criteria are crisp numbers (fuzzy numbers were transformed into crisp numbers). In addition, many methods for achieving life cycle sustainability ranking under uncertainties were developed. Ren et al. (2017a, b) developed an improved weighting method and an extended extension theory for ranking energy and industrial systems under uncertainties. Ren (2018a) employed the fuzzy two-stage logarithmic goal programming method and the interval grey relational analysis method for determining the sustainability sequence of four electricity generation systems. Ren et al. (2018) developed an interval best-worst method for determining the weights of the criteria for sustainability assessment based on the opinions of multiple stakeholders and developed an interval multi-criteria decision-making method for sustainability ranking of industrial systems, which address the decision-making matrix composed by using interval numbers. Moreover, there also some studies focusing on developing some methods for achieving life cycle sustainability ranking of alternatives when the users do not have the real data of the alternatives with respect to the evaluation criteria and all the data were based on the judgments of the decision-makers/stakeholders.

Manzardo et al. (2012) employed the improved grey relational analysis to select the most sustainable scenario among twelve hydrogen production technologies, and all the data (relative performances) of these technologies with respect to the evaluation criteria were determined based on the judgments of the experts. Onat et al. (2016) employed the intuitionistic fuzzy TOPSIS (technique for order of preference by similarity to ideal solution) to rank seven alternative vehicle technologies. Based on the above-mentioned analysis, there is still a great challenge to be overcome, because LCSA usually involves multiple types of information besides data uncertainty problems, and linguistic variables corresponding to fuzzy numbers were also usually used to describe the relative performances of the alternatives with respect to some "soft" criteria, the data to which cannot be quantified directly. The crisp numbers and the interval numbers are usually used in LCC and LCA for the "hard" criteria, the linguistic variables corresponding to intuitionistic fuzzy numbers are employed to describe the relative performances of the energy and industrial systems with respect to the "soft" criteria.

Besides the introduction section, the remaining parts of this study have been organized as follows: the developing multi-criteria decision-making method under multi-type data condition is developed in Section 13.2; an illustrative case is studied in Section 13.3; sensitivity analysis is carried out in Section 13.4; and finally, this study is concluded in Section 13.5.

13.2 Decision-making under multi-type data condition

The basics of interval number and intuitionistic fuzzy numbers were firstly introduced; then, the multi-criteria decision-making method for life cycle sustainability ranking of energy and industrial systems under hybrid information was developed.

13.2.1 Preliminary of interval numbers and intuitionistic fuzzy numbers

Definition 13.1 Interval numbers (Xu, 2008; Yue, 2011).
Let $x^{\pm}=\left[x^L,x^U\right]=\left\{x\middle|x^L\leq x\leq x^U,x^L\leq x^U,x^L,x^U\in R\right\}$ was defined as an interval number, which varies from x^L to x^U, and is a positive interval number if $0\leq x^L\leq x^U$. $x^{\pm}$ turns into a real number when $x^L=x^U$.

Definition 13.2 Arithmetic operations (Xu, 2008; Yue, 2011).
Let $x^{\pm}=\left[x^L,x^U\right]=\left\{x\middle|0<x^L\leq x\leq x^U,x^L\leq x^U,x^L,x^U\in R\right\}$ and $y^{\pm}=\left[y^L,y^U\right]=\left\{y\middle|0<y^L\leq y\leq y^U,y^L\leq y^U,y^L,y^U\in R\right\}$, and $k>0$, then,

$$k\cdot x^{\pm}=k\left[x^L,x^U\right]=\left[kx^L,kx^U\right] \tag{13.1}$$

$$x^{\pm}+y^{\pm}=\left[x^L,x^U\right]+\left[x^L,x^U\right]=\left[x^L+y^L,x^U+y^U\right] \tag{13.2}$$

$$x^{\pm}\times y^{\pm}=\left[x^L,x^U\right]\times\left[x^L,x^U\right]=\left[x^Ly^L,x^Uy^U\right] \tag{13.3}$$

$$\left(x^{\pm}\right)^k=\left(\left[x^L,x^U\right]\right)^k=\left[\left(x^L\right)^k,\left(x^U\right)^k\right] \tag{13.4}$$

Definition 13.3 (Xu and Da, 2002)
Let $x^{\pm}=\left[x^L,x^U\right]$and $y^{\pm}=\left[y^L,y^U\right]$ be two interval numbers; the possibility that $x^{\pm}\geq y^{\pm}$:

$$P\left(x^{\pm}\geq y^{\pm}\right)=\max\left\{1-\max\left(\frac{y^U-x^L}{L_{x^{\pm}}+L_{y^{\pm}}},0\right),0\right\} \tag{13.5}$$

where $P(x^{\pm}\geq y^{\pm})$ represents the possibility that $x^{\pm}\geq y^{\pm}$, $L_{x^{\pm}}=x^U-x^L$ represents the length of $x^{\pm}=\left[x^Lx^U\right]$, and $L_{y^{\pm}}=y^U-y^L$ represents the length of $y^{\pm}=\left[y^Ly^U\right]$.
In a similar way, the possibility that $y^{\pm}\geq x^{\pm}$ can be determined by Eq. (13.6).

$$P\left(x^{\pm}\geq y^{\pm}\right)=\max\left\{1-\max\left(\frac{y^U-x^L}{L_{x^{\pm}}+L_{y^{\pm}}},0\right),0\right\} \tag{13.6}$$

where $P(y^{\pm}\geq x^{\pm})$ represents the possibility that $y^{\pm}\geq x^{\pm}$.
$P(x^{\pm}\geq y^{\pm})$ satisfies the following:

$$0\leq P\left(x^{\pm}\geq y^{\pm}\right)\leq 1$$

(1) $P(x^{\pm} \geq y^{\pm}) = 1$ if and only if $y^U \leq x^L$;
(2) $P(x^{\pm} \geq y^{\pm}) = 0$ if and only if $x^U \leq y^L$;
(3) $P(x^{\pm} \geq y^{\pm}) = 0.5$ if and only if $x^{\pm} = y^{\pm}$; and

$$P\left(x^{\pm} \geq y^{\pm}\right) + P\left(y^{\pm} \geq x^{\pm}\right) = 1$$

Definition 13.4 Distance between two interval numbers (Xu, 2008).
Let $x^{\pm} = \left[x^L, x^U\right]$ and $y^{\pm} = \left[y^L, y^U\right]$ be two interval numbers; the distance between $x^{\pm}$ and $y^{\pm}$ can be determined by Eq. (13.7).

$$d\left(x^{\pm}, y^{\pm}\right) = \frac{1}{2}\left(\left|x^L - y^L\right| + \left|x^U - y^U\right|\right) \tag{13.7}$$

Where $d(x^{\pm}, y^{\pm})$ represents the distance between $x^{\pm} = \left[x^L, x^U\right]$ and $y^{\pm} = \left[y^L, y^U\right]$.

Definition 13.5 Intuitionistic fuzzy set (Atanassov, 1986; Szmidt and Kacprzyk, 2001).
An intuitionistic fuzzy set A in X was defined by Atanassov (1986); the intuitionistic fuzzy set A on X can be defined as (Atanassov, 1986):

$$A = \{(A, \mu_A(x), \upsilon_A(x)) | \, x \in X\} \tag{13.8}$$

where
$\mu_A(x): X \rightarrow [0,1]$ and $\upsilon_A(x): X \rightarrow [0,1]$ should satisfy:

$$0 \leq \mu_A(x) + \upsilon_A(x) \leq 1 \tag{13.9}$$

for all $x \in X$.
$\mu_A(x): X \rightarrow [0,1]$ and $\upsilon_A(x): X \rightarrow [0,1]$ represent the degree of membership of x to A and that of non-membership of x to A, respectively.
After determining the degree of membership and that of the non-membership, the indeterminacy degree which represents the hesitancy degree of the decision-makers for$x$ to A can be determined, as presented in Eq. (13.10).

$$\pi_A(x) = 1 - \mu_A(x) - \upsilon_A(x), x \in X \tag{13.10}$$

where $\pi_A(x)$ represents the indeterminacy degree of x to X.
The indeterminacy degree $\pi_A(x)$ is different from the degree of membership $\mu_\beta(x)$and the degree of non-membership $\upsilon_\beta(x)$ of x to X; it can be used as a measure of the degree of indeterminacy of x to X. Accordingly, an intuitionistic fuzzy number A can usually be represented by $A = (\mu_A, \upsilon_A, \pi_A)$ which consists of the degree of membership, non-membership, and indeterminacy.

Definition 13.6 Transfer intuitionistic fuzzy set into interval number (Zhou et al., 2005).
Let $A = (\mu_A, \upsilon_A, \pi_A)$ be an intuitionistic fuzzy set, and it can be transferred into an interval number by Eq. (13.11).

$$A^{\pm} = [\mu_A \quad 1 - \upsilon_A] \tag{13.11}$$

Definition 13.7 Addition between intuitionistic fuzzy numbers (Xu and Yager, 2006).
Let $\gamma=(\mu_\gamma,\upsilon_\gamma,\pi_\gamma)$ and $\beta=(\mu_\beta,\upsilon_\beta,\pi_\beta)$ be two intuitionistic fuzzy numbers; the addition operation between these two intuitionistic fuzzy numbers can be determined by Eq. (13.12).

$$A\oplus B=(\mu_A,\upsilon_A,\pi_A)\oplus(\mu_B,\upsilon_B,\pi_B)=(\mu_A+\mu_B-\mu_A\mu_B,\upsilon_A\upsilon_B,1+\mu_A\mu_B-\mu_A-\mu_B-\upsilon_A\upsilon_B) \tag{13.12}$$

$$\overset{n}{\underset{j=1}{\oplus}} A_j=\overset{n}{\underset{j=1}{\oplus}}\left(\mu_{A_j},\upsilon_{A_j},\pi_{A_j}\right)=\left(1-\prod_{j=1}^{n}\left(1-\mu_{A_j}\right),\prod_{j=1}^{n}\upsilon_{A_j},\prod_{j=1}^{n}\left(1-\mu_{A_j}\right)-\prod_{j=1}^{n}\upsilon_{A_j}\right) \tag{13.13}$$

Definition 13.8 Scale multiplication (Xu and Yager, 2006).
Let $A=(\mu_A,\upsilon_A,\pi_A)$ be an intuitionistic fuzzy set and λ be a real number, then,

$$\lambda A=\left(1-(1-\mu_A)^\lambda,(\upsilon_A)^\lambda,(1-\mu_A)^\lambda-(\upsilon_A)^\lambda\right) \tag{13.14}$$

The criteria determined by LCA, LCC, and SLCA can be divided into two types: the so-called soft criteria and hard criteria. The data of the alternative industrial or energy systems with respect to the "hard" criteria can be determined through field survey, simulation, estimation, and calculation, based on the LCA database or software. However, the data with respect to the "soft" criteria usually cannot be quantified or described in a quantitative way. Moreover, the alternative industrial or energy systems usually involve different stakeholders and different stakeholders have different willingness, preferences, and interests. Therefore, it is usually difficult to determine the data of the alternative industrial or energy systems with respect to the "soft" criteria. Therefore, a novel way for determining the data with respect to the "soft" criteria was developed in this study, and it consists of four steps:

Step 1: Determining all the groups of stakeholders. A representative stakeholder will be selected for each group to collect the preferences, opinions and interests of each group. A focus group meeting can be held to determine the relative performances of the alternative industrial or energy systems with respect to the "soft" criteria based on the opinions of each group of stakeholders. The representative stakeholder in each group will work as the coordinator, and a consensus will be achieved in each group.

Step 2: Rate the alternative industrial or energy systems with respect to each "soft" criterion (Zhou et al., 2005). The stakeholders are asked to use the eleven linguistic variables to describe the relative performances of the alternatives with respect to the "soft" criteria, and they are absolutely good (AG), very good (VG), good (G), pretty good (PG), moderately good (MG), medium (M), moderately bad (MB), pretty bad (PB), bad (B), very bad (VB), and absolutely bad (AB). AG, VG, G, PG, MG, M, MB, PB, B, VB, and AB correspond to ten intuitionistic fuzzy numbers, which are (1,0,0), (0.90,0.05,0.05), (0.80,0.10,0.10), (0.70,0.15,0.15), (0.60, 0.20,0.20), (0.50,0.50,0), (0.40,0.40,0.20), (0.30,0.55,0.15), (0.20,0.70,0.10), (0,.10,0.85,0.05), and (0,1.00,0), respectively (Zhou et al., 2005).

Step 3: Determining the data of the alternatives with respect to each "soft" criterion. Assuming that there are a total of K groups of stakeholders, and the kth group of stakeholders use the intuitionistic fuzzy numbers $A_{ij}^k=(\mu_{ij}^k,\upsilon_{ij}^k,\pi_{ij}^k)$ to describe the relative performances of the ith alternative with respect to the jth criterion, which is a "soft" criterion. According to

Eqs. (13.13), (13.14), the average intuitionistic fuzzy score of the ith alternative with respect to the j-th criterion can be determined by Eq. (13.15).

$$x_{ij} = \frac{\sum_{k=1}^{K} A_{ij}^{k}}{K} = \frac{\overset{K}{\underset{k=1}{\oplus}} \left(\mu_{ij}^{k}, \upsilon_{ij}^{k}, \pi_{ij}^{k}\right)}{K} = \left(1 - \prod_{k=1}^{K}\left(1-\mu_{ij}^{k}\right), \prod_{j=1}^{K}\upsilon_{ij}^{k}, \prod_{j=1}^{K}\left(1-\mu_{ij}^{k}\right) - \prod_{j=1}^{K}\upsilon_{ij}^{k}\right) \tag{13.15}$$

where x_{ij} represents the average intuitionistic fuzzy score of the ith alternative with respect to the jth criterion.

Step 4: Transforming the average intuitionistic fuzzy score into the interval number. The average intuitionistic fuzzy score is transformed into the interval number according to Eq. (13.11), as presented in Eq. (13.16).

$$x_{ij}^{\pm} = \left[x_{ij}^{L} \;\; x_{ij}^{U}\right] = \left[1 - \prod_{k=1}^{K}\left(1-\mu_{ij}^{k}\right) \;\; 1 - \prod_{j=1}^{K}\upsilon_{ij}^{k}\right] \tag{13.16}$$

where $x_{ij}^{\pm}$, which is an interval number, represents the average performance of the ith alternative with respect to the jth criterion based on the opinions of the K groups of stakeholders, x_{ij}^{L} and x_{ij}^{U} are the lower and upper bounds of the interval number $x_{ij}^{\pm}$, respectively.

13.2.2 Multi-criteria decision analysis under multi-data condition

The multi-criteria decision analysis model can be described as follows:

(1) There are a total of M alternative industrial or energy systems, and they are $\{S_1, S_2, \ldots, S_M\}$;
(2) There are N criteria in environmental, economic, and social dimensions for sustainability assessment of the alternative industrial or energy systems, and they are $\{C_1, C_2, \ldots, C_N\}$; and
(3) The weights of the N criteria for sustainability assessment are $\{\omega_1, \omega_2, \ldots, \omega_N\}$, and they can represent the relative importance of these criteria in the decision-making process and the preferences of the stakeholders.

The framework of the developed multicriteria decision analysis for life cycle sustainability ranking of energy and industrial systems is presented in Fig. 13.1.

The multi-criteria decision analysis under multi-data condition developed in this study is specified as follows:

Step 1: Determining the decision-making matrix. The decision-making matrix consists of all the alternatives (i.e., alternative energy or industrial systems), the criteria for evaluating or prioritizing the alternatives, and the data of the alternatives with respect to each of the evaluation criteria. As for the data with respect to the "hard" criteria, they can be described by using the real numbers or the interval numbers directly. As for the data with respect to the "soft" criteria, they can be determined by using the eleven linguistic variables; subsequently, these linguistic variables can be transformed into intuitionistic fuzzy numbers; then, these intuitionistic fuzzy numbers can be aggregated and averaged into the average intuitionistic fuzzy scores by Eq. (13.15); and finally, the average intuitionistic fuzzy scores

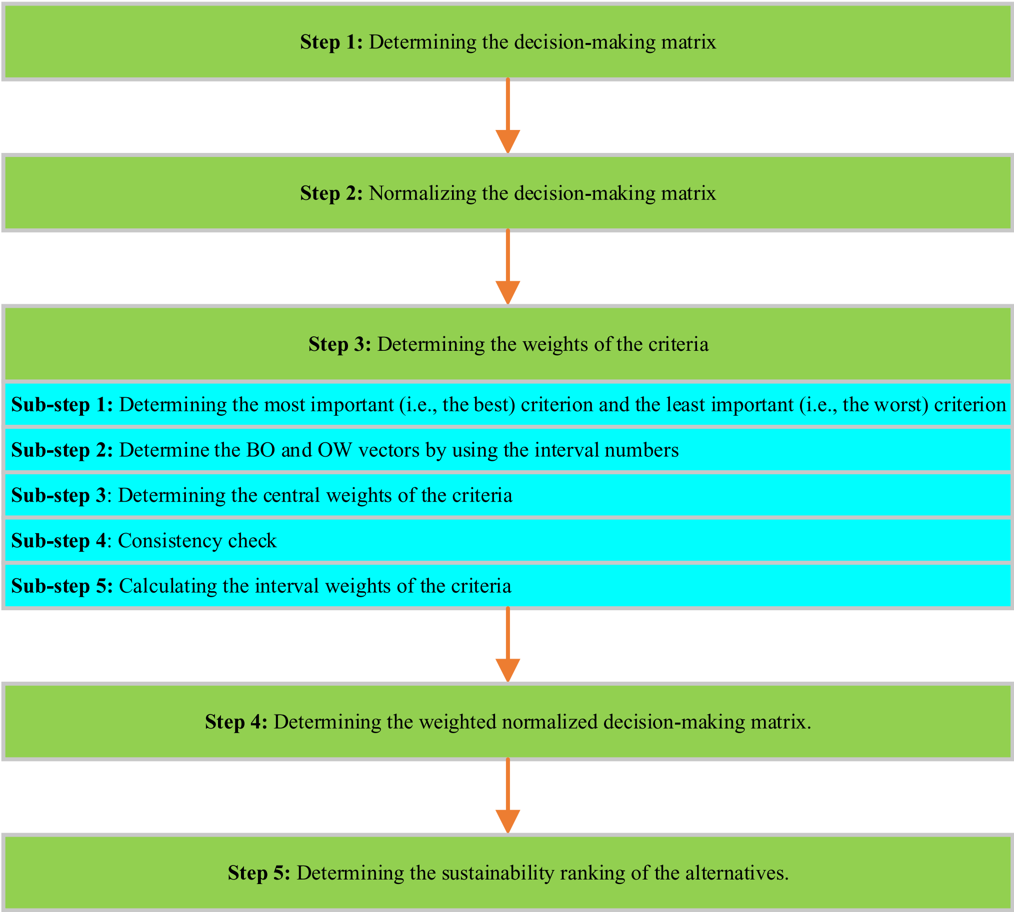

FIG. 13.1 The framework of the developed multi-criteria decision analysis under hybrid information.

can be transformed into interval numbers by Eq. (13.16). After this, the decision-making matrix, which is composed of interval numbers, can be determined, as presented in Eq. (13.17).

$$X = \begin{array}{cccccc} & C_1 & C_2 & \cdots & C_N \\ A_1 & x_{11}^{\pm} & x_{12}^{\pm} & \cdots & x_{1N}^{\pm} \\ A_2 & x_{21}^{\pm} & x_{22}^{\pm} & \cdots & x_{2N}^{\pm} \\ \vdots & \vdots & \vdots & \ddots & \vdots \\ A_M & x_{M1}^{\pm} & x_{M2}^{\pm} & \cdots & x_{MN}^{\pm} \end{array} \tag{13.17}$$

$$x_{ij}^{\pm} = \left[x_{ij}^{L} \;\; x_{ij}^{U} \right] \tag{13.18}$$

where X represents the decision-making matrix, $x_{ij}^{\pm}$, which is an interval number, represents the data of the ith alternative with respect to the jth criterion, and $x_{ij}^{L}$and x_{ij}^{U} are the lower and upper bounds of the interval number $x_{ij}^{\pm}$, respectively.

Step 2: Normalizing the decision-making matrix. The decision-making matrix presented in Eq. (13.17) should be normalized to avoid the influences caused by different dimensions or units. As for the normalization of the data in the decision-making matrix, the data should be normalized according to the types of the criteria. As for the data with respect to the benefit-type criteria:

$$y_{ij}^{\pm} = \left[\frac{x_{ij}^{L}}{\overline{x_j}} \;\; \frac{x_{ij}^{U}}{\overline{x_j}} \right] = \left[y_{ij}^{L} \;\; y_{ij}^{U} \right] \tag{13.19}$$

where $y_{ij}^{\pm}$, which is an interval number, represents the normalized data of the ith alternative with respect to the jth criterion, and y_{ij}^{L} and y_{ij}^{U} represent the lower and upper bounds of the interval number$y_{ij}^{\pm}$, respectively.

$\overline{x_j}$, which represents the average value of the upper bounds of the data with respect to the jth criterion can be determined by Eq. (13.20).

$$\overline{x_j} = \frac{\sum_{i=1}^{M} x_{ij}^{U}}{M} \tag{13.20}$$

As for the data with respect to the cost-type criteria:

$$y_{ij}^{\pm} = \left[\frac{1/x_{ij}^{U}}{\overline{x_j}} \;\; \frac{1/x_{ij}^{L}}{\overline{x_j}} \right] = \left[y_{ij}^{L} \;\; y_{ij}^{U} \right] \tag{13.21}$$

where $y_{ij}^{\pm}$, which is an interval number, represents the normalized data of the ith alternative with respect to the jth criterion, and y_{ij}^{L} and y_{ij}^{U} represent the lower and upper bounds of the interval number$y_{ij}^{\pm}$, respectively.

$\overline{x_j}$ can be determined by Eq. (13.22).

$$\overline{x_j} = \frac{\sum_{i=1}^{M} \frac{1}{x_{ij}^{L}}}{M} \tag{13.22}$$

Then, the normalized decision-making matrix can be determined, as presented in Eq. (13.23).

$$Y = \begin{array}{c} \\ A_1 \\ A_2 \\ \vdots \\ A_M \end{array} \begin{array}{cccc} C_1 & C_2 & \cdots & C_N \\ y_{11}^{\pm} & y_{12}^{\pm} & \cdots & y_{1N}^{\pm} \\ y_{21}^{\pm} & y_{22}^{\pm} & \cdots & y_{2N}^{\pm} \\ \vdots & \vdots & \ddots & \vdots \\ y_{M1}^{\pm} & y_{M2}^{\pm} & \cdots & y_{MN}^{\pm} \end{array} \tag{13.23}$$

where Y represents the normalized decision-making matrix.

Step 3: Determining the weights of the N criteria for sustainability assessment. The interval best-worst method developed by Ren (2018b) is based on the works of Rezaei (2015, 2016) and Entani et al. (2001). It consists of three sub-steps:

Sub-step 1: Determining the most important (i.e., the best) criterion and the least important (i.e., the worst) criterion according to the opinions of the stakeholders in the decision-making process, denoted by C_B and C_W, respectively (Rezaei, 2015, 2016).

Sub-step 2: Determine the BO and OW vectors by using the interval numbers (Ren, 2018a, b).

The comparison method is usually used in many weighting methods, and the numbers from 1 to 9 (corresponding to different linguistic variables, as presented in Table 13.1) and their reciprocals are used to describe the relative importance of one criterion over another.

The single number approach sometimes cannot describe the relative importance of one criterion over another accurately because of the vagueness, hesitations, and ambiguity existing in the minds of the stakeholders. The interval numbers such as [1 3] and [2 4] rather than the single numbers are used to describe the relative importance. Then, the BO and the OW vectors can be determined:

$$BO = \left[a_{B1}^{\pm} \quad a_{B2}^{\pm} \quad \cdots \quad a_{BT}^{\pm}\right] \tag{13.24}$$

$$OW = \left[a_{1W}^{\pm} \quad a_{2W}^{\pm} \quad \cdots \quad a_{TW}^{\pm}\right] \tag{13.25}$$

where $a_{Bj}^{\pm}(j=1,2,\cdots,T)$ and $a_{jW}^{\pm}(j=1,2,\cdots,T)$ represent the relative preference of the most important criterion comparing with the jth criterion and that of the jth criterion comparing with the worst criterion; a_{Bj}^{L} and a_{Bj}^{U} are the upper and lower bounds of $a_{Bj}^{\pm}(j=1,2,\cdots,T)$, respectively; and a_{jW}^{L} and a_{jW}^{U} are the upper and lower bounds of $a_{jW}^{\pm}(j=1,2,\cdots,T)$, respectively.

Sub-step 3: Determining the central weights of the criteria (Rezaei, 2015, 2016). The central weights of the criteria can be determined by solving Eq. (13.26).

$$\begin{aligned}
&\min \xi \\
&s.t. \\
&\left|\frac{\omega_B^C}{\omega_j^C} - \frac{a_{Bj}^{L} + a_{Bj}^{U}}{2}\right| \le \xi, j=1,2,\cdots,n \\
&\left|\frac{\omega_j^C}{\omega_W^C} - \frac{a_{jW}^{L} + a_{jW}^{U}}{2}\right| \le \xi, j=1,2,\cdots,n \\
&\quad \sum_{j=1}^{n} \omega_j^C = 1 \\
&\quad \omega_j^C \ge 0, j=1,2,\cdots,T
\end{aligned} \tag{13.26}$$

where ω_j^C,ω_B^C and ω_W^C represent the central weight of the jth criterion, the central weights of the best criterion and that of the worst criterion, respectively.

TABLE 13.1 Nine-scale in Saaty method (Saaty, 1978).

Scale	Definition	Scale	Definition
1	Equally important	2	Between equally important and moderately important
3	Moderately important	4	Between moderately important and essentially important
5	Essentially important	6	Between essentially important and very strongly important
7	Very Strongly important	8	Between very strongly important and absolutely important
9	Absolutely important		

Sub-step 4: Consistency check (Rezaei, 2015). The consistency ratio can be determined by Eq. (13.27) (Ren, 2018a, b).

$$CR = \frac{\xi^*}{\dfrac{a_{BW}^L + a_{BW}^U + 1 - \sqrt{4a_{BW}^L + 4a_{BW}^U + 1}}{2}} \tag{13.27}$$

where ξ^* is the minimum value of the objective function in programming (13.26), and CR represents the consistency ratio.

The value of CR represents the consistency level of the decision-makers' judgments in determining the BO and OW vectors, and the closer to zero, the more consistent the judgments are.

Sub-step 5: Calculating the interval weights of the criteria. The radius of the weight of each criterion can be determined by solving the programming (13.28).

$$\begin{aligned} &\min \quad \lambda \\ &s.t. \quad \frac{\omega_B^C - d_B}{\omega_j^C + d_j} \le a_{Bj}^L \\ &\frac{\omega_B^C + d_B}{\omega_j^C - d_j} \ge a_{Bj}^U \\ &\frac{\omega_j^C - d}{\omega_W^C + d_W} \le a_{jW}^L \\ &\frac{\omega_j^C + d_j}{\omega_W^C - d_W} \ge a_{jW}^U \\ &d_j \le \lambda \\ &\omega_j^C - d_j \ge 0 \\ &j = 1,2,\cdots,T \end{aligned} \tag{13.28}$$

where d_j, d_B, and d_W represent the radius of the weights of the jth criterion, the best criterion, and the worst criterion, respectively.

After determining the radius of each weight, the interval weight of each criterion can be determined by Eq. (13.29).

$$\omega_j^{\pm} = \left[\omega_j^L \quad \omega_j^U\right] = \left[\omega_j^C - d_j \quad \omega_j^C + d_j\right] \tag{13.29}$$

where $\omega_j^{\pm}$ represents the interval weight of the jth criterion, and ω_j^L and ω_j^U represent the lower and upper bounds of the interval weight of the jth criterion, respectively.

Based on the above-mentioned five sub-steps in Step 3, the weights of the three dimensions of sustainability and the local of the criteria in each dimension can be determined. Then, the global weight of each criterion can be determined by using the local weight of each criterion multiplied with the weight of the corresponding dimension to which it belongs.

Step 4: Determining the weighted normalized decision-making matrix. The weighted normalized decision-making matrix can be determined by Eqs. (13.30), (13.31) after determining the normalized decision-making matrix and the global weights of the criteria.

$$Z=\begin{array}{c} \\ A_1 \\ A_2 \\ \vdots \\ A_M \end{array}\begin{array}{cccc} C_1 & C_2 & \cdots & C_N \\ z_{11}^{\pm} & z_{12}^{\pm} & \cdots & z_{1N}^{\pm} \\ z_{21}^{\pm} & z_{22}^{\pm} & \cdots & z_{2N}^{\pm} \\ \vdots & \vdots & \ddots & \vdots \\ z_{M1}^{\pm} & z_{M2}^{\pm} & \cdots & z_{MN}^{\pm} \end{array} \tag{13.30}$$

$$z_{ij}^{\pm}=\left[z_{ij}^{L}\ \ z_{ij}^{U}\right]=\omega_j^{\pm}y_{ij}^{\pm}=\left[\omega_j^{L}y_{ij}^{L}\ \ \omega_j^{U}y_{ij}^{U}\right] \tag{13.31}$$

Step 5: Determining the sustainability ranking of the alternatives. The ideal solutions with respect to the criteria can be determined by Eq. (13.32).

$$\left\{\max_{i=1}^{M}\left\{z_{i1}^{U}\right\}\ \ \max_{i=1}^{M}\left\{z_{i2}^{U}\right\}\ \cdots\ \max_{i=1}^{M}\left\{z_{iN}^{U}\right\}\right\} \tag{13.32}$$

where $\max_{i=1}^{M}\left\{z_{ij}^{U}\right\}$ $j=1,2,\cdots,N$ represents the best ideal solution with respect to the jth criterion.

This study develops a goal programming model for selecting the best alternative or the most sustainable alternative among multiple choices based on the work of Ren et al. (2015b). The principle of this model is that the best alternative should be the choice that is the closest to the ideal solution. In other words, the best alternative should have the shortest distance to the ideal solution.

The objective function is to minimize the total distance to the ideal solution, as presented in Eq. (13.33).

$$\text{Min}\ \sum_{j=1}^{N}\left(\frac{g_j^{L}+g_j^{U}}{2}\right) \tag{13.33}$$

with the following constraints, including goal constraints, 0-1 constraint, and the selection constraint (Ren et al., 2015a, b).

Goal constraints:

$$\sum_{i=1}^{M}\left[z_{ij}^{L}\ \ z_{ij}^{U}\right]p_i+\left[g_j^{L}\ \ g_j^{U}\right]=\left[\max_{i=1}^{M}\left\{z_{ij}^{U}\right\}\ \ \max_{i=1}^{M}\left\{z_{ij}^{U}\right\}\right]\quad j=1,2,\cdots,T \tag{13.34}$$

The constraints presented in Eq. (13.34) can be further rewritten into:

$$\sum_{i=1}^{M}z_{ij}^{L}p_i+g_j^{L}=\max_{i=1}^{M}\left\{z_{ij}^{U}\right\}\quad j=1,2,\cdots,T \tag{13.35}$$

$$\sum_{i=1}^{M}z_{ij}^{U}p_i+g_j^{U}=\max_{i=1}^{M}\left\{z_{ij}^{U}\right\}\quad j=1,2,\cdots,T \tag{13.36}$$

0-1 constraint:

$$p_i=\begin{cases}1 & \text{if the } i\text{th alternative has been selected as the best alternative}\\ 0 & \text{otherwise}\end{cases} \tag{13.37}$$

Selection constraint:

$$\sum_{i=1}^{M} p_i = 1 \tag{13.38}$$

$p_i = 1$ shows that the ith has been recognized as the best alternative. After determining the best alternative among the M alternatives, the best alternative among the $M-1$ alternatives can also be determined by repeating, according to the programming shown in Eqs. (13.33)–(13.36). With $M-1$ times, the priority sequence of these M alternatives can be determined.

13.3 Case study

In order to illustrate the developed multi-criteria decision analysis method for life cycle sustainability ranking of energy and industrial systems, five electricity generation systems, including electricity generation from coal, oil, biomass, ocean energy, and wind energy, in Mexico (Santoyo-Castelazo, 2011) were studied by the proposed method. Santoyo-Castelazo (2011) carried out a comprehensive sustainability analysis of these alternative electricity generation systems. A total of eight criteria were used to evaluate the sustainability of electricity generation systems: overnight investment costs (EC_1) and levelized costs (EC_2) in the economic dimension, global warming potential (EN_1), acidification potential (EN_2), abiotic depletion potential (EN_3) and eutrophication potential (EN_4) in the environmental dimension, and public acceptability (S_1) and technology maturity (S_2) in the social dimension.

The data of these five electricity generation systems, with respect to these six hard criteria, EC_1, EC_2, EN_1, EN_2, EN_2, and EN_4, were derived from the work of Santoyo-Castelazo (2011). However, the data with respect to the two soft criteria, S_1 and S_2, were evaluated by three groups of stakeholders by using the eleven linguistic variables, and they are researcher and engineer group (DM#1), administration group (DM#2), and user group (DM#3). The performances of these five alternative electricity generation systems by using multiple types of data are presented in Table 13.2. After this, the developed multi-criteria decision analysis method was employed to rank these five alternatives.

TABLE 13.2 The performance of the five alternative electricity generation systems by using multiple types of data.

			Coal	Oil	Biomass	Ocean	Wind
LCC	EC_1	$USD.kW^{-1}$	[602 4671]	1817	[2500 7431]	[3186 6354]	[1223 3716]
	EC_2	$USD.MWh^{-1}$	[33 114]	102	[63 197]	[224 347]	[70 234]
LCA	EN_1	gCO_2-eq.kWh^{-1}	[950 1300]	[40 110]	[17 388]	[8 50]	[8 55]
	EN_2	gSO_2-eq.kWh^{-1}	[0.7 11]	[2 7]	[0.2 0.8]	0.04	[0.05 0.3]
	EN_3	gSb-eq.kWh^{-1}	[5 10]	[3 8]	[0.1 1.1]	0.05	[0.1 0.4]
	EN_4	gPO_4-eq.kWh^{-1}	[0.1 0.6]	[0.05 0.22]	[0.07 0.6]	0.01	[0.01 0.04]
SLCA	S_1	/	VB,VB,B	MB,PB,M	MG,M,MG	PG,G,G	VG,G,VG
	S_2	/	VG,VG,VG	VG,VG,G	G,G,VG	M,MB,MB	MG,M,M

Reference: Santoyo-Castelazo (2011).

Step 1: Determining the decision-making matrix with multi-type data. It is apparent that all the data of these five electricity generation technologies derived from the literature are interval numbers or real numbers. It is very easy to transform the real numbers into interval numbers. For instance, the data of the electricity based on ocean energy with respect to acidification potential (EN_2) is 0.04 gSO_2-eq.kW h^{-1}, and it can be transformed into interval number [0.04 0.04] g SO_2-eq.kW h^{-1}. In a similar way, all the real numbers can be transformed into the format of interval numbers.

As for the relative performances of the five alternative electricity generation systems described by using linguistic variables, these can be transformed into average intuitionistic fuzzy numbers by Eq. (13.15). Then, the average intuitionistic fuzzy numbers can be transformed into interval numbers by Eq. (13.16). Taking the electricity generation system based on coal with respect to public acceptability (S_1) as an example, three linguistic variables, including VB, VB, and B were used, and they correspond to (0,0.10,0.85,0.05), (0,0.10,0.85,0.05), and (0.20,0.70,0.10). The average intuitionistic fuzzy score can be determined, as presented in Eq. (13.39).

$$x_{ij} = \begin{pmatrix} 1-(1-0.1)\times(1-0.1)\times(1-0.2), 0.85\times0.85\times0.70, \\ (1-0.1)\times(1-0.1)\times(1-0.2)-0.85\times0.85\times0.70 \end{pmatrix} \\ = (0.352, 0.50575, 0.14225) \tag{13.39}$$

After this, the average intuitionistic fuzzy score can be transformed into an interval number.

$$x_{ij}^{\pm} = \begin{bmatrix} x_{ij}^{L} & x_{ij}^{U} \end{bmatrix} = [0.352 \quad 1-0.50575] = [0.352 \quad 0.49425] \tag{13.40}$$

In a similar way, all the data of these five electricity generation systems with respect to public acceptability (S_1) and technology maturity (S_2) can be determined (Table 13.3).

Step 2: Normalizing the decision-making matrix. There are two benefit-type criteria (S_1 and S_2) and six cost-type criteria (EC_1, EC_2, EN_1, EN_2, EN_3 and EN_4). The data with respect to these two benefit-type criteria can be normalized by Eqs. (13.19), (13.20), the data with

TABLE 13.3 The performance of the five alternative electricity generation systems by using interval numbers.

			Coal	Oil	Biomass	Ocean	Wind
LCC	EC_1	USD.kW^{-1}	[602 4671]	[1817 1817]	[2500 7431]	[3186 6354]	[1223 3716]
	EC_2	USD.MW h^{-1}	[33 114]	[102 102]	[63 197]	[224 347]	[70 234]
LCA	EN_1	gCO_2-eq.kW h^{-1}	[950 1300]	[40 110]	[17 388]	[8 50]	[8 55]
	EN_2	gSO_2-eq.kW h^{-1}	[0.7 11]	[2 7]	[0.2 0.8]	[0.04 0.04]	[0.05 0.3]
	EN_3	gSb-eq.kW h^{-1}	[5 10]	[3 8]	[0.1 1.1]	[0.05 0.05]	[0.1 0.4]
	EN_4	gPO_4-eq.kW h^{-1}	[0.1 0.6]	[0.05 0.22]	[0.07 0.6]	[0.01 0.01]	[0.01 0.04]
SLCA	S_1	/	[0.352 0.49425]	[0.79 0.89]	[0.92 0.98]	[0.988 0.9985]	[0.998 0.99975]
	S_2	/	[0.999 0.999875]	[0.998 0.99975]	[0.996 0.9995]	[0.82 0.92]	[0.90 0.95]

TABLE 13.4 The normalized decision-making matrix.

	Coal	Oil	Biomass	Ocean	Wind
EC_1	[0.2860 2.2190]	[0.7352 0.7352]	[0.1798 0.5343]	[0.2102 0.4193]	[0.3595 1.0922]
EC_2	[0.5869 2.0275]	[0.6560 0.6560]	[0.3396 1.0620]	[0.1928 0.2987]	[0.2859 0.9558]
EN_1	[0.0115 0.0157]	[0.1357 0.3733]	[0.0385 0.8783]	[0.2986 1.8664]	[0.2715 1.8664]
EN_2	[0.0088 0.1376]	[0.0138 0.0481]	[0.1204 0.4814]	[2.4072 2.4072]	[0.3210 1.9257]
EN_3	[0.0123 0.0247]	[0.0154 0.0411]	[0.1121 1.2336]	[2.4671 2.4671]	[0.3084 1.2336]
EN_4	[0.0341 0.2047]	[0.0930 0.4094]	[0.0341 0.2924]	[2.0468 2.0468]	[0.5117 2.0468]
S_1	[0.4034 0.5665]	[0.9054 1.0201]	[1.0544 1.1232]	[1.1324 1.1444]	[1.1438 1.1458]
S_2	[1.0259 1.0268]	[1.0248 1.0266]	[1.0228 1.0264]	[0.8420 0.9447]	[0.9242 0.9755]

respect to these six cost-type criteria can be normalized by Eqs. (13.21), (13.22). The results are presented in Table 13.4.

Step 3: Determining the weights of the three dimensions of sustainability and the local weights of the criteria in each dimension. Taking the three dimensions of sustainability as an example, the environmental dimension and the social dimension were recognized as the most important and the least important, respectively. The BO and the OW are presented in Table 13.5.

Based on the BO and OW vectors, the following programming model was established for determining the central weights of these three dimensions:

$$\begin{gathered}\min \xi \\ s.t. \\ \left|\frac{\omega_{EN}^C}{\omega_{EC}^C}-\frac{1+3}{2}\right| \leq \xi \\ \left|\frac{\omega_{EN}^C}{\omega_S^C}-\frac{3+6}{2}\right| \leq \xi \\ \left|\frac{\omega_{EC}^C}{\omega_S^C}-\frac{2+3}{2}\right| \leq \xi \\ \omega_{EC}^C+\omega_{EN}^C+\omega_S^C=1 \\ \omega_{EC}^C \geq 0 \\ \omega_N^C \geq 0 \\ \omega_S^C \geq 0\end{gathered} \tag{13.41}$$

TABLE 13.5 The BO and OW vectors for determining the weights of economic, environmental, and social dimensions.

	Economic	Environmental	Social
BO	[1 3]	1	[3 6]
OW	[2 3]	[3 6]	1

TABLE 13.6 The results of programming (Eq. 13.41).

Variables/objectives	ω_{EC}^{C}	ω_{EN}^{C}	ω_{S}^{C}	ξ^*
Values	0.3009	0.5741	0.1250	0.0925
Radius	0.1366	0.1366	0.0231	NA
Interval weights	[0.1643 0.4375]	[0.4375 0.7107]	[0.1019 0.1481]	NA

where ω_{EC}^{C}, ω_{EN}^{C} and ω_{S}^{C} represent the central weights of the economic, environmental, and social dimension, respectively.

The results of programming (Eq. 13.41) are presented in Table 13.6. The consistency ratio can be calculated by Eq. (13.27), and the results are presented in Eq. (13.42). It is apparent that the consistency ratio is zero, which is less than 0.10 (0.10 was set as the threshold value for judging the consistency level; the judgments can be recognized as consistency when the consistency ratio is less than 0.10, or the users need to revise the BO or/and OW vectors until it is less than 0.10).

$$CR = \frac{0.0925}{\frac{3+6+1-\sqrt{4(3+6)+1}}{2}} = 0.0018 < 0.10 \tag{13.42}$$

After determining the central weights of these three dimensions, the radius of the weight of each dimension can be determined according to Eq. (13.28), and the following programming model was established:

$$\begin{aligned}
&\min \lambda \\
&s.t.\ \frac{\omega_{EN}^{C} - d_{EN}}{\omega_{EC}^{C} + d_{EC}} \leq 1 \\
&\frac{\omega_{EN}^{C} + d_{EN}}{\omega_{EC}^{C} - d_{EC}} \geq 3 \\
&\frac{\omega_{EN}^{C} - d_{EN}}{\omega_{S}^{C} + d_{S}} \leq 3 \\
&\frac{\omega_{EN}^{C} + d_{EN}}{\omega_{S}^{C} - d_{S}} \geq 5 \\
&\frac{\omega_{EC}^{C} - d_{EC}}{\omega_{S}^{C} + d_{S}} \leq 1 \\
&\frac{\omega_{EC}^{C} + d_{EC}}{\omega_{S}^{C} - d_{S}} \geq 3 \\
&d_{EC} \leq \lambda \\
&d_{EN} \leq \lambda \\
&d_{S} \leq \lambda \\
&\omega_{EC}^{C} - d_{EC} \geq 0 \\
&\omega_{EN}^{C} - d_{EN} \geq 0 \\
&\omega_{S}^{C} - d_{S} \geq 0
\end{aligned} \tag{13.43}$$

The radiuses of the weights of these three dimensions can be determined after solving programming (Eq. 13.3), and they are 0.1366, 0.1366, and 0.1032, respectively. Then, the interval weights of these three dimensions can be determined according to Eq. (13.29).

In a similar way, all the interval weights of the criteria in each dimension can be determined. The results are presented in Table 13.7.

The global weights of these eight criteria can be determined after the local weights of the criteria in each dimension and the weights of the dimensions. Taking the global weight of the overnight investment costs (EC_1) as an example (Table 13.8):

$$[0.1643\,0.4375] \times [0.3000\,0.5000] = [0.0493\,0.2188] \tag{13.44}$$

Step 4: Determining the weighted normalized decision-making matrix. The data of each cell in the weighted normalized decision-making matrix can be determined by

TABLE 13.7 The local weights of the criteria in each dimension.

EN	**EN_1**	**EN_2**	**EN_3**	**EN_4**
BO	1	1	[2 4]	[5 7]
OW	[5 7]	[5 7]	[1 4]	1
Central weights	0.3964	0.3964	0.1436	0.0635
Radius	0.0291	0.0219	0.0400	0.0400
Interval Weights	[0.3673 0.4255]	[0.3673 0.4255]	[0.1036 0.1836]	[0.0235 0.1035]
$\xi^* = 0.2396, CR = \dfrac{0.2396}{\dfrac{5+7+1-\sqrt{4(5+7)+1}}{2}} = 0.0799 < 0.10$				
EC	EC_1	EC_2		
BO	[1 2]	1		
OW	1	[1 2]		
Central weights	0.4000	0.6000		
Radius	0.1000	0.1000		
Interval weights	[0.3000 0.5000]	[0.5000 0.7000]		
$\xi^* = 0,\ CR = 0 < 0.10$				
S	S_1	S_2		
BO	1	[2 3]		
OW	[2 3]	1		
Central weights	0.7143	0.2857		
Radius	0.0476	0.0476		
Interval weights	[0.6667 0.7619]	[0.2381 0.3333]		
$\xi^* = 0,\ CR = 0 < 0.10$				

TABLE 13.8 The global weights of the eight criteria for sustainability assessment of electricity generation systems

		Local weights	Global weights
Economic ([0.1643 0.4375])	EC_1	[0.3000 0.5000]	[0.0493 0.2188]
	EC_2	[0.5000 0.7000]	[0.0822 0.3062]
	EN_1	[0.3673 0.4255]	[0.1607 0.3024]
Environmental ([0.4375 0.7107])	EN_2	[0.3673 0.4255]	[0.1607 0.3024]
	EN_3	[0.1036 0.1836]	[0.0453 0.1305]
	EN_4	[0.0235 0.1035]	[0.0103 0.0736]
Social ([0.1019 0.1481])	S_1	[0.6667 0.7619]	[0.0679 0.1128]
	S_2	[0.2381 0.3333]	[0.0243 0.0494]

calculating the product of the data with respect to the evaluation criterion in the normalized decision-making and the interval weight of the evaluation criteria. For instance, the data of cell (1,1) in the weighted normalized decision-making matrix can be determined by Eq. (13.45) (using the data of the coal-based electricity generation system with respect to EC_1 to multiply with the global weight of EC_1). In a similar way, all the data in the weighted normalized decision-making matrix can be determined, and the results are presented in Table 13.9. The ideal solutions with respect to each evaluation criterion can be determined.

$$[0.2860\,2.2190] \times [0.0493\,0.2188] = [0.0141\,0.4855] \tag{13.45}$$

Then, the following programming was established to determine the most sustainable electricity generation system:

TABLE 13.9 The weighted normalized decision-making matrix.

	Coal	Oil	Biomass	Ocean	Wind	Ideal solutions
EC_1	[0.0141 0.4855]	[0.0362 0.1609]	[0.0089 0.1169]	[0.0104 0.0917]	[0.0177 0.2390]	0.4855
EC_2	[0.0482 0.6208]	[0.0539 0.2009]	[0.0279 0.3252]	[0.0158 0.0915]	[0.0235 0.2927]	0.6208
EN_1	[0.0018 0.0048]	[0.0218 0.1129]	[0.0062 0.2656]	[0.0480 0.5644]	[0.0436 0.5644]	0.5644
EN_2	[0.0014 0.0416]	[0.0022 0.0146]	[0.0193 0.1456]	[0.3868 0.7279]	[0.0516 0.5823]	0.7279
EN_3	[0.0006 0.0032]	[0.0007 0.0054]	[0.0051 0.1610]	[0.1118 0.3220]	[0.0140 0.1610]	0.3220
EN_4	[0.0004 0.0151]	[0.0010 0.0301]	[0.0004 0.0215]	[0.0211 0.1506]	[0.0053 0.1506]	0.1506
S_1	[0.0274 0.0639]	[0.0615 0.1151]	[0.0716 0.1267]	[0.0769 0.1291]	[0.0777 0.1293]	0.1293
S_2	[0.0249 0.0507]	[0.0249 0.0507]	[0.0249 0.0507]	[0.0205 0.0467]	[0.0225 0.0482]	0.0507

$$
\begin{aligned}
&\text{Min} \quad \sum_{j=1}^{8}\left(\frac{g_j^L + g_j^U}{2}\right) \\
&Z^L P^T + \left(G^L\right)^T = (Z_{\max})^T \\
&Z^U P^T + \left(G^U\right)^T = (Z_{\max})^T \\
&Z^L = \begin{vmatrix} 0.0141 & 0.0362 & 0.0089 & 0.0104 & 0.0177 \\ 0.0482 & 0.0539 & 0.0279 & 0.0158 & 0.0235 \\ 0.0018 & 0.0218 & 0.0062 & 0.0480 & 0.0436 \\ 0.0014 & 0.0022 & 0.0193 & 0.3868 & 0.0516 \\ 0.0006 & 0.0007 & 0.0051 & 0.1118 & 0.0140 \\ 0.0004 & 0.0010 & 0.0004 & 0.0211 & 0.0053 \\ 0.0274 & 0.0615 & 0.0716 & 0.0769 & 0.0777 \\ 0.0249 & 0.0249 & 0.0249 & 0.0205 & 0.0225 \end{vmatrix} \\
&Z^U = \begin{vmatrix} 0.4855 & 0.1609 & 0.1169 & 0.0917 & 0.2390 \\ 0.6208 & 0.2009 & 0.3252 & 0.0915 & 0.2927 \\ 0.0048 & 0.1129 & 0.2656 & 0.5644 & 0.5644 \\ 0.0416 & 0.0146 & 0.1456 & 0.7279 & 0.5823 \\ 0.0032 & 0.0054 & 0.1610 & 0.3220 & 0.1610 \\ 0.0151 & 0.0301 & 0.0215 & 0.1506 & 0.1506 \\ 0.0639 & 0.1151 & 0.1267 & 0.1291 & 0.1293 \\ 0.0507 & 0.0507 & 0.0507 & 0.0467 & 0.0482 \end{vmatrix} \\
&P = |p_1 \ \ p_2 \ \ p_3 \ \ p_4 \ \ p_5| \\
&G^L = |g_1^L \ \ g_2^L \ \ g_3^L \ \ g_4^L \ \ g_5^L \ \ g_6^L \ \ g_7^L \ \ g_8^L| \\
&G^U = |g_1^U \ \ g_2^U \ \ g_3^U \ \ g_4^U \ \ g_5^U \ \ g_6^U \ \ g_7^U \ \ g_8^U| \\
&Z_{\max} = |0.4855\ 0.6208\ 0.5644\ 0.7279\ 0.3220\ 0.1506\ 0.1293\ 0.0507| \\
&p_i = \begin{cases} 1 & \text{if the } i\text{th alternative has been selected as the best alternative} \\ 0 & \text{otherwise} \end{cases}, \quad i = 1,2,\cdots,5 \\
&\sum_{i=1}^{5} p_i = 1
\end{aligned}
\tag{13.46}
$$

The results of the programming model (Eq. 13.46) are presented in Table 13.10.

TABLE 13.10 The results of programming model (Eq. 13.46).

Variables	p_1	p_2	p_3	p_4	p_5
Value	0	0	0	1	0

FIG. 13.2 The sustainability rankings of the five electricity generation systems.

It is apparent that the electricity generation system based on ocean energy was recognized as the most sustainable. In a similar way, the sustainability rankings of these other four alternative electricity generation systems can also be determined, and the results are presented in Fig. 13.2. It is worth pointing out that the ideal solutions with respect to the evaluation criteria may be different in the ranking process when the number of the alternatives changes.

The ranking of ocean-energy-based electricity generation system as the most sustainable is reasonable, because this scenario performs the best in all the four environmental criteria. Meanwhile, it has very good public acceptability. The weights of global warming potential (EN_1), acidification potential (EN_2), abiotic depletion potential (EN_3), and public acceptability (S_1) are also very high; this is also the reason why the ocean-energy-based electricity generation system was recognized as the most sustainable.

13.4 Sensitivity analysis

In order to test the influences of the weights of the evaluation criteria on the sustainability ranking of the five electricity generation systems, the following nine scenarios were studied, and they are:

Scenario 0: using the weights determined by the interval best-worst method;
Scenario 1 (all the eight criteria play an equally important role): 0.1250 was set as the weight for all the eight criteria; and
Scenarios 2–9 (a dominant criterion and seven equally important criteria): 0.3700 was set as the weight of the dominant criterion and 0.0900 was set as the weights of the other seven criteria. For instance, 0.3700 was set as the weight of EC_1 in scenario 2 and 0.0900 was set as the weights of the other seven criteria.

The most sustainable electricity generation system determined under the conditions in the above-mentioned ten scenarios is presented in Table 13.11. It is apparent that ocean-energy-based electricity generation system was recognized as the most sustainable in all the ten scenarios. It means that the ocean-energy-based electricity generation system has a dominant advantage compared with the other four electricity generation systems.

TABLE 13.11 The most sustainable electricity generation system in sensitivity analysis.

Scenarios	**0**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**
The most sustainable	Ocean	Ocean	Ocean	Ocean	Ocean	Ocean	Ocean	Ocean	Ocean	Ocean

13.5 Conclusions

Life cycle sustainability assessment can be used to determine the economic sustainability, environmental sustainability, and social sustainability of different energy and industrial systems. However, it is still difficult for the decision-makers to determine the most sustainable alternative after life cycle sustainability assessment. This study aims to develop a novel multi-criteria decision analysis method for achieving life cycle sustainability ranking of energy and industrial systems under hybrid information, because there are usually multiple types of data after life cycle sustainability assessment. All in all, the developed method in this study has the following advantages:

(1) Linguistic variables corresponding to intuitionistic fuzzy numbers are used to accurately describe the alternative energy and industrial systems with respect to the "soft" criteria, which cannot be quantified directly.
(2) Uncertainties can be addressed by using the interval numbers, and decision-making is achieved under uncertainties.
(3) The ambiguity and hesitations existing in the decision-makers' judgments can be solved by using the interval best-worst method.
(4) The developed method can help the decision-makers to select the most sustainable energy and industrial system among different alternatives using hybrid information.

However, the weighting method used cannot incorporate the preferences and opinions of different decision-makers simultaneously; thus, the weights can only reflect the willingness of a specific group of stakeholders. Meanwhile, all the criteria for sustainability assessment were assumed to be independent; thus, the interdependences among these criteria were not considered in the decision-making. Therefore, the future work of the authors is to develop a multi-criteria decision analysis method that can solve the above-mentioned two problems for life cycle sustainability ranking of energy and industrial systems.

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References

Atanassov, K.T., 1986. IFSs. Fuzzy Sets Syst. 20 (1), 87–96.
Entani, T., Ichihashi, H., Tanaka, H., 2001. Optimistic priority weights with an interval comparison matrix. In: New Frontiers in Artificial Intelligence, pp. 344–348.
Guinée, J., 2016. Life cycle sustainability assessment: what is it and what are its challenges? In: Taking stock of industrial ecology. Springer, Cham, pp. 45–68.
Manzardo, A., Ren, J., Mazzi, A., Scipioni, A., 2012. A grey-based group decision-making methodology for the selection of hydrogen technologies in life cycle sustainability perspective. Int. J. Hydrog. Energy 37 (23), 17663–17670.
Onat, N.C., Gumus, S., Kucukvar, M., Tatari, O., 2016. Application of the TOPSIS and intuitionistic fuzzy set approaches for ranking the life cycle sustainability performance of alternative vehicle technologies. Sustain. Prod. Consumption 6, 12–25.

Ren, J., 2018a. Multi-criteria decision making for the prioritization of energy systems under uncertainties after life cycle sustainability assessment. Sustain. Prod. Consumption 16, 45–57.

Ren, J., 2018b. Selection of sustainable prime mover for combined cooling, heat, and power technologies under uncertainties: an interval multicriteria decision-making approach. Int. J. Energy Res. 42 (8), 2655–2669.

Ren, J., Manzardo, A., Mazzi, A., Zuliani, F., Scipioni, A., 2015a. Prioritization of bioethanol production pathways in China based on life cycle sustainability assessment and multicriteria decision-making. Int. J. Life Cycle Assess. 20 (6), 842–853.

Ren, J., Gao, S., Tan, S., Dong, L., 2015b. Hydrogen economy in China: strengths–weaknesses–opportunities–threats analysis and strategies prioritization. Renew. Sust. Energ. Rev. 41, 1230–1243.

Ren, J., Ren, X., Liang, H., Dong, L., Zhang, L., Luo, X., Yang, Y., Gao, Z., 2017a. Multi-actor multi-criteria sustainability assessment framework for energy and industrial systems in life cycle perspective under uncertainties. Part 1: weighting method. Int. J. Life Cycle Assess. 22 (9), 1397–1405.

Ren, J., Ren, X., Liang, H., Dong, L., Zhang, L., Luo, X., Yang, Y., Gao, Z., 2017b. Multi-actor multi-criteria sustainability assessment framework for energy and industrial systems in life cycle perspective under uncertainties. Part 2: improved extension theory. Int. J. Life Cycle Assess. 22 (9), 1406–1417.

Ren, J., Ren, X., Dong, L., Manzardo, A., He, C., Pan, M., 2018. Multiactor multicriteria decision making for life cycle sustainability assessment under uncertainties. AICHE J. 64 (6), 2103–2112.

Rezaei, J., 2015. Best-worst multi-criteria decision-making method. Omega 53, 49–57.

Rezaei, J., 2016. Best-worst multi-criteria decision-making method: some properties and a linear model. Omega 64, 126–130.

Saad, R., Chayer, J.A., Clément, E., 2011. Guidelines for Environmental Life Cycle Assessment. This file is available at: http://www.eeq.ca/wp-content/uploads/lignesdirectrices_emballages_engl.pdf (Accessed 12 March 2019).

Saaty, T.L., 1978. Exploring the interface between the hierarchies, multiple objectives and the fuzzy sets. Fuzzy Sets Syst. 1, 57–68.

Santoyo-Castelazo, E. (2011). Sustainability Assessment of Electricity Options for Mexico: Current Situation and Future Scenarios. Ph.D. Thesis, University of Manchester. The file is available at: https://core.ac.uk/download/pdf/40016472.pdf (Accessed 12 March 2019).

Sherif, Y.S., Kolarik, W.J., 1981. Life cycle costing: concept and practice. Omega 9 (3), 287–296.

Szmidt, E., Kacprzyk, J., 2001. Entropy for intuitionistic fuzzy sets. Fuzzy Sets Syst. 118 (3), 467–477.

UNEP (United Nations Environment Programme). (2009). Guidelines for Social Life Cycle Assessment of Products. This file is available at: http://www.unep.fr/shared/publications/pdf/dtix1164xpa-guidelines_slca.pdf (Accessed 15 March 2019).

Xu, Z., 2008. Dependent uncertain ordered weighted aggregation operators. Inf. Fusion 9 (2), 310–316.

Xu, Z.S., Da, Q.L., 2002. The uncertain OWA operator. Int. J. Intell. Syst. 17 (6), 569–575.

Xu, Z.S., Yager, R.R., 2006. Some geometric aggregation operators based on intuitionistic fuzzy sets. Int. J. Gen. Syst. 35, 417–433.

Xu, D., Lv, L., Ren, J., Shen, W., Wei, S.A., Dong, L., 2017. Life cycle sustainability assessment of chemical processes: a vector-based three-dimensional algorithm coupled with AHP. Ind. Eng. Chem. Res. 56 (39), 11216–11227.

Yue, Z., 2011. An extended TOPSIS for determining weights of decision makers with interval numbers. Knowl.-Based Syst. 24 (1), 146–153.

Zhou, X., Zhang, Q., Hu, W., 2005. Research on TOPSIS methods based on vague set theory. Syst. Eng. 14 (6), 537–541 (in Chinese).

CHAPTER

14

Life cycle sustainability decision-making framework for the prioritization of electrochemical energy storage under uncertainties

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14.1 Introduction

Nowadays, fossil fuel energy contributes about 70% of electricity generation all over the world, which has caused some issues such as environment worsening and energy shortage (Luo et al., 2015). To tackle this kind of issue, many countries have issued incentive policies and measures to develop renewable energy, which is used to generate electricity. However, the renewable energy such as wind power and solar PV power has the characteristics of intermittency, volatility, and uncertainty, which bring great negative impacts on the stable operation of an electric power system (Ahlborg and Hammar, 2014; Luthra et al., 2015). Energy storage, which can charge and discharge electricity energy, is deemed an important support for renewable energy power deployment in electric power system, because it can charge the redundant electricity from renewable energy and discharge the stored electricity when there is little or even no wind or sunlight (Ren and Ren, 2018).

There are several kinds of energy storage, including mechanical storage, electro-magnetic storage, and electrochemical energy storage (Dunn et al., 2011; Poullikkas, 2013). In the past few years, the electrochemical energy storage, such as lead-acid battery, Li-ion battery and Nas battery, has attracted more and more attention. In China and the USA, electrochemical energy storage has developed rapidly, and 100-MW-level electrochemical energy storage has

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been deployed in the renewable energy-based electric power system. However, how to select the most suitable electrochemical energy storage is a practical and vital issue for electric power operators and planners. Different electrochemical energy storage systems have various characteristics regarding life time, power rating, discharge time, and energy density, and one kind of electrochemical energy storage may have an advantage related to one criterion but may be weak on other criteria. So, the prioritization and selection of electrochemical energy storage can be classified as a multi-criteria decision-making (MCDM) issue, including various conflicting criteria.

Currently, there are many MCDM methods, such as AHP, TOPSIS, matter-element extension model, grey relation decision-making, and best-worst method, some of which have been employed in the prioritization of electrochemical energy storage. Barin et al. (2009) employed the analytic hierarchy process (AHP) and fuzzy logic to assess the operations of different mechanical energy storage systems. Daim et al. (2012) evaluated the priority values of three energy storage technologies by using fuzzy Delphi, AHP, and fuzzy consistent matrix based on experts' judgments. Gumus et al. (2013) developed a new MCDM model based on Buckley extension fuzzy AHP and linear normalization fuzzy grey relation analysis (GRA) for the selection of Turkey's hydrogen storage. Ozkan et al. (2015) conducted the evaluation on electric energy storage based on decision-makers' opinions and judgments by using AHP and type-2 fuzzy TOPSIS MCDM method.

Ren (2018) proposed a novel intuitionistic fuzzy combinative distance-based assessment approach for sustainability prioritization of energy storage technologies, which combines interval AHP for criteria weight determination and intuitionistic fuzzy combinative distance-based assessment method for alternative prioritization. Ren and Ren (2018) performed the sustainability ranking of energy storage technologies under uncertainties using the non-linear fuzzy prioritization and interval MCDM method, and the ranking result was compared with that of interval TOPSIS. Zhao et al. (2018) conducted the comprehensive performance assessment on various battery energy storage systems using a multi-criteria decision-making (MCDM) model, in which a fuzzy-Delphi approach was used to establish the comprehensive assessment indicator system, the entropy weight determination method and the best-worst method (BWM) were used to calculate weights of all sub-criteria, and a Vlsekriterijumska Optimizacija I Kompromisno Resenje (VIKOR) used to choose the optimal battery ESS. Zhao et al. (2019) proposed an integrated MCDM method combining fuzzy-Delphi, best-worst method, and fuzzy cumulative prospect theory for comprehensive assessment on battery energy storage systems considering risk preferences of decision-makers.

The above-mentioned studies have provided valuable tools and methods for the selection of electric energy storage. However, there are still several research gaps, as follows:

(1) The evaluation index system is fragmentary, and mostly focuses on the production stage, not considering the life cycle sustainability view.
(2) The criteria weight determination rarely considers the opinions of different decision-makers, and only considers one integrated decision-maker's judgment.
(3) The prioritization of electrochemical energy storage still lacks consideration of uncertainties from data collection and decision-making processes.

To tackle these issues, this chapter aims at developing a life cycle sustainability decision-making framework for the prioritization of electrochemical energy storage under uncertainties. The evaluation criteria are selected from the perspective of life-cycle sustainability, the criteria weights are determined using group BWM, and the performance of different electrochemical energy storage technologies are ranked using fuzzy TOPSIS considering the uncertainties. Compared with the previous research, the developed life cycle sustainability decision-making framework for the prioritization of electrochemical energy storage under uncertainties has the following advantages:

(1) Criteria system from the perspective of life cycle sustainability: both the quantitative criteria and qualitative criteria in multiple dimensions including economic, environmental, social, and technological aspects based on the life cycle view for sustainability assessment of electrochemical energy storage are determined.
(2) Accurate criteria weight determination: the Bayesian BWM is employed to determine the weights of all the criteria, which can consider the opinions and judgments of multiple decision-makers or stakeholders.
(3) Decision-making under uncertainties: the fuzzy TOPSIS method, which can address uncertainties, is used to assess the performance of electrochemical energy storage technologies, in which the uncertainties of criteria values are representative by triangular fuzzy numbers.

The remainder of this chapter is organized as follows: Section 14.2 presents the criteria system for life cycle sustainability decision-making for the prioritization of electrochemical energy storage under uncertainties; the developed MCDM method for the prioritization of electrochemical energy storage, which combines the Bayesian BWM for criteria weight determination and the fuzzy TOPSIS method for alternative ranking is introduced in Section 14.3; four energy storage technologies, namely lead-acid battery, Li-ion battery, Nas battery, and NiMH battery, are evaluated using the developed MCDM method in Section 14.4; and Section 14.5 concludes the chapter.

14.2 Criteria system for life cycle sustainability decision-making of the prioritization of electrochemical energy storage

The life cycle sustainability decision-making criteria should include three pillars, namely economy, society, and environment, "from cradle to grave," not only the production stage (Ren and Toniolo, 2018). The life cycle sustainability assessment (LCSA) on the prioritization of electrochemical energy storage should integrate life cycle assessment (LCA) for the environmental pillar, life cycle costing (LCC) for the economic pillar, and social life cycle assessment (SLCA) for the social pillar, which can achieve sustainability assessment from the perspective of life cycle (Ren and Toniolo, 2018). For the life cycle sustainability framework of the prioritization of electrochemical energy storage, the criteria of different pillars need to be valued from a life cycle perspective. According to the results of LCC, the criteria values of electrochemical energy storage related to the economic pillar (such as production cost and life cycle cost) can be determined. According to the results of LCA, the criteria of electrochemical energy storage related to the environmental pillar (such as CO2 intensity) can be valued. The SLCA, can be used to determine the criteria value of electrochemical energy storage related to the social pillar (such as social acceptance). Besides the above-mentioned three pillars of sustainability, namely economic criteria, environmental criteria, and social criteria, the technological criteria should be included into the sustainability assessment, especially for the assessment objectives with new technology. The electrochemical energy storage is an emerging matter with advanced technology. Therefore, the life cycle sustainability assessment on the prioritization of electrochemical energy storage includes four pillars, namely economy, society, environment, and technology.

The criteria system is quite important for the life cycle sustainability assessment on the prioritization of electrochemical energy storage; and a vital criteria system can accurately assess the performance of different electrochemical energy storage technologies. However, there is no standard for the criteria system establishment of the prioritization of electrochemical energy storage, because different decision-makers have their own thinking and preferences. In this chapter, a criteria system is built for life cycle sustainability assessment of the prioritization of electrochemical energy storage. According to the related published works and the comments from the expert panel including university professors and electric power system practitioners, the criteria system of life cycle sustainability assessment of the prioritization of electrochemical energy storage is built, which includes eight criteria in four pillars, namely: capital intensity (C1) and operation cost (C2) in the economic pillar; social acceptance (C3) and electric power system reserve capacity reduction (C4) in the social pillar; CO2 intensity (C5) in the environmental pillar; and cycle life (C6), energy efficiency (C7), and self-discharge rate (C8) in the technological pillar. Just as in the above discussion, this criteria system of life cycle sustainability assessment on the prioritization of electrochemical energy storage can be revised according to the opinions and preferences of decision-makers and the actual situations.

The criteria system of life cycle sustainability assessment on the prioritization of electrochemical energy storage is shown in Fig. 14.1. Among eight criteria, there are seven hard criteria (C1, C2, C4, C5, C6, C7, and C8), one soft criteria (C3), four minimum-type criteria (C1, C2, C5, and C8), and four maximum-type criteria (C3, C4, C6, and C7).

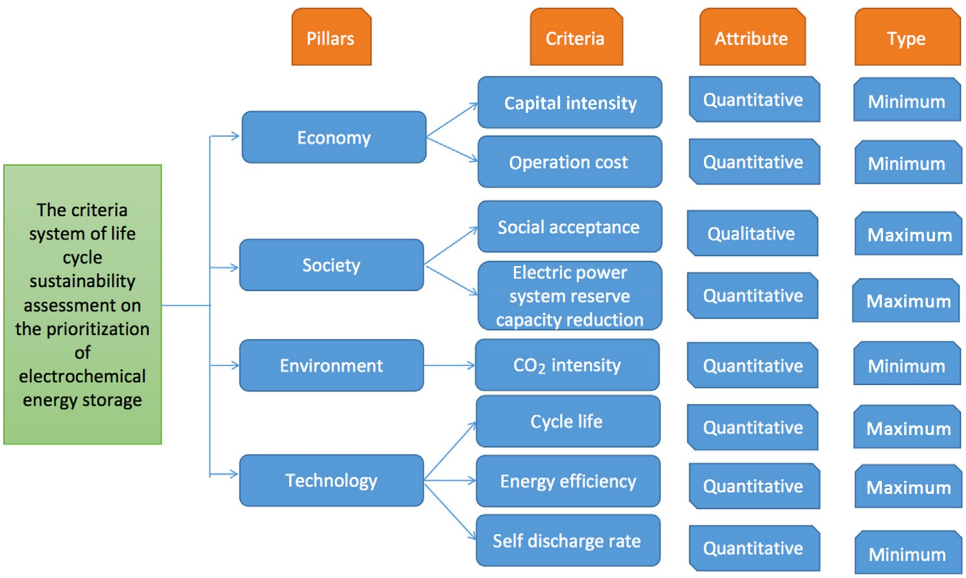

FIG. 14.1 The criteria system of life cycle sustainability assessment on the prioritization of electrochemical energy storage.

14.3 Methods

The framework of life cycle sustainability assessment for the prioritization of electrochemical energy storage is introduced in Section 14.3.1; secondly, the Bayesian BWM method for life cycle sustainability criteria weight determination is presented in Section 14.3.2; finally, the fuzzy TOPSIS method for sustainability ranking of different electrochemical energy storage technologies from the perspective of life cycle is presented.

14.3.1 Life cycle sustainability assessment framework

The life cycle sustainability assessment framework for the prioritization of electrochemical energy storage aims to rank the comprehensive performances of different electrochemical energy storage technologies and then select the best one based on economic, social, environmental, and technological aspects. This framework can be divided into three stages, which are:

Stage 1: Build the criteria system and determine the criteria value. The criteria system needs to be built from the life cycle sustainability perspective. According to Section 14.2, the criteria system of the life cycle sustainability assessment for electrochemical energy storage includes four pillars, namely economy, society, environment, and technology, which consist of eight criteria. The LCA, LCC, and SLCA can be used to collect the criteria value in environmental, economic, and social aspects, respectively. Different from the crisp numbers used by the traditional LCA, LCC, and SLCA, the triangular fuzzy number will be used to value the criteria, which can address the uncertain issue related to criteria value.

Stage 2: Determine the criteria weights for life cycle sustainability assessment. The latest MCDM method, Bayesian best-worst method (BBWM) (Mohammadi and Rezaei, 2019), which can consider the opinions and judgments of multiple decision-makers, is employed to determine the criteria weights of life cycle sustainability assessment for the prioritization of electrochemical energy storage from group MCDM view. Meanwhile, the BWM is more convenient and easy to operate compared with AHP (Rezaei, 2015).

Stage 3: Rank the life cycle sustainability of electrochemical energy storage technologies. The fuzzy TOPSIS method, which can consider the uncertainties of criteria data, is employed to rank and prioritize the life cycle sustainability of electrochemical energy storage.

14.3.2 Criteria weights determination using Bayesian BWM

Best-worst method (BWM) is a new pairwise comparison-based MCDM method (Rezaei, 2015; Rezaei, 2016). It only needs $2n-3$ pairwise comparisons, while the most popular pairwise comparison-based MCDM method, AHP, needs $n(n-1)/2$ pairwise comparisons. For BWM, the decision-makers firstly select the best criteria and the worst criteria, and then compare them with other criteria, not performing pairwise comparisons between any two criteria. This gives a structure to the problem and can help decision-makers provide more reliable pairwise comparisons (Guo and Zhao, 2017). However, the BWM determine the optimal weights of a set of evaluation criteria based on the preference of only decision-maker or stakeholder, which cannot consider the opinions of multiple decision-makers. If there are multiple decision-makers, the BWM will use the average operator (such as arithmetic or geometric mean) to aggregate the preferences of multiple decision-makers and calculate the arithmetic mean of the criteria weights obtained from the individual decision-makers, which has the disadvantages of outlier sensitivity and restricted information provision. Actually, different decision-makers maybe select different best criteria and worst criteria when using BWM to determine the criteria weights. So, the BWM needs to be extended to the group decision-making environment. In 2019, the Bayesian best-worst method (BBWM) was proposed by Mohammadi and Rezaei (2019), which can determine the aggregated criteria weights for a group of decision-makers at once, other than the average operator. The inputs between BWM and BBWM are same, but the outputs of BBWM is the optimal aggregated weights, which can consider the total preferences of all decision-makers or stakeholders along with the confidence level for ranking the criteria.

14.3.2.1 The basic theory and step of BWM

The BWM determine the criteria weights by pairwise comparisons between the best criteria and other criteria as well as between the worst criteria and other criteria conducted by only one decision-maker. The detailed theory and steps of BWM are listed as follows (Guo and Zhao, 2017; Rezaei, 2015; Rezaei, 2016).

Step 1: The decision criteria system needs to be built, which consists of a set of decision criteria. The criteria values can represent the overall performances of different alternatives. Suppose there are n criteria $\{c_1, c_2, \cdots, c_n\}$.

Step 2: The best criterion c_B and the worst criterion c_W are identified by decision-makers in this step. The best criterion is the most desirable or the most important based on the preferences of decision-makers, while the worst criterion is the least desirable or the least important criterion among all the criteria based on decision-makers' preferences.

Step 3: The pairwise comparison between the best criterion and other criteria is conducted in this step. The decision-makers calibrate their preferences of the best criterion to other criteria using a number from one to nine, where one indicates the best criterion is equally important to the compared criterion, and nine means the best criterion is extremely more important to the compared criterion. Based on the pairwise comparisons between the best criterion and other criteria, the "best-to-others" vector A_B can be obtained as:

$$A_B = (a_{B1}, a_{B2}, \cdots, a_{Bn}) \tag{14.1}$$

where a_{Bj} represents the preference of the best criterion to the criterion c_j.

Step 4: The pairwise comparison between the worst criterion and other criteria is conducted in this step. The decision-makers calibrate their preferences of other criteria to the worst criterion using a number from one to nine. Based on the pairwise comparisons between other criteria and the worst criterion, the "others-to-worst" vector $A_W$can be obtained as:

$$A_W = (a_{W1}, a_{W2}, \cdots, a_{Wn}) \tag{14.2}$$

where a_{jW} represents the preference of the criterion c_j over the best criterion.

Step 5: The optimal weights $(w_1^*, w_2^*, \cdots, w_n^*)$of all the criteria can be obtained in this step. According to the rules that the weight vector must be in the neighborhood of the equations

$w_B/w_j = a_{Bj}$ and $w_j/w_W = a_{jW}$, it can minimize the maximum absolute differences $\left|\frac{w_B}{w_j} - a_{Bj}\right|$ and $\left|\frac{w_j}{w_W} - a_{jW}\right|$. Therefore, the following optimization issue can be built.

$$\min_j \max \left\{ \left|\frac{w_B}{w_j} - a_{Bj}\right|, \left|\frac{w_j}{w_W} - a_{jW}\right| \right\} s.t. \begin{cases} \sum_{j=1}^{n} w_j = 1 \\ w_j \geq 0 \\ j = 1, 2, \cdots, n \end{cases} \tag{14.3}$$

Then, the weight vector can also be calculated by the following equation:

$$\min \xi$$
$$s.t. \begin{cases} \left|\dfrac{w_B}{w_j} - a_{Bj}\right| \leq \xi \\ \left|\dfrac{w_j}{w_W} - a_{jW}\right| \leq \xi \\ \sum_{j=1}^{n} w_j = 1 \\ w_j \geq 0 \\ j = 1, 2, \cdots, n \end{cases} \tag{14.4}$$

To check the consistency degree of pairwise comparison for criteria weight determination, the veracity between the pairwise comparisons and their associated weight ratios can be checked using the following consistency ratio (CR):

$$CR = \frac{\xi^*}{CI} \tag{14.5}$$

where ξ^* is the optimal value of ξ, and CI is the consistency index, which is listed in Table 14.1.

14.3.2.2 *The basic theory BBWM*

For BBWM, the inputs and outputs have probabilistic interpretations. The value of criteria indicates the importance of the corresponding criteria. From a probabilistic perspective, the decision criteria can be seen as the random events, and then the decision criteria weights are their occurrence likelihoods. Therefore, all the inputs and outputs need to be modelled as the probability distributions, and the multinomial distribution is employed (Mohammadi and Rezaei, 2019). The probability mass function (PMF) of the multinomial distribution for A_W is:

$$P(A_W|w) = \frac{\left(\sum_{j=1}^{n} a_{jW}\right)!}{\prod_{j=1}^{n} a_{jW}!} \prod_{j=1}^{n} w_j^{a_{jW}} \tag{14.6}$$

TABLE 14.1 CI Table.

a_{BW}	1	2	3	4	5	6	7	8	9
CI	0.00	0.44	1.00	1.63	2.30	3.00	3.73	4.47	5.23

where w is the probability distribution.

Based on the multinomial distribution, the probability of the event j is proportionate to the number of event occurrence to the total number of trials, namely:

$$w_j \alpha \frac{a_{jW}}{\sum_{j=1}^{n} a_{jW}} \tag{14.7}$$

Then, it can be obtained:

$$\frac{w_j}{w_W} \alpha a_{jW} \tag{14.8}$$

Meanwhile, A_B can be modeled using the multinomial distribution, but it is different from A_W because the operation orders of the pairwise comparisons for the best criterion and the worst criterion are reverse. So, there exists:

$$A_B \alpha \text{ multinomial}(1/w) \tag{14.9}$$

where/represents the element-wise division operator.

Similarly,

$$\frac{w_B}{w_j} \alpha a_{Bj} \tag{14.10}$$

Therefore, the criteria weights determination in the BWM is transferred to the estimation of a probability distribution, and the statistical inference techniques can be used to find w in the multinomial distribution.

The maximum likelihood estimation (MLE) is arguably the most popular inference technique that can find the optimal criteria weight vector, for which the Bayesian estimation can be used. In the Bayesian inference, the Dirichlet distribution is employed to model the criteria weights because of the non-negativity and sum-to-one properties of weight vector. However, the MLE inference containing both A_B and A_W does not bear an analytical solution due to the complexity of the corresponding optimization problem, and the simple Dirichlet-multinomial conjugate cannot encompass A_B and A_W together. Thus, a Bayesian hierarchical model is needed.

Assume that there are k decision-makers to assess n criteria using the vectors A_B^k and A_W^k, and w^{agg} is the overall optimal weight. The w^{agg}can be calculated based on the optimal weights of k decision-makers shown by w^k. In the BBWM, $A_B^{1:k}$ and $A_W^{1:k}$ are given, and $w^{1:k}$ and w^{agg}need to be estimated. Therefore, the following joint probability distribution can be sought:

$$P\left(w^{agg}, w^{1:k} \middle| A_B^{1:k}, A_W^{1:k}\right) \tag{14.11}$$

Then, the probability of each individual variable can be computed using the following probability rule:

$$P(x) = \sum_{y} P(x, y) \tag{14.12}$$

where x and y are two arbitrary random variables.

The development of a Bayesian hierarchical model can refer to (Mohammadi and Rezaei, 2019). To compute the posterior distribution of a Bayesian hierarchical model, the Markov-chain Monte Carlo (MCMC) techniques need to be used, and one of the best available probabilistic languages "just another Gibbs sampler" (JAGS) is employed.

For BBWM, the optimization problem of the original BWM is substituted with a probabilistic model, and the above-mentioned model will replace Step 5 of the original BWM. The inputs between the BWM and BBWM are identical, but the output of BBWM can provide more information including the confidence of the relation between each pair of evaluation criteria. Meanwhile, the credal ordering and ranking are also introduced. Interested readers can refer to (Mohammadi and Rezaei, 2019).

14.3.3 Fuzzy TOPSIS method for sustainability ranking of different electrochemical energy storage technologies from the perspective of life cycle

The original TOPSIS method combined with fuzzy set theory, namely fuzzy TOPSIS, is employed to deal with the ambiguity and uncertainty of sustainability ranking of different electrochemical energy storage technologies. The fuzzy TOPSIS conducts the combination of fuzzy set theory and traditional TOPSIS method, which uses triangular fuzzy numbers to represent the value of criteria (Guo and Zhao, 2015).

Fuzzy set theory, proposed by Zadeh, is an extension of the classical set theory (Zadeh, 1965). A fuzzy set $\tilde{a}$ is a pair (U, m) where U is a set and $m: U \to [0,1]$ is the membership function, denoted by $\mu_{\tilde{a}}(x)$. A triangular fuzzy number (TFN) is represented as a triplet $\tilde{a} = \left[a^L, a^M, a^R\right]$, and its membership function $\mu_{\tilde{a}}(x)$ is expressed as:

$$\mu_{\tilde{a}}(x) = \begin{cases} 0 & x < a^L \\ \dfrac{x - a^L}{a^M - a^L} & a^L \le x < a^M \\ \dfrac{a^R - x}{a^R - a^M} & a^M \le x \le a^R \\ 0 & x > a^R \end{cases} \tag{14.13}$$

where a^L, a^M, and a^R are crisp numbers $(-\infty < a^L \le a^M \le a^R < \infty)$; a^L and a^R are the lower and upper bounds of available area for evaluation data, respectively.

To transform the linguistic variables of decision-makers into TFN, the transformation rules between the linguistic terms and fuzzy ratings need to be set first, as listed in Table 14.2 (Liao et al., 2013; Zhao and Guo, 2014).

For fuzzy TOPSIS method, the entry in the decision matrix is represented by TFN, which can characterize the fuzzy and uncertainty issues (Hwang et al., 1993; Wang, 2015). The detailed steps of fuzzy TOPSIS method are introduced as bellow.

Step 1: Calculate the values of the quantitative criteria and the qualitative criteria respective to alternatives. Suppose that there are m alternatives $A = \{A_1, A_2, \cdots A_m\}$ to be ranked. The quantitative criteria of m alternatives can be valued based on the practical calculations and survey. The qualitative criteria can be calculated based on the aggregate fuzzy linguistic ratings for criteria performance of alternatives.

Let $\tilde{a}_{ikj} = \left(a^L_{ikj}, a^M_{ikj}, a^R_{ikj}\right)$, $0 \le a^L_{ikj} \le a^M_{ikj} \le a^R_{ikj} \le 1$, $i = 1, 2, \cdots, m$, $k = 1, 2, \cdots, n$, $j = 1, 2, \cdots, r$ be the superiority linguistic rating on criteria performance assigned to alternative A_i by

TABLE 14.2 Transformation rules of linguistic variables of decision-maker.

Linguistic terms	Membership function
Very low(VL)	(0,0,0.2)
Low (L)	(0,0.2,0.4)
Good (G)	(0.3,0.5,0.7)
High (H)	(0.6,0.8,1)
Very High (VH)	(0.8,1,1)

decision-maker D_j for criteria C_k. Then, the aggregate fuzzy linguistic rating $\tilde{a}_{ik} = \left(a_{ik}^L, a_{ik}^M, a_{ik}^R\right)$ for criteria C_k of alternative A_i can be calculated by:

$$\tilde{a}_{ik} = (1/r) \otimes \left(\tilde{a}_{ik1} \oplus \cdots \oplus \tilde{a}_{ikj} \oplus \cdots \oplus \tilde{a}_{ikr}\right) \tag{14.14}$$

where $a_{ik}^L = \sum_{j=1}^{r} \frac{a_{ikj}^L}{r}$, $a_{ik}^M = \sum_{j=1}^{r} \frac{a_{ikj}^M}{r}$, and $a_{ik}^R = \sum_{j=1}^{r} \frac{a_{ikj}^R}{r}$.

Step 2: Determine the weights of all the criteria. In this chapter, the criteria weights $(w_1, w_2, \cdots, w_n)$of sustainability ranking for different electrochemical energy storage technologies from the perspective of life cycle is determined using the Bayesian best-worst method (BBWM).

Step 3: Build the initial fuzzy decision matrix. The initial fuzzy decision matrix A, as shown in Eq. (14.15), can be obtained based on Step 1, and the entries are given in the form of a triangular fuzzy number.

$$A = \left(\tilde{a}_{ik}\right)_{m \times n} = \begin{bmatrix} \tilde{a}_{11} & \tilde{a}_{12} & \cdots & \tilde{a}_{1n} \\ \tilde{a}_{21} & \tilde{a}_{22} & \cdots & \tilde{a}_{2n} \\ \vdots & \vdots & \vdots & \vdots \\ \tilde{a}_{m1} & \tilde{a}_{m2} & \cdots & \tilde{a}_{mn} \end{bmatrix} = \begin{bmatrix} \left(a_{11}^L, a_{11}^M, a_{11}^R\right) & \left(a_{12}^L, a_{12}^M, a_{12}^R\right) & \cdots & \left(a_{1n}^L, a_{1n}^M, a_{1n}^R\right) \\ \left(a_{21}^L, a_{21}^M, a_{21}^R\right) & \left(a_{22}^L, a_{22}^M, a_{22}^R\right) & \cdots & \left(a_{2n}^L, a_{2n}^M, a_{2n}^R\right) \\ \vdots & \vdots & \vdots & \vdots \\ \left(a_{m1}^L, a_{m1}^M, a_{m1}^R\right) & \left(a_{m2}^L, a_{m2}^M, a_{m2}^R\right) & \cdots & \left(a_{mn}^L, a_{mn}^M, a_{mn}^R\right) \end{bmatrix} \tag{14.15}$$

Step 4: Normalize the initial fuzzy decision matrix. The criteria hold different attributes, including benefit-type attribute (the larger the better) and cost-type attribute (the smaller the better). Therefore, the normalization processing on all criteria need to be performed first.

For benefit-type criteria, the normalization processing is expressed as Eq. (14.16):

$$\tilde{b}_{ik} = \left(a_{ik}^L / t_k, a_{ik}^M / t_k, a_{ik}^R / t_k\right) \tag{14.16}$$

where $t_k = \max_i \left\{a_{ik}^R\right\}$

For cost-type criteria, the normalization processing is expressed as Eq. (14.17):

$$\tilde{b}_{ik} = \left(t_k / a_{ik}^R, t_k / a_{ik}^M, t_k / a_{ik}^L\right) \tag{14.17}$$

where $t_k = \min_i \left\{a_{ik}^L\right\}$.

Then, the normalized fuzzy decision matrix B can be obtained as:

$$B=\left(\tilde{b}_{ik}\right)_{m\times n}=\begin{bmatrix}\tilde{b}_{11} & \tilde{b}_{12} & \cdots & \tilde{b}_{1n}\\ \tilde{b}_{21} & \tilde{b}_{22} & \cdots & \tilde{b}_{2n}\\ \vdots & \vdots & \vdots & \vdots\\ \tilde{b}_{m1} & \tilde{b}_{m2} & \cdots & \tilde{b}_{mn}\end{bmatrix}=\begin{bmatrix}\left(b_{11}^L,b_{11}^M,b_{11}^R\right) & \left(b_{12}^L,b_{12}^M,b_{12}^R\right) & \cdots & \left(b_{1n}^L,b_{1n}^M,b_{1n}^R\right)\\ \left(b_{21}^L,b_{21}^M,b_{21}^R\right) & \left(b_{22}^L,b_{22}^M,b_{22}^R\right) & \cdots & \left(b_{2n}^L,b_{2n}^M,b_{2n}^R\right)\\ \vdots & \vdots & \vdots & \vdots\\ \left(b_{m1}^L,b_{m1}^M,b_{m1}^R\right) & \left(b_{m2}^L,b_{m2}^M,b_{m2}^R\right) & \cdots & \left(b_{mn}^L,b_{mn}^M,b_{mn}^R\right)\end{bmatrix} \tag{14.18}$$

Step 5: Construct the weighted normalized fuzzy decision matrix. The weighted normalized fuzzy decision matrix C is equal to the normalized fuzzy decision matrix B times the criteria weights, as follows:

$$C=(c_{ik})_{m\times n}=\begin{bmatrix}w_1\otimes\tilde{b}_{11} & w_2\otimes\tilde{b}_{12} & \cdots & w_n\otimes\tilde{b}_{1n}\\ w_1\otimes\tilde{b}_{21} & w_2\otimes\tilde{b}_{22} & \cdots & w_n\otimes\tilde{b}_{2n}\\ \vdots & \vdots & \vdots & \vdots\\ w_1\otimes\tilde{b}_{m1} & w_2\otimes\tilde{b}_{m2} & \cdots & w_n\otimes\tilde{b}_{mn}\end{bmatrix}$$

$$=\begin{bmatrix}\left(w_1b_{11}^L,w_1b_{11}^M,w_1b_{11}^R\right) & \left(w_2b_{12}^L,w_2b_{12}^M,w_2b_{12}^R\right) & \cdots & \left(w_nb_{1n}^L,w_nb_{1n}^M,w_nb_{1n}^R\right)\\ \left(w_1b_{21}^L,w_1b_{21}^M,w_1b_{21}^R\right) & \left(w_2b_{22}^L,w_2b_{22}^M,w_2b_{22}^R\right) & \cdots & \left(w_nb_{2n}^L,w_nb_{2n}^M,w_nb_{2n}^R\right)\\ \vdots & \vdots & \vdots & \vdots\\ \left(w_1b_{m1}^L,w_1b_{m1}^M,w_1b_{m1}^R\right) & \left(w_2b_{m2}^L,w_2b_{m2}^M,w_2b_{m2}^R\right) & \cdots & \left(w_nb_{mn}^L,w_nb_{mn}^M,w_nb_{mn}^R\right)\end{bmatrix} \tag{14.19}$$

Step 6: Determine the fuzzy positive and negative ideal solution. Let C^+ and C^- respectively represent the fuzzy positive ideal solution and fuzzy negative ideal solution, which can be computed by:

$$\begin{cases}C^+=\left(\tilde{c}_k^+\right)=\left\{\left(\max\limits_i c_{ik}\,|j\in J_1)\right),\left(\min\limits_i c_{ik}\,|j\in J_2)\right)\right\}\\ C^-=\left(\tilde{c}_k^-\right)=\left\{\left(\min\limits_i c_{ik}\,|j\in J_1)\right),\left(\max\limits_i c_{ik}\,|j\in J_2)\right)\right\}\end{cases} \tag{14.20}$$

where,

$$\max_i c_{ik}=\left(\max_i s_k^Lb_{ik}^L,\ \max_i s_k^Mb_{ik}^M,\ \max_i s_k^Rb_{ik}^R\right);\min_i c_{ik}=\left(\min_i s_k^Lb_{ik}^L,\ \min_i s_k^Mb_{ik}^M,\ \min_i s_k^Rb_{ik}^R\right);$$
$$\tilde{c}_k^+=\left(c_k^{+L},c_k^{+M},c_k^{+R}\right)\tilde{c}_k^-=\left(c_k^{-L},c_k^{-M},c_k^{-R}\right)i=1,2,\cdots,m;\ k=1,2,\cdots,n$$

where J_1 and J_2, respectively, represent the benefit-type criteria set and cost-type criteria set.

Step 7: Calculate the distance of each alternative from fuzzy positive and negative ideal solution. A modified geometrical distance method is employed to calculate the distance between two TFNs, which has the merits of powerful concept and easy implementation. The distance $d\left(\tilde{a}_i,\tilde{a}_j\right)$ between two triangular fuzzy number $\tilde{a}_i$ and $\tilde{a}_j$ can be computed by:

$$d\left(\tilde{a}_i,\tilde{a}_j\right)=\left\{\left[\left(a_i^L-a_j^L\right)^2+2\left(a_i^M-a_j^M\right)^2+\left(a_i^R-a_j^R\right)^2\right]/4\right\}^{1/2} \tag{14.21}$$

Therefore, the distance (d_i^+, d_i^-) of alternative i $(i=1,2,\cdots,m)$ from the fuzzy positive and negative ideal solution can be calculated as follows:

$$d_i^+ = \left\{ \sum_{k=1}^{n} \left\{ \left\{ \left[\left(c_{ik}^L - c_k^{+L}\right)^2 + 2\left(c_{ik}^M - c_k^{+M}\right)^2 + \left(c_{ik}^R - c_k^{+R}\right)^2 \right] / 4 \right\}^{1/2} \right\}^2 \right\}^{1/2} \tag{14.22}$$

$$d_i^- = \left\{ \sum_{k=1}^{n} \left\{ \left\{ \left[\left(c_{ik}^L - c_k^{-L}\right)^2 + 2\left(c_{ik}^M - c_k^{-M}\right)^2 + \left(c_{ik}^R - c_k^{-R}\right)^2 \right] / 4 \right\}^{1/2} \right\}^2 \right\}^{1/2} \tag{14.23}$$

Step 8: Compute the closeness coefficient (CC_i) of each alternative. The closeness coefficient represents the distance farthest from the fuzzy negative ideal solution C^- and closet to the fuzzy positive ideal solution C^+ simultaneously, which can be computed by

$$CC_i = \frac{d_i^-}{d_i^+ + d_i^-}, 0 \le CC_i \le 1 \tag{14.24}$$

Step 9: Rank the alternatives. The alternative with the maximum value of CC_i has the highest ranking score, which should be selected as the optimal alternative.

14.4 Empirical analysis

In order to illustrate the developed life cycle sustainability assessment framework for the prioritization of electrochemical energy storage, four kinds of electrochemical energy storage technologies are selected in this chapter: lead-acid battery (B1), Li-ion battery (B2), Nas battery (B3), and NiMH battery (B4). Eight criteria—namely capital intensity (C1) and operation cost (C2) in the economic pillar; social acceptance (C3) and electric power system reserve capacity reduction (C4) in the social pillar; CO2 intensity (C5) in the environmental pillar; and cycle life (C6), energy efficiency (C7) and self-discharge rate (C8) in the technological pillar—are all employed to assess the life cycle sustainability of lead-acid battery, Li-ion battery, Nas battery, and NiMH battery. Of these, the values of seven criteria, including C1, C2, C4, C5, C6, C7, and C8, can be obtained from the related literatures and field research. The value of criteria C3 cannot be obtained directly, but can be obtained based on the linguistic terms of decision-makers.

The BBWM is employed to determine the weights of all the criteria and the relative performances of the four batteries with respect to the soft criteria social acceptance (C3). Five top-tier experts of energy storage including three professors focusing on energy engineering and two practitioners were invited to give their preferences on different criteria related to the criteria weight and performance.

14.4.1 Criteria weights calculation

Through rounds of questionnaire, the invited experts filled in a form to obtain the basic information for BBWM. These five experts respectively recognize C5, C7, C8, C1, and C7 as the best criterion, and respectively consider C6, C2, C1, C6, and C3 as the worst criterion. Meanwhile, these five experts also give their pairwise comparisons between the best criterion

TABLE 14.3 The pairwise comparisons between the best criterion and other criteria for five experts.

	C1	C2	C3	C4	C5	C6	C7	C8
The best criterion C5 obtained from Expert 1	4	6	5	5	1	7	3	4
The best criterion C7 obtained from Expert 2	5	7	3	3	5	5	1	2
The best criterion C8 obtained from Expert 3	6	6	5	2	4	4	2	1
The best criterion C1 obtained from Expert 4	1	2	3	3	2	4	2	3
The best criterion C7 obtained from Expert 5	3	3	6	4	3	2	1	2

TABLE 14.4 The pairwise comparisons between other criteria and the worst criterion for five experts.

	The worst criterion C6 obtained from Expert 1	The worst criterion C2 obtained from Expert 2	The worst criterion C1 obtained from Expert 3	The worst criterion C6 obtained from Expert 4	The worst criterion C3 obtained from Expert 5
C1	3	2	1	4	4
C2	2	1	2	3	3
C3	2	5	2	2	1
C4	3	4	5	1	3
C5	8	3	3	2	4
C6	1	2	3	1	3
C7	5	7	5	2	6
C8	4	5	7	2	5

and other criteria, which are tabulated in Table 14.3, and their pairwise comparisons between other criteria and the worst criterion, which are listed in Table 14.4.

Therefore, we can obtain the "best-to-others" vector A_B, namely:

$$A_B = \begin{pmatrix} 4 & 6 & 5 & 5 & 1 & 7 & 3 & 4 \\ 5 & 7 & 3 & 3 & 5 & 5 & 1 & 2 \\ 6 & 6 & 5 & 2 & 4 & 4 & 2 & 1 \\ 1 & 2 & 3 & 3 & 2 & 4 & 2 & 3 \\ 3 & 3 & 6 & 4 & 3 & 2 & 1 & 2 \end{pmatrix}$$

We can also obtain the "others-to-worst" vector A_W, namely:

$$A_W = \begin{pmatrix} 3 & 2 & 1 & 4 & 4 \\ 2 & 1 & 2 & 3 & 3 \\ 2 & 5 & 2 & 2 & 1 \\ 3 & 4 & 5 & 1 & 3 \\ 8 & 3 & 3 & 2 & 4 \\ 1 & 2 & 3 & 1 & 3 \\ 5 & 7 & 5 & 2 & 6 \\ 4 & 5 & 7 & 2 & 5 \end{pmatrix}$$

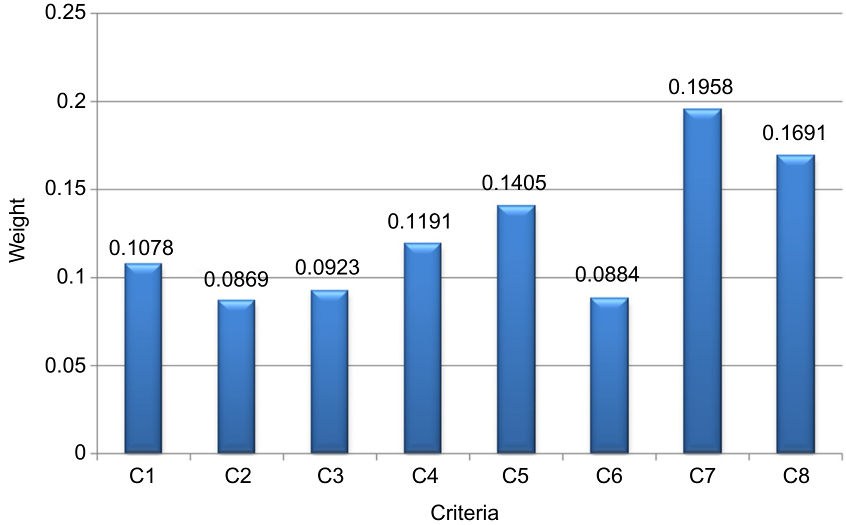

FIG. 14.2 The optimal weights values of eight criteria obtained from BBWM.

Thus, according to the theory of Bayesian BWM introduced in Section 14.3.2, the averages of the Dirichlet distribution of w^{agg} can be calculated by using Matlab software, giving the final optimal weights values of eight criteria, which are listed in Fig. 14.2. It should be mentioned that BBWM obtains the aggregated distribution and all the individual preferences at once by using probabilistic modeling, which are valid and make perfect sense.

Fig. 14.3 displays the credal ranking of eight criteria for life cycle sustainability assessment of electrochemical energy storage. It can be seen that energy efficiency (C7) is the most important criterion based on the preferences of the invited five experts, followed by self-discharge rate (C8), CO2 intensity (C5), electric power system reserve capacity reduction (C4), capital intensity (C1), social acceptance (C3), cycle life (C6), and the operation cost (C2) is the least desirable criteria. Meanwhile, the degree of certainty about the relation of eight criteria can also be evident from Fig. 14.3. For example, the energy efficiency (C7) is certainly more important than operation cost (C2), but it is more desirable than self-discharge rate (C8) with the confidence of 0.71.

14.4.2 Life cycle sustainability ranking for electrochemical energy storage technologies

Five experts gave their linguistic ratings judgments for the performance of soft criteria "social acceptance (C3)" of lead-acid battery, Li-ion battery, Nas battery, and NiMH battery, which are listed in Table 14.5.

According to Table 14.5 and the values of all the criteria expect C3 represented by TFN with the consideration of uncertainties based on the references (Zhao et al., 2018; Zhao et al., 2019), the initial fuzzy decision matrix A can be obtained in the standardized value, as follows.

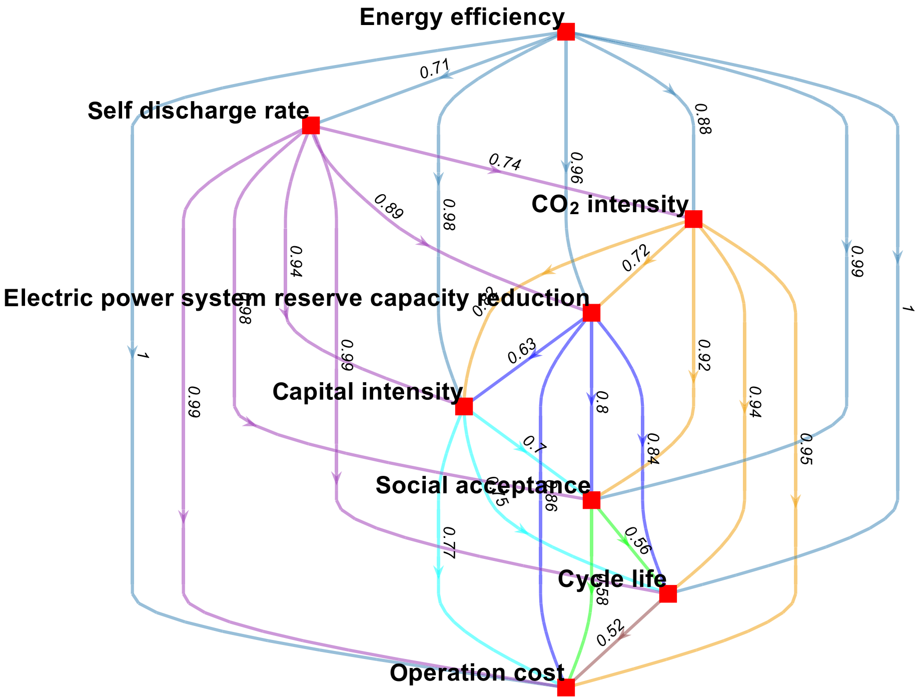

FIG. 14.3 The credal ranking of eight criteria for life cycle sustainability assessment of electrochemical energy storage.

TABLE 14.5 Linguistic ratings for social acceptance criteria performance of four energy storage batteries.

	lead-acid battery	Li-ion battery	Nas battery	NiMH battery
Expert 1	G	G	G	L
Expert 2	G	H	G	G
Expert 3	H	H	H	G
Expert 4	G	G	L	L
Expert 5	L	G	L	L

$$A=\begin{bmatrix} \begin{array}{c} C1 \\ \begin{pmatrix} 0.90 & 0.95 & 1.00 \\ 0.35 & 0.40 & 0.45 \\ 0.55 & 0.60 & 0.65 \\ 0.15 & 0.20 & 0.25 \end{pmatrix} \end{array} & \begin{array}{c} C2 \\ \begin{pmatrix} 0.90 & 0.95 & 1.00 \\ 0.32 & 0.40 & 0.48 \\ 0.52 & 0.60 & 0.68 \\ 0.12 & 0.20 & 0.28 \end{pmatrix} \end{array} & \begin{array}{c} C3 \\ \begin{pmatrix} 0.3 & 0.5 & 0.7 \\ 0.42 & 0.62 & 0.82 \\ 0.24 & 0.44 & 0.64 \\ 0.12 & 0.32 & 0.52 \end{pmatrix} \end{array} & \begin{array}{c} C4 \\ \begin{pmatrix} 0.22 & 0.27 & 0.32 \\ 0.90 & 0.95 & 1.00 \\ 0.42 & 0.48 & 0.53 \\ 0.00 & 0.05 & 0.10 \end{pmatrix} \end{array} \\ \begin{array}{c} C5 \\ \begin{pmatrix} 0.01 & 0.05 & 0.10 \\ 0.90 & 0.95 & 1.00 \\ 0.30 & 0.34 & 0.38 \\ 0.01 & 0.02 & 0.05 \end{pmatrix} \end{array} & \begin{array}{c} C6 \\ \begin{pmatrix} 0.05 & 0.08 & 0.10 \\ 0.70 & 0.76 & 0.80 \\ 0.12 & 0.16 & 0.20 \\ 0.00 & 0.05 & 0.10 \end{pmatrix} \end{array} & \begin{array}{c} C7 \\ \begin{pmatrix} 0.30 & 0.40 & 0.50 \\ 0.90 & 0.95 & 1.00 \\ 0.75 & 0.80 & 0.85 \\ 0.00 & 0.05 & 0.10 \end{pmatrix} \end{array} & \begin{array}{c} C8 \\ \begin{pmatrix} 0.70 & 0.75 & 0.80 \\ 0.90 & 0.95 & 0.98 \\ 0.92 & 0.95 & 1.00 \\ 0.02 & 0.06 & 0.12 \end{pmatrix} \end{array} \end{bmatrix}$$

Among the eight criteria, four criteria C1, C2, C5, C8, are minimum-type and four criteria C3, C4, C6, C7, are maximum-type. So, the normalized fuzzy decision matrix B can be calculated as:

$$B=\left[\begin{array}{cccc} C1 & C2 & C3 & C4 \\ \begin{pmatrix} 0.15 & 0.16 & 0.17 \\ 0.33 & 0.38 & 0.43 \\ 0.23 & 0.25 & 0.27 \\ 0.60 & 0.75 & 1.00 \end{pmatrix} & \begin{pmatrix} 0.12 & 0.13 & 0.13 \\ 0.25 & 0.30 & 0.38 \\ 0.18 & 0.20 & 0.23 \\ 0.43 & 0.60 & 1.00 \end{pmatrix} & \begin{pmatrix} 0.37 & 0.61 & 0.85 \\ 0.51 & 0.76 & 1.00 \\ 0.29 & 0.54 & 0.78 \\ 0.15 & 0.39 & 0.63 \end{pmatrix} & \begin{pmatrix} 0.22 & 0.27 & 0.32 \\ 0.90 & 0.95 & 1.00 \\ 0.42 & 0.48 & 0.53 \\ 0.00 & 0.05 & 0.10 \end{pmatrix} \\ C5 & C6 & C7 & C8 \\ \begin{pmatrix} 0.10 & 0.20 & 1.00 \\ 0.01 & 0.01 & 0.01 \\ 0.03 & 0.03 & 0.03 \\ 0.20 & 0.50 & 1.00 \end{pmatrix} & \begin{pmatrix} 0.06 & 0.10 & 0.13 \\ 0.88 & 0.95 & 1.00 \\ 0.15 & 0.20 & 0.25 \\ 0.00 & 0.06 & 0.13 \end{pmatrix} & \begin{pmatrix} 0.30 & 0.40 & 0.50 \\ 0.90 & 0.95 & 1.00 \\ 0.75 & 0.80 & 0.85 \\ 0.00 & 0.05 & 0.10 \end{pmatrix} & \begin{pmatrix} 0.03 & 0.03 & 0.03 \\ 0.02 & 0.02 & 0.02 \\ 0.02 & 0.02 & 0.02 \\ 0.17 & 0.33 & 1.00 \end{pmatrix} \end{array}\right]$$

Then, the fuzzy positive ideal solution and fuzzy negative ideal solution can be computed according to Eq. (14.20), and the distances of each battery alternative from fuzzy positive ideal solution and fuzzy negative ideal solution can be calculated according to Eqs. (14.22) and (14.23), i.e.:

$$d_1^+ = 0.2006, d_2^+ = 0.1373, d_3^+ = 0.1712, d_4^+ = 0.2231$$
$$d_1^- = 0.1052, d_2^- = 0.2249, d_3^- = 0.1570, d_4^- = 0.1511$$

Finally, the closeness coefficient (CC_i) of each battery alternative can be calculated according to Eq. (14.24), namely:

$$CC_1 = \frac{d_1^-}{d_1^- + d_1^+} = 0.3441, CC_2 = \frac{d_2^-}{d_2^- + d_2^+} = 0.6209$$
$$CC_3 = \frac{d_3^-}{d_3^- + d_3^+} = 0.4783, CC_4 = \frac{d_4^-}{d_4^- + d_4^+} = 0.4039$$

So, $CC_2 \succ CC_3 \succ CC_4 \succ CC_1$. It can be seen that the life cycle sustainability of Li-ion battery outranks other three kinds of batteries, namely Nas battery, NiMH battery, and lead-acid battery, in descending order, related to life cycle sustainability ranking. Therefore, the prioritization of electrochemical energy storage from the perspective of life cycle sustainability is Li-ion battery.

14.5 Conclusions

This chapter aims at developing a life cycle sustainability assessment framework for the prioritization of electrochemical energy storage under uncertainties, and two MCDMs including Bayesian BWM as well as fuzzy TOPSIS, and life cycle sustainability assessment were combined for ranking the sustainability performance of four kinds of energy storage batteries, namely lead-acid battery, Li-ion battery, Nas battery, and NiMH battery. The LCC, LCA, and SLCA were employed to obtain the values of criteria in economic, environmental, and social pillars, respectively. Meanwhile, considering the novel nature and high technological requirement of energy storage batteries, the technological criteria were also included in the criteria system for life cycle sustainability assessment of electrochemical energy storage. The latest group MCDM—Bayesian Best Worst method—was employed to determine the weights of eight criteria in four aspects, which can consider the preferences of multiple decision-makers or stakeholders and the degree of certainty about the relation of eight criteria judged by decision-makers. The fuzzy TOPSIS method was used to rank the life cycle sustainability performances of lead-acid battery, Li-ion battery, Nas battery, and NiMH battery, which can take the uncertainties of criteria value into consideration.

The proposed framework for life cycle sustainability assessment for the prioritization of electrochemical energy storage under uncertainties holds the following merits:

(1) Life cycle sustainability thinking has been considered in the prioritization of electrochemical energy storage, in which the criteria system is built from the perspective of sustainability and the criteria values are based on LCA, LCC, and SLCA.
(2) The Bayesian group BWM has been used to determine the criteria weights, which can consider the preferences of multiple decision-makers in probabilistic modeling view other than the aggregation of individual priorities or judgments and calculate the degree of certainty about the relation of criteria.
(3) The fuzzy TOPSIS method has been used to rank the life cycle sustainability performance of electrochemical energy storage technologies based on the decision-making matrix built by TFN for tackling the uncertainty issue.

Although there are several merits in this chapter, there are still drawbacks. For example, the sensitivity analysis should be performed to verify the ranking robustness of the proposed method for life cycle sustainability assessment of electrochemical energy storage. Meanwhile, other MCDM methods need to be compared with this proposed hybrid method in this chapter in terms of ranking result. These drawbacks will be tacked one by one in the following research.

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References

Ahlborg, H., Hammar, L., 2014. Drivers and barriers to rural electrification in Tanzania and MozambiqueeGrid-extension, off-grid, and renewable energy technologies. Renew. Energy 61, 117–124.

Barin, A., Canha, L.N., Abaide, A.D.R., et al., 2009. Storage energy management with power quality concerns the analytic hierarchy process and the fuzzy logic. In: Power Electronics Conference.

Daim, T.U., Li, X., Kim, J., et al., 2012. Evaluation of energy storage technologies for integration with renewable electricity: quantifying expert opinions. Environ. Innov. Soc Trans. 3, 29–49.

Dunn, B., Kamath, H., Tarascon, J.M., 2011. Electrical energy storage for the grid: a battery of choices. Science 334 (6058), 928–935.

Gumus, A.T., Yayla, A.Y., Çelik, E., et al., 2013. A combined fuzzy-AHP and fuzzy-GRA methodology for hydrogen energy storage method selection in Turkey. Energies 6 (6), 3017–3032.

Guo, S., Zhao, H., 2015. Optimal site selection of electric vehicle charging station by using fuzzy TOPSIS based on sustainability perspective. Appl. Energy 158, 390–402.

Guo, S., Zhao, H., 2017. Fuzzy best-worst multi-criteria decision-making method and its applications. Knowl.-Based Syst. 121, 23–31.

Hwang, C.L., Lai, Y.J., Liu, T.Y., 1993. A new approach for multiple-objective decison-making. Comput. Oper. Res. 20, 889–899.

Liao, M.-S., Liang, G.-S., Chen, C.-Y., 2013. Fuzzy grey relation method for multiple criteria decision-making problems. Qual. Quant. 47, 3065–3077.

Luo, X., Wang, J., Dooner, M., et al., 2015. Overview of current development in electrical energy storage technologies and the application potential in power system operation. Appl. Energy 137, 511–536.

Luthra, S., Kumar, S., Garg, D., et al., 2015. Barriers to renewable/sustainable energy technologies adoption: Indian perspective. Renew. Sust. Energ. Rev. 41, 762–776.

Mohammadi, M., Rezaei, J., 2019. Bayesian best-worst method: a probabilistic group decision making model. Omega. https://doi.org/10.1016/j.omega.2019.06.001.

Ozkan, B., Kaya, I., Cebeci, U., et al., 2015. A hybrid multicriteria decision making methodology based on type-2 fuzzy sets for selection among energy storage alternatives. Int. J. Comput. Intell. Syst. 8 (5), 914–927.

Poullikkas, A., 2013. A comparative overview of large-scale battery systems for electricity storage. Renew. Sust. Energ. Rev. 27, 778–788.

Ren, J., 2018. Sustainability prioritization of energy storage technologies for promoting the development of renewable energy: a novel intuitionistic fuzzy combinative distance-based assessment approach. Renew. Energy 121, 666–676.

Ren, J., Ren, X., 2018. Sustainability ranking of energy storage technologies under uncertainties. J. Clean. Prod. 170, 1387–1398.

Ren, J., Toniolo, S., 2018. Life cycle sustainability decision-support framework for ranking of hydrogen production pathways under uncertainties: an interval multi-criteria decision making approach. J. Clean. Prod. 175, 222–236.

Rezaei, J., 2015. Best-worst multi-criteria decision-making method. Omega 53, 49–57.

Rezaei, J., 2016. Best-worst multi-criteria decision-making method: some properties and a linear model. Omega 64, 126–130.

Wang, E., 2015. Benchmarking whole-building energy performance with multi-criteria technique for order preference by similarity to ideal solution using a selective objective-weighting approach. Appl. Energy 146, 92–103.

Zadeh, L.A., 1965. Fuzzy sets. Inf. Control. 8, 338–353.

Zhao, H., Guo, S., 2014. Selecting green supplier of thermal power equipment by using a hybrid MCDM method for sustainability. Sustainability 6, 217–235.

Zhao, H., Guo, S., Zhao, H., 2018. Comprehensive performance assessment on various battery energy storage systems. Energies. 11 (10), 2841.

Zhao, H., Guo, S., Zhao, H., 2019. Comprehensive assessment for battery energy storage systems based on fuzzy-MCDM considering risk preferences. Energy 168, 450–461.

CHAPTER

15

Barriers identification and prioritization for sustainability enhancement: Promoting the sustainable development of the desalination industry

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15.1 Introduction

Water, like energy, is a basic element for the life on the earth and a key driving force for the development of the modern economy (Shatat et al., 2013). Many developing countries are currently facing serious water shortage crises with the rapid population growth and the industrialization (Gude et al., 2010). Freshwater for drinking and irrigation has also become a serious problem for some developed countries in North Africa, the Middle East, and South Asia (Morillo et al., 2014). It was estimated that two thirds of the population in the world would face insufficient water supply crisis by 2025 (UNEP, 2012). Seawater desalination processes such as reverse osmosis (RO), multiple effect distillation (MED), and multistage flash (MSF) have been recognized as promising ways for solving this problem by generating freshwater for drinking and irrigation from seawater (Mezher et al., 2011). However, the desalination industry in many countries, including China, still faces many severe barriers that hinder the sustainable development.

Höpner and Windelberg (1997) pointed out that there are various potential environmental impacts in desalination plants such as the impacts on land-use change, negative impacts on

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ecosystem, and the emissions to the atmosphere and the sea. The seawater desalination technologies usually consume some fossil fuels, and this will lead to various negative environmental impacts, it will also consume some water even if it is a renewable-energy-based technique (Semiat, 2008). The expensiveness (i.e., high capital cost and high operations cost) is the most severe barrier influencing the renewable-energy-based desalination techniques such as solar powered water desalination technology (Fiorenza et al., 2003; Gude et al., 2010). Moreover, there are also some challenges when adopting solar, wind, and some other forms of renewable energy sources to ensure energy sustainability (Logan, 2017).

As for the sustainability issues of desalination, Balfaqih et al. (2017) developed indicators and metrics for measuring the economic and environmental performance of the desalination supply chain, and analytic hierarchy process method was employed to determine the weight of each indicator/criterion based on their significance in the desalination supply chain. Shahabi et al. (2015) employed life cycle assessment and levelized cost to evaluate the environmental impacts and economic performance of water supply from decentralized and centralized desalination systems. Besides the barriers in economic and environmental aspects, there are also various barriers in technological and social-managerial aspects (Lior, 2017). There are two knowledge gaps when determining the corresponding effective for solving these barriers: one is that the stakeholders do not know the relative importance of these barriers with respect to their influences on sustainable development of desalination industry; the other is that the complex cause-effect relationships among these barriers are unclear to the decision-makers. In order to solve the above-mentioned two problems, this study aims to develop a novel analytic method for investigating the complex cause-effect relationships among these barriers that hinder the sustainable development of the desalination industry, and the desalination industry in China was illustrated and studied by the developed method.

Besides the introduction section, the residual parts of this study has been organized as follows: Section 15.2 summarizes the barriers of sustainable desalination industry in China, Section 15.3 presents the developed DEMATEL method, Section 15.4 presents the results and discussions, Section 15.5 presents the policy implications, and finally, this study is concluded in Section 15.6.

15.2 Barriers to sustainable desalination industry in China

A total of 12 key barriers that hinder the sustainable development of the desalination industry in China were identified, based on a comprehensive literature and focus group meeting, and these barriers were categorized into four aspects including technological aspect, economic aspect, environmental aspect, and economic aspect.

(1) Technological aspect: the barriers in the technological aspect consist of the lack of key technologies, the localization degree of desalination facilities, high energy consumption, and the lack of technical innovation. Therefore, the following two barriers were identified in the technological aspect.

I. Lack of advanced technologies (T_1): there is a big gap between the desalination technologies in China and the world-class advanced desalination technologies, the localization degree of desalination facilities is not low, although the desalination industry has already developed for more than 30 years in China (Yang, 2014).

II. High energy consumption (T_2): seawater desalination is an energy-intensive industry, and the traditional desalination technologies usually require steady and reliable energy sources, but the current renewable clean energy sources are usually intermittent and unstable (Wang et al., 2014).

(2) Economic aspect: the barriers in the economic aspect mainly refer to high capital cost, high production cost, and the lack of special research and development (R&D) funding for seawater desalination.

I. High capital cost (EC_1): seawater desalination is a capital-intensive industry. For instance, the capital cost of reverse osmosis is 6000–8000 Yuan m^{-3}. Taking the 15,000 m^3 day^{-1} seawater desalination project as an example, the capital costs of multistage flash system, low temperature multieffect distillation, reverse osmosis, and low temperature-vapor compression distillation are 1.70E+08, 1.50E+08, 0.90E+08, and 1.60E+08 Yuan, respectively (Li, 2010).

II. High production cost (EC_2): the high production cost is one of the most important factors hindering the sustainable development of desalination technologies, because the seawater desalination production cost consists of operations and management cost and capital cost. For instance, the production costs of multistage flash system, low temperature multieffect distillation, reverse osmosis, and low temperature-vapor compression distillation are 5.105, 4.825, 4.272, and 5.025 Yuan t^{-1}, respectively (Li, 2010). Therefore, the water from seawater desalination is not competitive compared with the municipal water.

III. Lack of special research and development (R&D) funding (EC_3): the current R&D investment on seawater desalination technologies is not enough; and it lack the public platform for research, development, and innovation (Li et al., 2016).

(3) Environmental aspect: the negative environmental impact of seawater desalination technologies is a barrier hindering sustainable development of the desalination industry with the increase in awareness of the citizens in China on environment protection.

I. Negative environmental impacts (EN_1): the negative environmental impacts are mainly caused by the effluent of the concentrated seawater to the sea, and many chemicals are commonly used in seawater desalination processes that lead to serious problems in marine ecology. In addition, the utilization of fossil fuels leads to the emissions of SO_x, NO_x, and CO_2, etc. (Li et al., 2016).

(4) Social-managerial aspect: the barriers in the social-managerial aspect include the lack of awareness and perception, lack of supporting policies and regulation, and the unclear responsibility in administration.

I. Lack of awareness and perceptions (SM_1): seawater desalination has usually been recognized as an advanced technology with high capital cost, the water from desalination is still in "debate," and the high energy consumption and the potential negative environmental impacts in desalination technologies cause these technologies to have very low social acceptability. Meanwhile, the role of desalination in solving water supply security is not well recognized by society (Li et al., 2016).

II. Lack of supporting policies and regulation (SM_2): there is a lack of incentive policies to stimulate and support the development of the desalination industry in China, especially financial support, including low-interest loans and subsidies. Meanwhile, the regulation and standard system is incomplete, and China has not established a standard system for the use of unconventional water resources. Meanwhile, the standards for seawater desalination technologies have not been included in the current standard system for water production, supply, use, and quality (Li et al., 2016).

III. Unclear responsibility in administration (SM_3): the seawater desalination processes usually involves multiple governmental sectors, i.e., Ministry of Water Resources of the People's Republic of China, National Bureau of Oceanography, and National Development and Reform Commission, etc. However, the responsibilities and administrations are unclear among these governmental sections. The coherence and cohesion between different governmental sectors should be improved (Li et al., 2016).

15.3 Methods

The basics of triangular fuzzy numbers are introduced in Section 15.3.1, then the developed DEMATEL method is presented in Section 15.3.2.

15.3.1 Basics of triangular fuzzy numbers

Definition 15.1 Triangular fuzzy number (Cheng and Lin, 2002)
Assume that $\tilde{a} = \left(a^L, a^M, a^U\right)$ is a triangular fuzzy number, a^L, a^M, and a^U which satisfy $a^L \le a^M \le a^U, a^L, a^M, a^U \in R$, are the three elements of the fuzzy number. The membership function of this fuzzy number $\mu_{\tilde{a}}(x)$: $R \to [0,1]$ is presented in Eq. (15.1).

$$\mu_{\tilde{a}}(x) = \begin{cases} 0 & x < a^L \\ \dfrac{x - a^L}{a^M - a^L} & a^L \le x \le a^M \\ \dfrac{x - a^U}{a^M - a^U} & a^M \le x \le a^U \\ 0 & x > a^U \end{cases} \tag{15.1}$$

where a^L, a^M and a^U are the three elements of the triangular fuzzy number $\tilde{a}$.
The difference of the upper bound (a^U) from the lower bound (a^L) can be as a measure of the fuzzy degree, and the greater the difference the bigger the fuzzy degree will be.

Definition 15.2 Graded mean integration representation value (Chen and Hsieh, 2000)
The graded mean integration representation value of the triangular fuzzy number $\tilde{a} = \left(a^L, a^M, a^U\right)$ can be determined by Eq. (15.2).

$$R\left(\tilde{a}\right) = \frac{a^L + 6a^M + a^U}{6} \tag{15.2}$$

Definition 15.3 Arithmetic operations
The arithmetic operations including addition, subtraction, multiplication, and division between two triangular fuzzy number are specified as follows (Zhao and Guo, 2014; Chen and Ren, 2018; Liang et al., 2019):

Addition

$$\widetilde{a}\oplus\widetilde{b}=\left(a^L,a^M,a^R\right)\oplus\left(b^L,b^M,b^R\right)=\left(a^L+b^L,a^M+b^M,a^R+b^R\right) \tag{15.3}$$

Subtraction

$$\widetilde{a}-\widetilde{b}=\left(a^L,a^M,a^R\right)-\left(b^L,b^M,b^R\right)=\left(a^L-b^L,a^M-b^M,a^R-b^R\right) \tag{15.4}$$

Multiplication

$$\widetilde{a}\otimes\widetilde{b}=\left(a^L,a^M,a^R\right)\otimes\left(b^L,b^M,b^R\right)=\left(a^Lb^L,a^Mb^M,a^Rb^R\right),a^L\geq 0,b^L\geq 0 \tag{15.5}$$

$$\lambda\widetilde{a}=\lambda\left(a^L,a^M,a^U\right)=\left(\lambda a^L,\lambda a^M,\lambda a^U\right),\lambda\geq 0 \tag{15.6}$$

Division

$$\widetilde{a}\phi\widetilde{b}=\left(a^L,a^M,a^R\right)\phi\left(b^L,b^M,b^R\right)=\left(\frac{a^L}{b^R},\frac{a^M}{b^M},\frac{a^R}{b^L}\right),a^L\geq 0,b^L>0 \tag{15.7}$$

$$\lambda\phi\widetilde{b}=\lambda\phi\left(b^L,b^M,b^R\right)=\left(\frac{\lambda}{b^R},\frac{\lambda}{b^M},\frac{\lambda}{b^L}\right),\lambda\geq 0,b^L>0 \tag{15.8}$$

15.3.2 The improved DEMATEL method

The traditional DEMATEL method usually employs five linguistic terms corresponding to the five numbers (0, 1, 2, 3, and 4) to depict the direct influence between each pair of items. However, the use of the discrete numbers to determine the relative influences usually has one severe problem-difficulty in quantifying the relative direct influence: it is difficult to use these five numbers to describe the relative direct influences accurately. For instance, the influence of factor A on factor C is different from that of factor B on factor C; if the direct influence of factor A on factor C can be described by using 1, then the direct influence of factor B on factor C should be described by using 2.1, but it is impossible to achieve this when employing the traditional DEMATEL method. In order to address this, an improved DEMATEL method is developed in this study, and it consists of six steps:

Step 1: Establishing the initial influence matrix.
Step 2: Determining the normalized initial influence matrix.
Step 3: Calculating the total influence matrix.
Step 4: Determining the sum of each row and the sum of each column.
Step 5: Determining the weight of each influential factor.
Step 6: Drawing the IRM (influence relation map).

These six steps are specified as follows.

Step 1: Establishing the initial influence matrix X

Assume that there are N barriers influencing the sustainable development of the desalination industry in China, and they are B_1, B_2, $\cdots$, B_N, the direct influence of the i-th barrier on the j-th barrier is represented by x_{ij}, then, the initial influence matrix X can be determined, as presented in Eq. (15.9).

$$X = \begin{array}{cccccc} & B_1 & B_2 & \cdots & B_N \\ B_1 & x_{11} & x_{12} & \cdots & x_{1N} \\ B_2 & x_{21} & x_{22} & \cdots & x_{2N} \\ \vdots & \vdots & \vdots & \ddots & \vdots \\ B_N & x_{N1} & x_{N2} & \vdots & x_{NN} \end{array} \tag{15.9}$$

where X represents the initial influence matrix, $B_j(j=1,2,\cdots,N)$ represents the j-th barrier influencing the sustainable development of the desalination industry in China, and $x_{ij}(i=1,2,\cdots,N;j=1,2,\cdots,N)$ represents the direct influence of the i-th barrier on the j-th barrier.

It is worth pointing out that x_{ij} equals 0 when the i-th barrier does not have direct influence on the j-th barrier. Meanwhile, all the elements in the principal diagonal of the initial influence matrix all equal zero.

The elements in the i-th row represent the direct influences of the i-th barrier on all the other barriers, and the elements in the j-th column represent the direct influences of all the barriers on the j-th barrier. In order to solve the above-mentioned problem existing in the traditional DEMATEL, the fuzzy best-worst (BW) method was employed to determine the elements of each column in the direct influence matrix. The elements in the j-th column representing the relative influences of the N criterion on the j-th barrier are determined by using the fuzzy best-worst method developed by Guo and Zhao (2017) based on the works of Rezaei (2015, 2016). The fuzzy best-worst method for determining the elements of each column of the initial influence matrix presented in Eq.1consists of five steps (Guo and Zhao, 2017):

Substep 1: Determining the factor which has the most influence on the j-th factor as well as the criterion which has the least influence on the j-th factor.

Note that sometimes only some of the N factors influence the j-th factor, and the users can firstly distinguish the factors that have influences on the j-th factor as well as the factors that do not have any influence on the j-th factor. It is worth pointing out that $x_{kj}=0$ if the k-th criterion does not have any influence on the j-th criterion. Suppose that there are a total of T factors ($T \le N$) influencing the j-th factor, and rearrange these T factors as C_1, C_2, $\cdots$, C_T, the users can then determine the factor that has the most influence (best) on the j-th factor as well as the criterion that has the least influence (worst) on the j-th factor, denoted by C_M and C_L, respectively.

Substep 2: Determining the BO and OW vectors.

The users can determine the best-to-others (BO) vector by comparing the relative preferences of C_M with all the T factors with respect to their influences on the j-th factor by using the following five linguistic phrases, and they are (Kilincci and Onal, 2011):

"Equal importance (EI)": (1,1,1);
"Weak importance (WI)": (2/3,1,3/2);
"Fair importance (FI)": (3/2, 2, 5/2);
"Strong importance (SI)": (5/2, 3, 7/2); and
"Absolute importance (AI)": (7/2, 4, 9/2).

In a similar way, the others-to-worst (BO) vector can also be determined by comparing the relative preferences of all the T criteria with C_L regarding their influences on the j-th factor by using the above-mentioned five linguistic phrases. The linguistic terms in BO and OW vectors can be transformed into triangular fuzzy numbers, denoted by $\widetilde{V}_{BO}$ and $\widetilde{V}_{OW}$, respectively. It is apparent that $\widetilde{a}_{Bk} == (1,1,1)$ when $k = M$, and $\widetilde{a}_{kW} = (1,1,1)$ when $k = L$.

$$\widetilde{V}_{BO} = [\widetilde{a}_{B1} \quad \widetilde{a}_{B2} \quad \cdots \quad \widetilde{a}_{BT}] \tag{15.10}$$

$$\widetilde{V}_{OW} = [\widetilde{a}_{1W} \quad \widetilde{a}_{2W} \quad \cdots \quad \widetilde{a}_{TW}] \tag{15.11}$$

where $\widetilde{a}_{Bk}(k = 1, 2, \cdots, T)$ and $\widetilde{a}_{kW}(k = 1, 2, \cdots, T)$, which are fuzzy numbers, represent the relative preference of C_M comparing with the k-th criterion and that of the k-th criterion comparing with C_L regarding their influences on the j-th criterion.

Substep 3: Determining the fuzzy optimum weights of the T elements $\left[\widetilde{\omega}_1^j \quad \widetilde{\omega}_2^j \quad \cdots \quad \widetilde{\omega}_T^j\right]$.

The weights of these T factors, which represent the relative influences of these T criteria on the j-th criterion, can be determined by solving programming (15.12).

$$\begin{aligned}
&\min \xi^* \\
&s.t. \\
&\left|\frac{\omega_B^{j,L}}{\omega_k^{j,U}} - a_{Bk}^{j,L}\right| \le \xi^* \\
&\left|\frac{\omega_B^{j,M}}{\omega_k^{j,M}} - a_{Bk}^{j,M}\right| \le \xi^* \\
&\left|\frac{\omega_B^{j,U}}{\omega_k^{j,L}} - a_{Bk}^{j,U}\right| \le \xi^* \\
&\left|\frac{\omega_k^{j,L}}{\omega_W^{j,U}} - a_{kW}^{j,L}\right| \le \xi^* \\
&\left|\frac{\omega_k^{j,M}}{\omega_W^{j,M}} - a_{kW}^{j,M}\right| \le \xi^* \\
&\left|\frac{\omega_k^{j,U}}{\omega_W^{j,L}} - a_{kW}^{j,U}\right| \le \xi^* \\
&\sum_{k=1}^{n} R\left(\widetilde{\omega}_k^j\right) = \sum_{k=1}^{n} \frac{\omega_j^{j,L} + 4\omega_j^{j,M} + \omega_j^{j,U}}{6} = 1 \\
&\widetilde{\omega}_k^j = \left(\omega_k^{j,L}, \omega_k^{j,M}, \omega_k^{j,U}\right) \\
&\omega_k^{j,L} \le \omega_k^{j,M} \le \omega_k^{j,U} \\
&\omega_k^{j,L} \ge 0 \\
&\qquad\qquad k = 1,2,\cdots,T
\end{aligned} \tag{15.12}$$

where $\widetilde{\omega}_B^j = \left(\omega_B^{j,L}, \omega_B^{j,M}, \omega_B^{j,U}\right)$ represents the fuzzy weight (relative influences) of the most influential factor on the j-th factor, $\widetilde{\omega}_k^j = \left(\omega_k^{j,L}, \omega_k^{j,M}, \omega_k^{j,U}\right)$ represents the fuzzy weight (relative influences) of the k-th factor on the j-th factor, $\widetilde{\omega}_W^j = \left(\omega_W^{j,L}, \omega_W^{j,M}, \omega_W^{j,U}\right)$ represents the weight (relative influences) of the least influential (worst) factor on the j-th factor, $\widetilde{a}_{Bk}^j = \left(a_{Bk}^{j,L}, a_{Bk}^{j,M}, a_{Bk}^{j,U}\right)$ represents the relative preference of the most influential factor comparing with the k-th factor regarding their relative influences on the j-th factor, and $\widetilde{a}_{kW}^j = \left(a_{kW}^{j,L}, a_{kW}^{j,M}, a_{kW}^{j,U}\right)$ represents the relative preference of the k-th factor comparing with the least influential factor regarding their relative influences on the j-th factor.

After solving the programming (15.12), the fuzzy weights (relative influences) of the T elements can be determined.

Substep 4: Defuzzying the fuzzy weights into crisp weights.

The fuzzy weights of these T elements $\left[\widetilde{\omega}_1^j \;\; \widetilde{\omega}_2^j \;\; \cdots \;\; \widetilde{\omega}_T^j\right]$, which represent the relative influences of these factors on the j-th factor, can be transformed into crisp numbers $\left[\omega_1^j \;\; \omega_2^j \;\; \cdots \;\; \omega_T^j\right]$ by Eq.(15.13).

$$\omega_k^j = \frac{\omega_k^{j,L} + 4\omega_k^{j,M} + \omega_k^{j,U}}{6} \tag{15.13}$$

where ω_k^j represents the defuzzied weight of the k-th factor with respect to its influence on the j-th factor.

Substep 5: Consistency check.

In this step, a consistency check will be carried out to measure the overall consistency of the BO and OW vectors. The consistency ratio (CR) can be calculated by Eq. (15.14). The consistency index (CI) can be determined according to the value of $\widetilde{a}_{BW}$ (see Table 15.1).

$$\mathrm{CR} = \frac{\xi^*}{\mathrm{CI}} \tag{15.14}$$

where CR represents the consistency ratio, and CI represents the consistency index.

The smaller the value of CR, the more consistent the BO and OW vectors are. The judgments can be recognized as consistent when $\mathrm{CR} \leq 0.10$， otherwise the users can revise the BO or the OW vector to make it acceptable.

After these, the elements of the j-th column in the initial influence matrix can be determined. In a similar way, all the elements in the initial influence matrix can be determined.

TABLE 15.1 CI with respect to $\widetilde{a}_{BW}$ used in the fuzzy best-worst method.

$\widetilde{a}_{BW}$	(1,1,1)	(2/3,1,3/2)	(3/2,2,5/2)	(5/2,3,7/2)	(7/2,4,9/2)
CI	3.00	3.80	5.29	6.69	8.04

From Guo, S., Zhao, H., 2017. Fuzzy best-worst multi-criteria decision-making method and its applications. Knowl.-Based Syst. 121, 23–31.

Step 2: Determining the normalized initial influence matrix A.

The normalized initial influence matrix A can be determined by normalizing the initial influence matrix by Eqs. (15.15), (15.16).

$$a_{ij} = \frac{x_{ij}}{\max\left[\max\limits_{1\le i\le N}\sum_{j=1}^{N} a_{ij},\ \max\limits_{1\le j\le N}\sum_{i=1}^{N} a_{ij}\right]} \quad (i=1,2,\cdots,N; j=1,2,\cdots,N) \tag{15.15}$$

$$A = \begin{array}{cccccc} & B_1 & B_2 & \cdots & B_N \\ B_1 & a_{11} & a_{12} & \cdots & a_{1N} \\ B_2 & a_{21} & a_{22} & \cdots & a_{2N} \\ \vdots & \vdots & \vdots & \ddots & \vdots \\ B_N & a_{N1} & a_{N2} & \vdots & a_{NN} \end{array} \tag{15.16}$$

where A represents the normalized initial influence matrix, $a_{ij}(i=1,2,\cdots,N;j=1,2,\cdots,N)$, which is the element of the cell (i, j) in the normalized initial influence matrix, A represents the normalized influence of the i-th factor on the j-th factor.

Step 3: Calculating the total influence matrix T

Thee total influence matrix T can be determined by Eq. (15.17).

$$T = \{t_{ij}\}_{N\times N} = \sum_{n=1}^{\infty} A^n = A(I-A)^{-1} \tag{15.17}$$

where T represents the total influence matrix, I is the identity matrix, and t_{ij} represents the element of cell (i, j) in the total influence matrix.

Step 4: Determining the sum of each row and the sum of each column

The sum of the i-th row and the sum of the j-th column can be determined by Eqs. (15.18), (15.19), respectively.

$$R_i = \sum_{j=1}^{N} t_{ij} \tag{15.18}$$

$$C_j = \sum_{i=1}^{N} t_{ij} \tag{15.19}$$

where R_i represents the sum of the i-th row in the total influence matrix, and C_j represents the sum of the j-th column in the total influence matrix.

R_i, as the sum of the i-th row, represents the total direct and indirect effects of the i-th factor on the other factors, and C_j, as the sum of the j-th column, shows the total direct and indirect effects of all the influential factors on the j-th factor. When $i=j$, R_i+C_j represents the total effects exerted and received by the i-th factor, and it can be used as an index to show the relative importance of the i-th factor in the system. R_i-C_j shows the new difference that contributed by the i-th factor to the system and it is the difference of the influences of the i-th factor exerted on the other factors from that received by the i-th factor from the other factors. If R_i-C_j is greater than zero, then the i-th factor can be recognized as a causal factor, and if it is less than zero, then, it can be recognized as an effect factor.

Step 5: Determining the weight of each influential factor

The weight of the i-th factor can be determined by Eqs. (15.20), (15.21).

$$\omega_i' = \sqrt{\left(R_i + C_j\right)^2 + \left(R_i - C_j\right)^2} \tag{15.20}$$

$$\omega_i = \frac{\omega_i'}{\sum_{i=1}^{N} \omega_i'} \tag{15.21}$$

where ω_i represents the relative weight of the i-th factor.

Step 6: Drawing the IRM (influence relation map)

The IRM is drawn according to ω_i, which represents the relative importance of the i-th factor and $R_i - C_j$, which shows the category (causal factor or effect factor) of the i-th factor.

15.4 Results and discussion

In order to determine the initial influence matrix, the interdependence relation matrix was firstly determined by using "O" and "X":

(1) "O" was used to describe the influence of the i-th factor on the j-th factor if the i-th factor does not exert the j-th factor;
(2) "X" was used to describe the influence of the i-th factor on the j-th factor if the i-th factor exerts the j-th factor.

For instance, there are three influential factors including lack of advanced technologies (T_1), lack of special research and development (R&D) funding (EC_3), lack of awareness and perceptions (SM_1), and lack of supporting policies and regulations (SM_2), having influences on high energy consumption (T_2), thus, "X" was put in the corresponding cells of the second column, as presented in Table (15.2). After these, the fuzzy best-worst method was employed to determine the relative influences of the barriers on each barrier (Step 1).

Step 1: Establishing the initial influence matrix X.

A total of nine experts on seawater desalination was invited to participate in a focus group meeting; three of them are full professors whose research mainly focuses on sustainable desalination technologies, three of them are senior researchers with PhD degrees who have worked on developing desalination processes for more than 10 years, and three of them are senior engineers who have worked in the national seawater desalination factories for more than 10 years. Taking the elements in the second column (the relative influences of T_1, EC_2, SM_1 and SM_2 on T_2) as an example:

Substep 1: Determine the most influential factor and the least influential factor.

TABLE 15.2 The interdependence relation matrix.

	T_1	T_2	EC_1	EC_2	EC_3	EN_1	SM_1	SM_2	SM_3
T_1	O	X	X	X	O	X	O	O	O
T_2	O	O	O	X	O	X	O	O	O
EC_1	O	O	O	O	O	O	O	O	O
EC_2	O	O	O	O	O	O	O	O	O
EC_3	X	X	X	O	O	O	O	O	O
EN_1	O	O	O	O	O	O	O	O	O
SM_1	X	X	O	O	X	X	O	X	O
SM_2	X	X	X	O	X	O	X	O	O
SM_3	O	O	X	X	X	X	X	X	O

The lack of advanced technologies (T_1) and the lack of awareness and perceptions (SM_1) were recognized as the most influential and the least influential, respectively.

Substep 2: Determining the BO and OW vectors.

For instance, these nine experts held the view that the relative influence of T_1 on T_2, compared with that of EC_3 on T_2, should be "fair importance (FI)," thus, "FI" was used to describe the preference of T_1 over EC_3. In a similar way, all the elements in the BO and OW vectors can be determined by using linguistic terms, and the BO and OW vectors by using linguistic terms, as presented in Table (15.3).

The linguistic terms in Table 15.3 can be transformed into fuzzy numbers, and the BO and OW vectors by using the triangular fuzzy numbers are presented in Table 15.4.

TABLE 15.3 The BO and OW vectors by using the linguistic terms for determining the relative influences of T_1, EC_2, SM_1, and SM_2 on T_2.

	The most influential: T_1		The least influential: SM_1	
	T_1	EC_3	SM_1	SM_2
BO	EI	FI	AI	SI
OW	AI	SI	EI	FI

TABLE 15.4 The BO and OW vectors by using the linguistic terms for determining the relative influences of T_1, EC_2, SM_1 and SM_2 on T_2.

	The most influential: T_1		The least influential: SM_1	
	T_1	EC_3	SM_1	SM_2
BO	(1,1,1)	(3/2, 2, 5/2)	(7/2, 4, 9/2)	(5/2, 3, 7/2)
OW	(7/2, 4, 9/2)	(5/2, 3, 7/2)	(1,1,1)	(3/2, 2, 5/2)

Substep 3: Determining the fuzzy optimum weights.

The fuzzy optimum weights (relative influences) of the four influential factors including T_1, EC_2, SM_1, and SM_2 on T_2 are determined by solving the following programming (15.22).

$$\begin{aligned}
&\min \xi^* \\
&s.t. \\
&\left|\frac{\omega_{T_1}^{T_2,L}}{\omega_{EC_3}^{T_2,U}}-\frac{3}{2}\right| \le \xi^*, \left|\frac{\omega_{T_1}^{T_2,M}}{\omega_{EC_3}^{T_2,M}}-2\right| \le \xi^* \left|\frac{\omega_{T_1}^{T_2,U}}{\omega_{EC_3}^{T_2,L}}-\frac{5}{2}\right| \le \xi^* \\
&\left|\frac{\omega_{T_1}^{T_2,L}}{\omega_{SM_1}^{T_2,U}}-\frac{7}{2}\right| \le \xi^*, \left|\frac{\omega_{T_1}^{T_2,M}}{\omega_{SM_1}^{T_2,M}}-4\right| \le \xi^* \left|\frac{\omega_{T_1}^{T_2,U}}{\omega_{SM_1}^{T_2,L}}-\frac{9}{2}\right| \le \xi^* \\
&\left|\frac{\omega_{T_1}^{T_2,L}}{\omega_{SM_2}^{T_2,U}}-\frac{5}{2}\right| \le \xi^*, \left|\frac{\omega_{T_1}^{T_2,M}}{\omega_{SM_2}^{T_2,M}}-3\right| \le \xi^* \left|\frac{\omega_{T_1}^{T_2,U}}{\omega_{SM_2}^{T_2,L}}-\frac{7}{2}\right| \le \xi^* \\
&\left|\frac{\omega_{EC_3}^{T_2,L}}{\omega_{SM_1}^{T_2,U}}-\frac{5}{2}\right| \le \xi^*, \left|\frac{\omega_{EC_3}^{T_2,M}}{\omega_{SM_1}^{T_2,M}}-3\right| \le \xi^*, \left|\frac{\omega_{EC_3}^{T_2,U}}{\omega_{SM_1}^{T_2,L}}-\frac{7}{2}\right| \le \xi^* \\
&\left|\frac{\omega_{SM_2}^{T_2,L}}{\omega_{SM_1}^{T_2,U}}-\frac{3}{2}\right| \le \xi^*, \left|\frac{\omega_{SM_2}^{T_2,M}}{\omega_{SM_1}^{T_2,M}}-2\right| \le \xi^*, \left|\frac{\omega_{SM_2}^{T_2,U}}{\omega_{SM_1}^{T_2,L}}-\frac{5}{2}\right| \le \xi^* \\
&\sum_{k=T_1,EC_3,SM_1,SM_2} \frac{\omega_k^{T_2,L}+4\omega_k^{T_2,M}+\omega_k^{T_2,U}}{6}=1 \\
&\omega_k^{T_2,L} \le \omega_k^{T_2,M} \le \omega_k^{T_2,U} \quad k=T_1,EC_3,SM_1,SM_2 \\
&\omega_k^{T_2,L} \ge 0 \quad k=T_1,EC_3,SM_1,SM_2
\end{aligned} \tag{15.22}$$

The results are presented in Table 15.5. The optimum value of ξ^* is 0.4074.

Substep 4: Defuzzying the fuzzy weights into crisp weights.

The defuzzied weights (the relative influences) of these four influential factors can be determined as presented in Eqs. (15.23)–(15.26).

$$\omega_{T_1}^{T_2}=\frac{0.4385+4\times 0.4385+0.4914}{6}=0.4473 \tag{15.23}$$

TABLE 15.5 The relative influences of T_1, EC_2, SM_1, and SM_2 on T_2.

	$\omega_k^{T_2,L}$	$\omega_k^{T_2,M}$	$\omega_k^{T_2,U}$
$k=T_1$	0.4385	0.4385	0.4914
$k=EC_3$	0.2348	0.2753	0.3101
$k=SM_1$	0.1001	0.1053	0.1122
$k=SM_2$	0.1511	0.1689	0.2096

$$\omega_{EC_3}^{T_2} = \frac{0.2348 + 4 \times 0.2753 + 0.3101}{6} = 0.2744 \tag{15.24}$$

$$\omega_{SM_1}^{T_2} = \frac{0.1001 + 4 \times 0.1053 + 0.1122}{6} = 0.1056 \tag{15.25}$$

$$\omega_{SM_2}^{T_2} = \frac{0.1511 + 4 \times 0.1689 + 0.2096}{6} = 0.1727 \tag{15.26}$$

Substep 5: Consistency check.

The consistency ratio can be determined by Eq. (15.27).

$$\mathrm{CR} = \frac{\xi^*}{\mathrm{CI}} = \frac{0.4074}{8.04} = 0.0507 \tag{15.27}$$

It is apparent that CR is less than 0.10; thus, the consistency of these judgments is acceptable, and the judgments of the experts for determining the weights (relative influences) of these four influential factors are effective.

In a similar way, the relative influences of the influential factors on each factor can be determined, and the results are presented in the Appendix. The initial influential matrix can then be determined, as presented in Table 15.6.

Step 2: Determining the normalized initial influence matrix.

The normalized initial influence matrix can be determined, and the results were presented in Table 15.7.

Step 3: Calculating the total influence matrix T.

Thee total influence matrix T can be determined, and the results are the same as with the normalized initial matrix.

TABLE 15.6 The initial influence matrix.

	T_1	T_2	EC_1	EC_2	EC_3	EN_1	SM_1	SM_2	SM_3
T_1	0	0.4473	0.4489	0.5304	0	0.2927	0	0	0
T_2	0	0	0	0.3021	0	0.4546	0	0	0
EC_1	0	0	0	0	0	0	0	0	0
EC_2	0	0	0	0	0	0	0	0	0
EC_3	0.5554	0.2744	0.1486	0	0	0	0	0	0
EN_1	0	0	0	0	0	0	0	0	0
SM_1	0.1425	0.1056	0	0	0.1412	0.1430	0	0.5015	0
SM_2	0.3021	0.1727	0.2900	0	0.4306	0	0.7496	0	0
SM_3	0	0	0.1125	0.1675	0.4283	0.1097	0.2503	0.4985	0

TABLE 15.7 The normalized initial influence matrix.

	T_1	T_2	EC_1	EC_2	EC_3	EN_1	SM_1	SM_2	SM_3
T_1	0	0.2300	0.2308	0.2727	0	0.1505	0	0	0
T_2	0	0	0	0.1553	0	0.2337	0	0	0
EC_1	0	0	0	0	0	0	0	0	0
EC_2	0	0	0	0	0	0	0	0	0
EC_3	0.2856	0.1411	0.0764	0	0	0	0	0	0
EN_1	0	0	0	0	0	0	0	0	0
SM_1	0.0733	0.0543	0	0	0.0726	0.0735	0	0.2578	0
SM_2	0.1553	0.0888	0.1491	0	0.2214	0	0.3854	0	0
SM_3	0	0	0.0578	0.0861	0.2202	0.0564	0.1287	0.2563	0

Step 4: Determining the sum of each row and the sum of each column.

The results were the total effects ($R_i + C_j$) exerted and received by each factor and the new difference ($R_i - C_j$) that contributed by each factor to the system can also be determined; they are presented in Table 15.8.

Step 5: Determining the weight of each influential factor.

The weight of each influential factor can be determined (also see Table 15.8).

Step 6: Drawing the IRM (influence relation map).

The influence relation map is presented in Fig. 15.1. It is apparent that there are five causal barriers, and they are: lack of advanced technologies (T_1), lack of special research and development (R&D) funding (EC_3), lack of awareness and perceptions (SM_1), lack of supporting policies and regulations (SM_2), and the unclear responsibility in administration (SM_3). The other four barriers, including high energy consumption (T_2), high capital cost (EC_1), high production cost (EC_2), and negative environmental impacts (EN_1) are effect barriers.

TABLE 15.8 The total influence matrix.

	T_1	T_2	EC_1	EC_2	EC_3	EN_1	SM_1	SM_2	SM_3
R_i	0.9734	0.3891	0	0	0.8359	0	1.1526	1.8150	1.6031
C_i	0.8922	0.9291	0.8829	0.9018	0.7306	0.9237	0.7908	0.7180	0
$R_i + C_i$	1.8657	1.3181	0.8829	0.9018	0.7306	0.9237	0.7908	0.7180	1.6031
$R_i - C_i$	0.0812	−0.5400	−0.8829	−0.9017	0.1053	−0.9237	0.3618	1.0970	1.6031
ω_i	0.1190	0.0907	0.0796	0.0812	0.1000	0.0832	0.1259	0.1759	0.1444

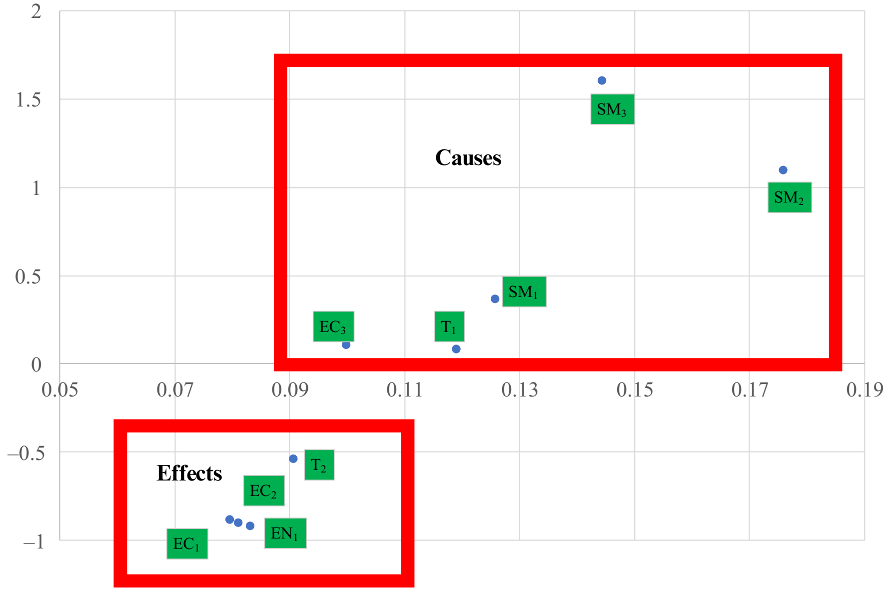

FIG. 15.1 Influence relation map.

Among these causal barriers, the threshold value was set as 0.10 to identify the most important barriers, the weights of T_1, SM_1, SM_2, and SM_3 are greater than 0.10. Meanwhile, all four barriers are the most important causes, and they are the essence of low sustainable development of desalination industry in China. Based on these, four policy implications for mitigating these causal barriers are proposed in Section 15.5.

15.5 Policy implications

As T_1, SM_1, SM_2, and SM_3 were identified as the four most important causes, the following four policy implications are presented:

(1) Improving the technological level of desalination technologies in China: the lack of advanced desalination technologies can lead to high energy consumption, high capital cost, high operations costs, and more negative environmental impacts. In order to reduce energy consumption, capital, cost and operations cost, and mitigate the negative environmental impacts, developing more advanced desalination technologies is prerequisite. There are several ways:

  (i) one of the most important ways for developing more advanced desalination technologies is to establish some special grants/funds for the research, development, and demonstration (R,D&D) of sustainable desalination projects;

(ii) the introduction of foreign investment and the collaboration with some foreign companies with advanced desalination technologies; and

(iii) establish a professional talent team for China's desalination industry.

(2) Enhancing the awareness and perceptions of the stakeholders of seawater desalination: the awareness and perceptions of the stakeholders of seawater desalination for mitigating water supply crises and solving environmental problems should be enhanced. The low awareness and perceptions lead to the lack of advanced technologies, high energy consumption, the lack of funds for R&D on desalination technologies, the negative environmental impacts, and the lack of governmental supporting policies and regulations. The most important ways are education, popular science, and publicity for making more citizens in China.

(3) Setting more governmental industry-supporting policies and regulations: the administration in China should set more industry-supporting policies and regulations for China's desalination industry. The lack of governmental supporting policies leads to the lack of advanced desalination technologies, high energy consumption, high capital cost, lack of R&D funds for desalination technologies, and low social acceptability of desalination technologies for water supply. There are three suggestions for supporting China's desalination industry:

(i) set low or even zero interest loans for China's desalination companies;

(ii) set special subsidies for China's desalination companies; and

(iii) set priority strategy for desalted water supply in the coastal cities of China.

(4) Establishing a special governmental sector responsible for desalination: the responsibilities of different governmental sectors for desalination industry are unclear. The unclear responsibility between different governmental sectors can lead to high capital cost, high operations cost, lack of R&D funds, negative environmental impacts, low social acceptability, and lack of governmental support.

15.6 Conclusions

This study aims to develop a generic method for identifying the barriers influencing the sustainable development of the desalination industry in China and investigating the interdependent cause-effect relationships among these barriers. A total of nine key barriers in four categories were identified. Subsequently, an improved DEMATEL method was developed to investigate the cause-effect relationships among the barriers. Different from the traditional DEMATEL method, the developed DEMATEL method can assure the accurate determination of the relative influences of the influential factors on each factor. After determining the cause-effect relationships among these barriers, some policy implications were provided for promoting the sustainable development of the desalination industry in China.

Appendix: The relative influences of the influential factors on each factor

TABLE 15.A1 The relative influences of the influential factors on T_1.

	The most influential: EC_3		The least influential: SM_1
T_1	**EC_3**	**SM_1**	**SM_2**
BO	(1,1,1)	(7/2, 4, 9/2)	(3/2, 2, 5/2)
OW	(7/2, 4, 9/2)	(1,1,1)	(2/3, 1, 3/2)
Weights	0.5554	0.1425	0.3021
$\xi^* = 0.5000$, $CR = \frac{\xi^*}{CI} = \frac{0.5000}{8.04} = 0.0622 < 0.10$			

TABLE 15.A2 The relative influences of the influential factors on EC_1.

	The most influential: T_1		The least influential: SM_3	
EC_1	**T_1**	**EC_3**	**SM_2**	**SM_3**
BO	(1,1,1)	(5/2, 3, 7/2)	(3/2, 2, 5/2)	(7/2, 4, 9/2)
OW	(7/2, 4, 9/2)	(2/3, 1, 3/2)	(5/2, 3, 7/2)	(1,1,1)
Weights	0.4489	0.1486	0.2900	0.1125
$\xi^* = 0.4273$, $CR = \frac{\xi^*}{CI} = \frac{0.4273}{8.04} = 0.0531 < 0.10$				

TABLE 15.A3 The relative influences of the influential factors on EC_2.

	The most influential: T_1		The least influential: SM_3
EC_2	**T_1**	**T_2**	**SM_3**
BO	(1,1,1)	(3/2, 2, 5/2)	(5/2, 3, 7/2)
OW	(5/2, 3, 7/2)	(3/2, 2, 5/2)	(1,1,1)
Weights	0.5304	0.3021	0.1675
$\xi^* = 0.2087$, $CR = \frac{\xi^*}{CI} = \frac{0.2087}{6.69} = 0.0312 < 0.10$			

TABLE 15.A4 The relative influences of the influential factors on EC_3.

	The most influential: SM_2		The least influential: SM_1
EC_3	**SM_1**	**SM_2**	**SM_3**
BO	(5/2, 3, 7/2)	(1,1,1)	(2/3, 1, 3/2)
OW	(5/2, 3, 7/2)	(3/2, 2, 5/2)	(1,1,1)
Weights	0.1412	0.4306	0.4283
$\xi^* = 0.0505$, $CR = \frac{\xi^*}{CI} = \frac{0.2087}{6.69} = 0.0075 < 0.10$			

TABLE 15.A5 The relative influences of the influential factors on EN_1.

	The most influential: T_2		The least influential: SM_3	
EN_1	**T_1**	**T_2**	**SM_1**	**SM_3**
BO	(3/2, 2, 5/2)	(1,1,1)	(5/2, 3, 7/2)	(7/2, 4, 9/2)
OW	(5/2, 3, 7/2)	(7/2, 4, 9/2)	(2/3, 1, 3/2)	(1,1,1)
Weights	0.2927	0.4546	0.1430	0.1097
$\xi^* = 0.4273$, $CR = \frac{\xi^*}{CI} = \frac{0.4273}{8.04} = 0.0531 < 0.10$				

TABLE 15.A6 The relative influences of the influential factors on SM_1.

	The most influential: SM_2		
SM_1	**SM_2**	**SM_3**	**The least influential: SM_3**
BO	(1,1,1)	(3/2, 2, 5/2)	
OW	(3/2, 2, 5/2)	(1,1,1)	
Weights	0.7496	0.2503	
$\xi^* = 0$, $CR = \frac{\xi^*}{CI} = \frac{0}{5.29} = 0 < 0.10$			

TABLE 15.A7 The relative influences of the influential factors on SM_2.

	The most influential: SM_3		
SM_2	**SM_1**	**SM_3**	**The least influential: SM_1**
BO	(2/3, 1, 3/2)	(1,1,1)	
OW	(1,1,1)	(2/3, 1, 3/2)	
Weights	0.5015	0.4985	
$\xi^* = 0$, $CR = \frac{\xi^*}{CI} = \frac{0}{3.80} = 0 < 0.10$			

References

Balfaqih, H., Al-Nory, M.T., Nopiah, Z.M., Saibani, N., 2017. Environmental and economic performance assessment of desalination supply chain. Desalination 406, 2–9.

Chen, S.H., Hsieh, C.H., 2000. Representation, ranking, distance, and similarity of LR type fuzzy number and application. Aus. J. Intell. Process. Syst. 6 (4), 217–229.

Chen, L., Ren, J., 2018. Multi-attribute sustainability evaluation of alternative aviation fuels based on fuzzy ANP and fuzzy grey relational analysis. J. Air Transp. Manag. 68, 176–186.

Cheng, C.H., Lin, Y., 2002. Evaluating the best main battle tank using fuzzy decision theory with linguistic criteria evaluation. Eur. J. Oper. Res. 142 (1), 174–186.

Fiorenza, G., Sharma, V.K., Braccio, G., 2003. Techno-economic evaluation of a solar powered water desalination plant. Energy Convers. Manag. 44 (14), 2217–2240.

Gude, V.G., Nirmalakhandan, N., Deng, S., 2010. Renewable and sustainable approaches for desalination. Renew. Sust. Energ. Rev. 14 (9), 2641–2654.

Guo, S., Zhao, H., 2017. Fuzzy best-worst multi-criteria decision-making method and its applications. Knowl.-Based Syst. 121, 23–31.

Höpner, T., Windelberg, J., 1997. Elements of environmental impact studies on coastal desalination plants. Desalination 108 (1–3), 11–18.

Kilincci, O., Onal, S.A., 2011. Fuzzy AHP approach for supplier selection in a washing machine company. Expert Syst. Appl. 38 (8), 9656–9664.

Li, X., 2010. Techno-economic analysis and comparison of the primary seawater desalination technologies. CFHI Technol. 2, 64–70. (in Chinese). https://doi.org/10.3969/j.issn.1673-3355.2010.02.019.

Li, P., Xiao, F., Gao, H., 2016. The development trend and study on seawater desaliantion industry in China. Water Resour. Hydropower Northeast China 2, 66–68 (in Chinese).

Liang, H., Ren, J., Lin, R., Liu, Y., 2019. Alternative-fuel based vehicles for sustainable transportation: a fuzzy group decision supporting framework for sustainability prioritization. Technol. Forecast. Soc. Chang. 140, 33–43.

Lior, N., 2017. Sustainability as the quantitative norm for water desalination impacts. Desalination 401, 99–111.

Logan, B.E., 2017. The Global Challenge of Sustainable Seawater Desalination. This file is available at: https://pubs.acs.org/doi/pdfplus/10.1021/acs.estlett.7b00167.

Mezher, T., Fath, H., Abbas, Z., Khaled, A., 2011. Techno-economic assessment and environmental impacts of desalination technologies. Desalination 266 (1–3), 263–273.

Morillo, J., Usero, J., Rosado, D., El Bakouri, H., Riaza, A., Bernaola, F.J., 2014. Comparative study of brine management technologies for desalination plants. Desalination 336, 32–49.

Rezaei, J., 2015. Best-worst multi-criteria decision-making method. Omega 53, 49–57.

Rezaei, J., 2016. Best-worst multi-criteria decision-making method: some properties and a linear model. Omega 64, 126–130.

Semiat, R., 2008. Energy issues in desalination processes. Environ. Sci. Technol. 42 (22), 8193–8201.

Shatat, M., Worall, M., Riffat, S., 2013. Opportunities for solar water desalination worldwide. Sustain. Cities Soc. 9, 67–80.

Shahabi, M.P., McHugh, A., Anda, M., Ho, G., 2015. Comparative economic and environmental assessments of centralised and decentralised seawater desalination options. Desalination 376, 25–34.

UNEP, 2012. United Nations Environment Programme. http://www.unep.org/themes/freshwater.html. (Accessed 28 September 2018).

Wang, X., Chu, X., Shan, K., Huang, P., Wang, R., Li, D., Chen, C., 2014. SWOT analysis and industry development strategy of seawater desalination on the islands. China Rural Water Hydropower 10, 96–100 (in Chinese).

Yang, S., 2014. Progress and prospect of seawater desalination in China. Technol. Water Treat. 40 (12), 1–4 (in Chinese).

Zhao, H., Guo, S., 2014. Selecting green supplier of thermal power equipment by using a hybrid MCDM method for sustainability. Sustainability 6 (1), 217–235.

CHAPTER

16

Integrated data envelopment analysis, weighting method and life cycle thinking: A quantitative framework for life cycle sustainability improvement

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16.1 Introduction

Industrial processes usually consume various energy sources and natural resources, and lead to various solid wastes, liquid wastes, and emissions. Accordingly, the sustainability of industrial processes attracts more and more attention nowadays. Sustainability can usually be defined as the management of resources (including natural, social, financial, and technological) to assure that the resources can satisfy the present human needs and will not influence the needs of future generation (Valenti et al., 2018). Three main pillars of sustainability including economic, environmental, and social (the so-called "triple bottom line") aspects are usually considered for sustainability assessment (Lim and Biswas, 2018; Hammer and Pivo, 2017).

Life cycle tools including life cycle assessment (LCA), life cycle costing (LCC), and social life cycle assessment (SLCA) were developed to investigate the performance of different processes or products with respect to the three pillars of sustainability. Heijungs et al. (2010) developed life cycle sustainability assessment (LCSA) by combining LCA, LCC, and SLCA to investigate the three pillars of sustainability from a life cycle perspective. After LCSA of processes or products, the economic performance, environmental impacts, and social influences of different processes or products can be obtained; but the users can only compare two alternative processes or products with respect to one evaluation criterion, and they may face a

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situation that one process performs betters than another with respect to one evaluation criterion, but it may performance worse with respect to another evaluation criterion. Therefore, it is difficult for the stakeholders to know whether or not these processes or products are sustainable, and they also don't know the ways for improving the sustainability of the nonsustainable alternatives. In addition, it is difficult for the decision-makers to get the data of the alternative processes or products with respect to some soft criteria (i.e., social acceptability, working environment, and influences on health, etc.), because these data sometime cannot be quantified. All in all, there are two problems should be addressed:

(1) LCSA can investigate the environmental, economic, and social performance of different processes or products, but the stakeholders do not know whether or not they are sustainable and the ways to improve the sustainability of the nonsustainable processes or products;

(2) It lacks the methods for quantifying the relative performances of the processes or products with respect to the soft criteria.

In order to solve the above two problems, a methodological framework was developed by combining life cycle sustainability assessment method, the intuitionistic fuzzy AHP, and data envelopment analysis (DEA) for judging whether or not these alternatives are sustainable and providing the methods for improving the sustainability of the nonsustainable alternatives. DEA is a linear programming method, which can be used to assess the comparative efficiency of different decision-making units (DMUs) with multiple inputs and outputs (Banker et al., 1984). In this study, LCSA and the intuitionistic fuzzy AHP method were combined to obtained the data with respect to the inputs and outputs. LCSA was employed to collect the data for the hard evaluation criteria that can be quantified by LCA, LCC, and SLCA. The intuitionistic fuzzy AHP as a weighting method is used for determining the data of the alternatives with respect to the soft criteria. After obtaining all the data of all the alternatives with respect to all the evaluation criteria, the benefit-type criteria and the cost-type criteria are used as the outputs and inputs in the DEA model, respectively.

16.2 Methods

An integrated data envelopment, weighting method, and life cycle thinking methodological framework is developed in this study to measure the sustainability efficiency of different energy and industrial systems, and these systems are recognized as the decision-making units (DMUs). The outputs and the inputs are determined based on life cycle thinking with the tools such as life cycle assessment, life cycle costing, and social life cycle assessment. As for the data with respect to the soft criteria, which cannot be obtained directly by using these life cycle tools, they are determined by using the weighting method, and the intuitionistic fuzzy AHP is employed to determine the relative performances of the alternatives with respect to the soft criteria. After determining the data of all the inputs and the outputs, the data envelopment analysis (DEA) method is employed to measure the sustainability efficiency of different energy and industrial systems. The DEA-efficient and the non-DEA-efficient scenarios can be identified, and the countermeasures for improving the non-DEA-efficient scenarios can also be obtained according to the results of DEA. The holistic framework of the method used in this study is presented in Fig. 16.1.

16.2.1 Intuitionistic fuzzy AHP

Definition 16.1 Intuitionistic fuzzy set (Atanassov, 1986; Wang et al., 2011). Let X be an ordinary finite nonempty set, and an intuitionistic fuzzy set A in X can be expressed as:

$$A = \{(A, \mu_A(x), \upsilon_A(x)) | x \in X\} \tag{16.1}$$

where:

$\mu_A(x) : X \to [0,1]$ and $\upsilon_A(x) : X \to [0,1]$ with condition;
$0 \leq \mu_A(x) + \upsilon_A(x) \leq 1$ for all x.in X; and

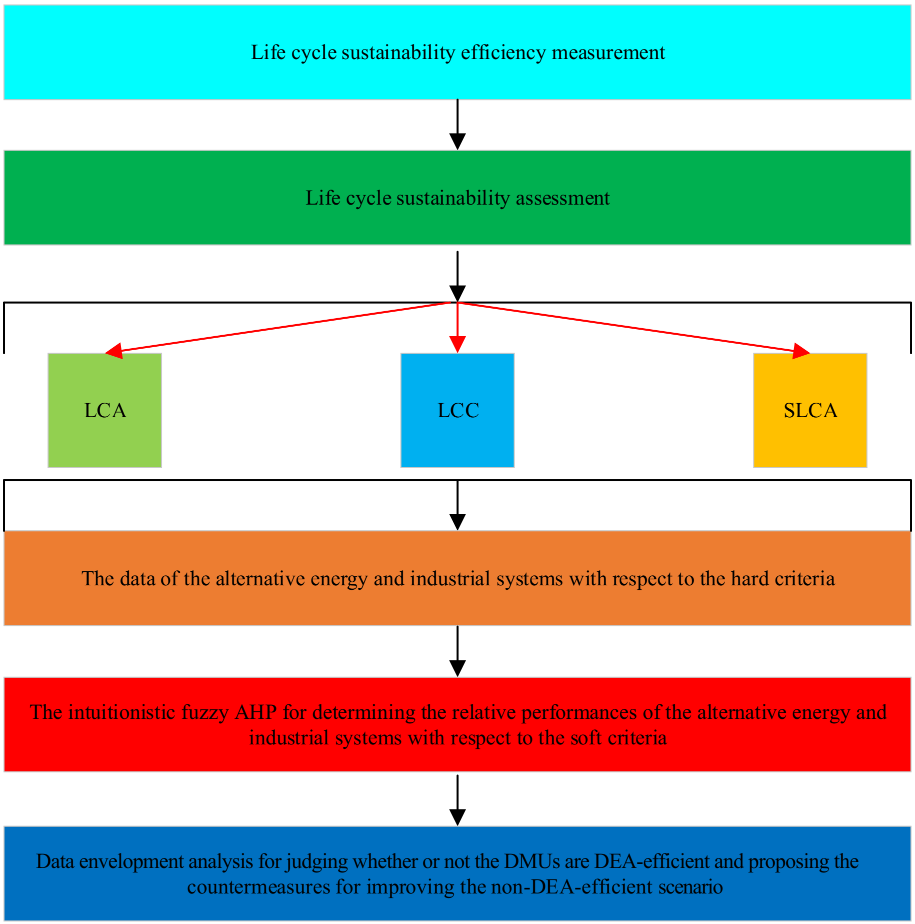

FIG. 16.1 The holistic framework of the method for life cycle sustainability efficiency measurement.

$\mu_A(x): X \to [0,1]$ and $\upsilon_A(x): X \to [0,1]$ represent the degree of membership and the degree of nonmembership of the element x in the set A, respectively.

$\pi_A(x) = 1 - \mu_A(x) - \upsilon_A(x), x \in X$ represents the hesitancy or nondeterminacy degree of the users of the element x to A, the so-called "the indeterminacy degree" or "intuitionistic index" (Wang et al., 2011). It can be determined by Eq. (16.2).

$$\pi_A(x) = 1 - \mu_A(x) - \upsilon_A(x), x \in X \tag{16.2}$$

where $\pi_A(x)$ represents the indeterminacy degree of x to A.
The ordered triple elements $\alpha_A(x) = (\mu_\alpha(x), \upsilon_\alpha(x), \pi_\alpha(x))$ is a typical intuitionistic fuzzy number with the condition that $\mu_\alpha(x)$, $\upsilon_\alpha(x) \in [0 \;\; 1]$ and $\mu_\alpha(x) + \upsilon_\alpha(x) \le 1$ (Xu, 2007). The $\alpha_A(x) = (\mu_\alpha(x), \upsilon_\alpha(x), \pi_\alpha(x))$ can usually be abbreviated as $\alpha_A(x) = (\mu_\alpha(x), \upsilon_\alpha(x))$.

Definition 16.2 Arithmetic operations (Xu, 2007). Let $A = (\mu_A, \upsilon_A)$ and $B = (\mu_B, \upsilon_B)$ be two intuitionistic fuzzy numbers:
Addition:

$$A \oplus B = (\mu_A, \upsilon_A) \oplus (\mu_B, \upsilon_B) = (\mu_A + \mu_B - \mu_A\mu_B, \upsilon_A\upsilon_B) \tag{16.3}$$

Multiplication:

$$A \otimes B = (\mu_A, \upsilon_A) \otimes (\mu_B, \upsilon_B) = (\mu_A\mu_B, \upsilon_A + \upsilon_B - \upsilon_A\upsilon_B) \tag{16.4}$$

Definition 16.3 Intuitionistic fuzzy comparison matrix (Wang et al., 2011). Assume that there are a total of n items ($I_1 \;\; I_2 \;\; \cdots \;\; I_n$), the intuitionistic fuzzy numbers are usually to compare each pair of items in the intuitionistic fuzzy comparison matrix (as presented in Eq. 16.5).

$$\begin{array}{ccccc} & I_1 & I_2 & \cdots & I_n \\ I_1 & (\mu_{11}, \upsilon_{11}) & (\mu_{12}, \upsilon_{12}) & \cdots & (\mu_{1n}, \upsilon_{1n}) \\ I_2 & (\mu_{21}, \upsilon_{21}) & (\mu_{22}, \upsilon_{22}) & \cdots & (\mu_{2n}, \upsilon_{2n}) \\ \vdots & \vdots & \vdots & \ddots & \vdots \\ I_n & (\mu_{n1}, \upsilon_{n1}) & (\mu_{n2}, \upsilon_{n2}) & \cdots & (\mu_{nn}, \upsilon_{nn}) \end{array} \tag{16.5}$$

where $(\mu_{ij}, \upsilon_{ij})$ represents the relative preference of the i-th item over the j-th item.
For instance, $(\mu_{ij}, \upsilon_{ij}) = (0.6, 0.2)$ presents the relative preference of the i-th item over the j-th item, 0.6 represents the certainty degree of the i-th item be superior to the j-th item, and 0.2 represents the certainty degree of the j-th item be superior to the i-th item. Accordingly, the indeterminacy degree $1 - 0.6 - 0.2 = 0.2$ represents the uncertainty degree of the i-th item be preferred than the j-th item. The nine-scale system can be employed to determine the elements in the intuitionistic fuzzy comparison matrix (as presented in Table 16.1).
The elements in the diagonal of the intuitionistic fuzzy comparison matrix representing the relative preference of one item over itself equals (05, 0.5) based on the nine-scale system presented in the work of Xu and Liao (2014).

Definition 16.4 Normalized intuitionistic fuzzy vector (Qian and Feng, 2008). The intuitionistic fuzzy vector:

$$X = (x_1 \;\; x_2 \;\; \cdots \;\; x_n) = ((\mu_1 \;\; \upsilon_1 \;\; \pi_1) \;\; (\mu_2 \;\; \upsilon_2 \;\; \pi_2) \;\; \cdots \;\; (\mu_n \;\; \upsilon_n \;\; \pi_n)) \tag{16.6}$$

TABLE 16.1 The nine-scale system for determining the intuitionistic fuzzy comparison matrix.

Scales	Meanings
0.1	Extremely not preferred
0.2	Very strongly not preferred
0.3	Strongly not preferred
0.4	Moderately not preferred
0.5	Equally preferred
0.6	Moderately preferred
0.7	Strongly preferred
0.8	Very strongly preferred
0.9	Extremely preferred
Other values between 0 and 1	Intermediate judgments between each pair of the above-mentioned adjacent judgments

Source: Xu, Z., Liao, H., 2014. Intuitionistic fuzzy analytic hierarchy process. IEEE Trans. Fuzzy Syst. 22(4), 749–761.

The intuitionistic fuzzy vector is normalized if it can satisfy:

$$\begin{cases} \sum_{i=1}^{n} \mu_i + \pi_j \le 1 \\ \sum_{i=1}^{n} \upsilon_i + \pi_j \le n-1 \end{cases} \quad j = 1, 2, \cdots, n \tag{16.7}$$

The intuitionistic fuzzy AHP developed by Wang et al. (2011) consists of three steps that are employed to determine the data of the alternative industrial systems with respect the soft criteria (i.e., social acceptability, technology maturity, and technology innovation, etc.) the data with respect to which cannot be quantified directly. It consists of the following three steps based on the work of Wang et al. (2011):

Step 1: Establishing the intuitionistic fuzzy comparison matrix. The users can use the nine-scale system (presented in Table 16.1) to determine the intuitionistic fuzzy comparison matrix, as presented in Eq. (16.5).
Step 2: Consistency check (Wang et al., 2011). The intuitionistic fuzzy comparison matrix can be recognized as consistent if and only if it can satisfy the following:

$$\max_k (\mu_{ik} + u_{kj} - 1) \le -\max_k (\upsilon_{ik} + \upsilon_{kj} - 1) \quad \text{for all } i, j = 1, 2, \cdots, n \tag{16.8}$$

If the intuitionistic fuzzy comparison matrix is consistent, the users can execute Step 3, or the users need to modify the comparison matrix until it is consistent.

Step 3: Determining the programming model to calculate the weights of these n items $(I_1 \;\; I_2 \;\; \cdots \;\; I_n)$ based on the work of Wang et al. (2011).

The intuitionistic fuzzy comparison matrix (see Eq. 16.5) can be divided into two nonnegative matrices, and they are presented in Eqs. (16.9), (16.10), respectively.

$$U_M = \begin{array}{cccccc} & I_1 & I_2 & \cdots & I_n \\ I_1 & \mu_{11} & \mu_{12} & \cdots & \mu_{1n} \\ I_2 & \mu_{21} & \mu_{22} & \cdots & \mu_{2n} \\ \vdots & \vdots & \vdots & \ddots & \vdots \\ I_n & \mu_{n1} & \mu_{n2} & \cdots & \mu_{nn} \end{array} \tag{16.9}$$

$$V_M = \begin{array}{cccccc} & I_1 & I_2 & \cdots & I_n \\ I_1 & \upsilon_{11} & \upsilon_{12} & \cdots & \upsilon_{1n} \\ I_2 & \upsilon_{21} & \upsilon_{22} & \cdots & \upsilon_{2n} \\ \vdots & \vdots & \vdots & \ddots & \vdots \\ I_n & \upsilon_{n1} & \upsilon_{n2} & \cdots & \upsilon_{nn} \end{array} \tag{16.10}$$

The normalized principle eigenvector of U_M and V_M can be determined, as presented in Eqs. (16.11), (16.12), respectively.

$$\hat{u} = (\hat{u}_1 \;\; \hat{u}_2 \;\; \cdots \;\; \hat{u}_n) \tag{16.11}$$

The normalized principle eigenvector means that it can satisfy the following condition:

$$\sum_{i=}^{n} \hat{u}_i = 1 \tag{16.12}$$

where $\hat{u}$ represents the normalized principle eigenvector of U_M, and $\hat{u}_i (i = 1, 2, \cdots, n)$ represents i-th element in $\hat{u}$.

$$\hat{\upsilon} = (\hat{\upsilon}_1 \;\; \hat{\upsilon}_2 \;\; \cdots \;\; \hat{\upsilon}_n) \tag{16.13}$$

where $\hat{\upsilon}$ represents the normalized principle eigenvector of V_M, and $\hat{\upsilon}_i (i = 1, 2, \cdots, n)$ represents i-th element in $\hat{\upsilon}$.

Similarly, the normalized principle eigenvector of V_M also satisfy the following condition:

$$\sum_{i=}^{n} \hat{\upsilon}_i = 1 \tag{16.14}$$

The programming model is presented in (Eq. 16.15) to calculate the k and l for determining the weights of the n items.

$$\begin{aligned} &\text{Min} \;\; k + \delta \\ &s.t. \\ &k(1 - \hat{u}_i) + l\hat{\upsilon}_i \le 1 \quad i = 1,2,\cdots,n \\ &l(1 - \hat{\upsilon}_i) + k\hat{u}_i \ge 1 \quad i = 1,2,\cdots,n \\ &l\hat{\upsilon}_i - k\hat{u}_i - \delta \ge 0 \quad i = 1,2,\cdots,n \\ &k, \delta \ge 0 \end{aligned} \tag{16.15}$$

Then, the intuitionistic fuzzy weight of the i-th item can be determined by Eq. (16.16).

$$\hat{\omega}_i = (\hat{\omega}_i^u \;\; \hat{\omega}_i^\upsilon) = (k\hat{u}_1 \;\; 1 - l\hat{\upsilon}_i) \tag{16.16}$$

where $\hat{\omega}_i$ which is an intuitionistic fuzzy number represents the relative weight of the i-th item. $\hat{\omega}_i^u$ and $\hat{\omega}_i^\upsilon$ represent the uncertainty membership and the uncertainty membership in $\hat{\omega}_i$.

Step 4: The normalized crisp weight of each item. The crisp weight of the i-th item can be determined by Eq. (16.17) by modifying the geometric method presented in the work of Abdullah et al. (2013).

$$\overline{\omega}_i = \sqrt{\frac{\hat{\omega}_i^u}{\hat{\omega}_i^v}} \tag{16.17}$$

where $\overline{\omega}_i$ represents the crisp weight of the i-th item.

Then, the normalized crisp weight of the i-th item can be determined by Eq. (16.18).

$$\omega_i = \frac{\overline{\omega}_i}{\sum_{i=1}^{n} \overline{\omega}_i} \tag{16.18}$$

where ω_i represents the normalized crisp weight of the i-th item.

16.2.2 Data envelopment analysis

Assuming that there are a total of m DMUs ($i=1, 2, \cdots, m$), and each DMU consists of p inputs and q outputs, as the following defined (Ren et al., 2013):

- $i=1, 2, \cdots, m$: the i-th DMU;
- $j=1, 2, \cdots, p$: the j-th input;
- $t=1, 2, \cdots, p$: the t-th output;
- x_{ij} $(i=1,2,\cdots,m; j=1,2,\cdots,p)$: the data of the i-th DMU with respect to the j-th input;
- y_{it} $(i=1,2,\cdots,m; t=1,2,\cdots,q)$: the data of the i-th DMU with respect to the j-th input;
- μ_j $(j=1,2,\cdots,p)$: the weight of the j-th input; and
- υ_t $(t=1,2,\cdots,q)$: the weight of the t-th output.

Accordingly, all the energy and industrial systems can be recognized as the decision-making units (DMUs), as presented in Fig. 16.2.

The inputs could be all the cost-type criteria, such as global warming potential, acidification potential, life cycle cost, and capital costs, etc. The outputs can be all the benefit-type criteria, such as added jobs, social acceptability, energy efficiency, and exergy efficiency, etc. The efficiency of each DMU can be calculated by determining the ratio of the sum of the weighted outputs to the sum of the weighted inputs (Cooper et al., 2000), as presented in Eq. (16.19).

$$e_i = \frac{\sum_{t=1}^{q} \upsilon_t y_{it}}{\sum_{j=1}^{p} u_j x_{ij}} \tag{16.19}$$

where e_i represents the efficiency of the i-th DMU.

There are various ways for determining the relative weights of the inputs and outputs based on the preferences and the opinions of the decision-makers/stakeholders, such as analytic hierarchy process (AHP) (Saaty, 2004) and best-worst method (BWM) (Rezaei, 2015); both of the two weighting methods are subjective approaches. Different from these weighting

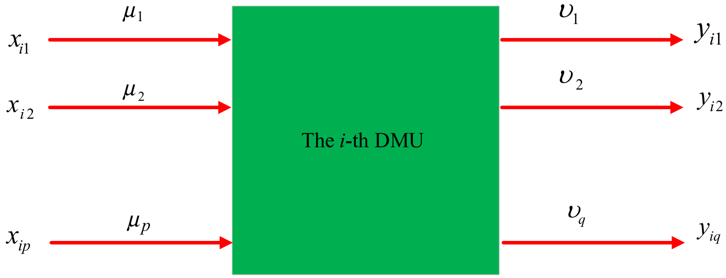

FIG. 16.2 The conceptual structure of the decision-making unit.

methods, the DEA model intends to maximize the efficiency of the target DMU under the constraints that the efficiencies of all the other DMUs are less than 100%. The maximum efficiency of the target DMU i_0 can be determined by solving programming (16.20) according to the work of Charnes et al. (1978):

$$\begin{aligned}
&\max e_{i_0} = \frac{\sum_{t=1}^{q} \upsilon_t y_{i_0 t}}{\sum_{j=1}^{p} u_j x_{i_0 j}} \\
&\frac{\sum_{t=1}^{q} \upsilon_t y_{it}}{\sum_{j=1}^{p} u_j x_{ij}} \le 1 \quad (t = 1, 2, \cdots, m) \\
&u_j \ge \varepsilon \quad j = 1, 2, \cdots, p \\
&\upsilon_t \ge \varepsilon \quad t = 1, 2, \cdots, q
\end{aligned} \tag{16.20}$$

where ε is a non-Archimedean factor.

The constraints in programming (see Eq. 16.20) represents that the upper bound of the efficiency of each DMU cannot exceed 100%. Programming (16.20) can be rewritten into the matrix format (Ye et al., 2006; Ren et al., 2013), as presented in Eq. (16.21):

$$\begin{aligned}
&\max e_{i_0} = \frac{\upsilon^T Y_{i_0}}{u^T X_{i_0}} \\
&\frac{\upsilon^T Y_i}{u^T X_i} \le 1 \quad (i = 1, 2, \cdots, m) \\
&u \ge \varepsilon \\
&\upsilon \ge \varepsilon \\
&u = (u_1, u_2, \cdots, u_p)^T \\
&\upsilon = (\upsilon_1, \upsilon_2, \cdots, \upsilon_q)^T \\
&X_{i_0} = \left(x_{i_0 1}, x_{i_0 2}, \cdots, x_{i_0 p}\right)^T \\
&Y_{i_0} = \left(y_{i_0 1}, y_{i_0 2}, \cdots, y_{i_0 q}\right)^T \\
&X_i = (x_{i1}, x_{i2}, \cdots, x_{ip})^T \\
&Y_i = (y_{i1}, y_{i2}, \cdots, y_{iq})^T
\end{aligned} \tag{16.21}$$

Based on programming (16.21), the corresponding equivalent linear programming can be obtained based on the Charnes-Cooper transformation (Charnes et al., 2013), as presented in Eq. (16.22).

$$\begin{aligned}
&\max v^T Y_{i_0}\\
&u^T X_{i_0} = 1\\
&v^T Y_i - u^T X_i \le 1 \quad (i = 1, 2, \cdots, m)\\
&u \ge \varepsilon \quad j = 1,2,\cdots,p\\
&v \ge \varepsilon \quad t = 1,2,\cdots,q\\
&u = (u_1, u_2, \cdots, u_p)^T\\
&v = (v_1, v_2, \cdots, v_q)^T\\
&X_{i_0} = \left(x_{i_0 1}, x_{i_0 2}, \cdots, x_{i_0 p}\right)^T\\
&Y_{i_0} = \left(y_{i_0 1}, y_{i_0 2}, \cdots, y_{i_0 q}\right)^T\\
&X_i = (x_{i1}, x_{i2}, \cdots, x_{ip})^T\\
&Y_i = (y_{i1}, y_{i2}, \cdots, y_{iq})^T
\end{aligned} \tag{16.22}$$

Then, the linear programming problem presented in Eq. (16.22) can be further transformed into (Ye et al., 2006; Charnes et al., 2013):

$$\begin{aligned}
&\max \left(u^T, v^T\right)\begin{pmatrix} 0 \\ Y_{i_0}\end{pmatrix}\\
&v^T Y_i - u^T X_i \le 0 \quad (i = 1, 2, \cdots, m)\\
&u \ge \varepsilon, v \ge \varepsilon
\end{aligned} \tag{16.23}$$

The CCR model can be determined according to the duality theory in the linear programming, as presented in programming (16.24):

$$\begin{aligned}
&\min \partial - \varepsilon\left(\sum_{j=1}^{p} s_j^+ + \sum_{t=1}^{q} s_t^-\right)\\
&\sum_{i=1}^{m} x_{ij}\Lambda_i + s_j^- = \partial x_{i_0 j}\\
&\sum_{i=1}^{m} y_{it}\Lambda_i - s_t^+ = y_{i_0 t}\\
&\Lambda_i \ge 0 \quad (i = 1, 2, \cdots, m)\\
&s_j^- \ge 0 \quad (j = 1, 2, \cdots, p)\\
&s_t^+ \ge 0 \quad (t = 1, 2, \cdots, q)
\end{aligned} \tag{16.24}$$

where ε represents a non-Archimedean infinitesimal.

Definition 16.5 The DMU can be identified as weak DEA effective if the optimum objective $\partial = 1$, and vice versa (Charnes et al., 1978).

Definition 16.6 If the optimum objective $\partial = 1$ and the solutions satisfy $s_j^- = 0 \quad (j = 1, 2, \cdots, p)$, $s_t^+ = 0 \quad (t = 1, 2, \cdots, q)$, then, the DMU can be recognized as DEA effective, and vice versa (Charnes et al., 1978).

Most of the inputs and outputs of the DMUs can be determined by LCA, LCC, or SLCA. However, the inputs with respect to some qualitative criteria can be determined by using the best-worst method, which is introduced in the following section.

As for the non-DEA-efficient DMUs, the projection of the inputs and outputs of each DMU can be determined by Eqs. (16.25), (16.26), respectively, to make it DEA-efficient (Ye et al., 2006).

$$\widehat{x}_{ij} = \partial_i x_{ij} - s_{ij}^{-} \tag{16.25}$$

$$\widehat{y}_{it} = y_{it} + s_{it}^{+} \tag{16.26}$$

where ∂_i represents the optimum DEA-efficiency of the i-th DMU, and $\widehat{x}_{ij}$ and $\widehat{y}_{it}$ are the improved data of the j-th input and the t-th output in the i-th DMU, respectively.

16.3 Case study

In order to illustrate the developed method for life cycle sustainability efficiency measurement and improvement of energy and industrial systems, four hydrogen production systems have been studied by the proposed method; they are coal gasification (CG), stream reforming of methane (SMR), biomass gasification (BG), and photovoltaic electrolysis (PVEL). There are three input indicators and the output indicators are presented as follows:

Input indicators

(1) Production cost (IP_1): the production cost for producing 1 kg hydrogen;
(2) Global warming potential (IP_2): the global warming potential in the whole life cycle of hydrogen; and
(3) Acidification potential (IP_3): the acidification potential in the whole life cycle of hydrogen.

Output indicators

(1) Hydrogen (OP_1): 1 kg hydrogen as the product in each of the DMUs;
(2) Energy efficiency (OP_2): the ratio of the total energy output to the energy input; and
(3) Social acceptability (OP_3): this criterion is to measure the acceptance by people of different hydrogen production systems, and it can reflect the impacts of the hypothesized hydrogen project on the society herein (Ren et al., 2016).

These hydrogen production systems can be expressed in the format of DMUs, as presented in Fig. 16.3.

The data with respect to the hard indicators in both the inputs and the outputs (e.g., hydrogen production cost, global warming potential, acidification potential, and energy efficiency) can be obtained through life cycle tools, and these were derived from the previously published work of Pilavachi et al. (2009), Acar and Dincer (2014), and Ozbilen et al. (2011). As for the data with respect to the soft criterion, namely social acceptability, this can be determined by the intuitionistic fuzzy AHP method.

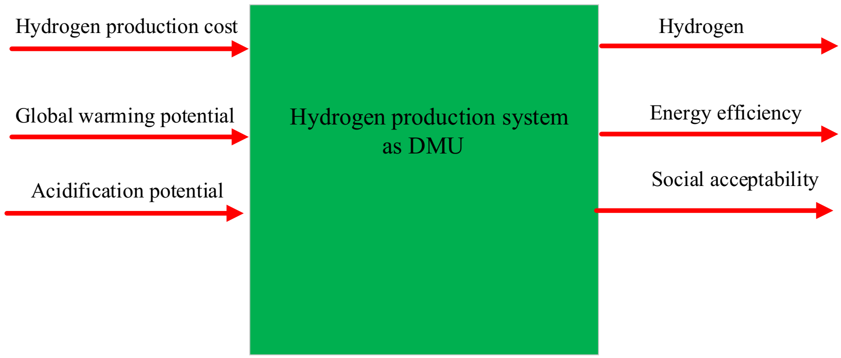

FIG. 16.3 The conceptual structure of the decision-making unit.

16.4 Results and discussion

The intuitionistic fuzzy comparison matrix for determining the relative performances of these four hydrogen production systems with respect to social acceptability was determined, as presented in Table 16.2.

The intuitionistic fuzzy comparison matrix can be divided into two matrices according to Eqs. (16.9), (16.10), as presented in Eqs. (16.27), (16.28), respectively.

$$U_M = \begin{array}{c|cccc} & CG & SMR & BG & PVEL \\ CG & 0.5 & 0.2 & 0.1 & 0.1 \\ SMR & 0.6 & 0.5 & 0.2 & 0.1 \\ BG & 0.8 & 0.7 & 0.5 & 0.2 \\ PVEL & 0.9 & 0.8 & 0.6 & 0.5 \end{array} \tag{16.27}$$

$$V_M = \begin{array}{c|cccc} & CG & SMR & BG & PVEL \\ CG & 0.5 & 0.6 & 0.8 & 0.9 \\ SMR & 0.2 & 0.5 & 0.7 & 0.8 \\ BG & 0.1 & 0.2 & 0.5 & 0.6 \\ PVEL & 0.1 & 0.1 & 0.2 & 0.5 \end{array} \tag{16.28}$$

The normalized eigenvector vector can be determined, as presented in Eqs. (16.29), (16.30).

$$\hat{u} = (\hat{u}_1 \ \ \hat{u}_2 \ \ \hat{u}_2 \ \ \hat{u}_4) = (0.1087 \ \ 0.1709 \ \ 0.2978 \ \ 0.4227) \tag{16.29}$$

$$\hat{v} = (\hat{v}_1 \ \ \hat{v}_2 \ \ \hat{v}_3 \ \ \hat{v}_4) = (0.4227 \ \ 0.2978 \ \ 0.1709 \ \ 0.1087) \tag{16.30}$$

Then, the following programming model can be determined, as presented in Eq. (16.31).

$$\begin{array}{l} \text{Min} \quad k + \delta \\ s.t. \\ k(1-\hat{u}_i) + l\hat{v}_i \le 1 \quad i = 1,2,\cdots,4 \\ l(1-\hat{v}_i) + k\hat{u}_i \ge 1 \quad i = 1,2,\cdots,4 \\ l\hat{v}_i - k\hat{u}_i - \delta \ge 0 \quad i = 1,2,\cdots,4 \\ \hat{u} = (\hat{u}_1 \ \ \hat{u}_2 \ \ \hat{u}_2 \ \ \hat{u}_4) = (0.1087 \ \ 0.1709 \ \ 0.2978 \ \ 0.4227) \\ \hat{v} = (\hat{v}_1 \ \ \hat{v}_2 \ \ \hat{v}_3 \ \ \hat{v}_4) = (0.4227 \ \ 0.2978 \ \ 0.1709 \ \ 0.1087) \\ k, \delta \ge 0 \end{array} \tag{16.31}$$

TABLE 16.2 The intuitionistic fuzzy comparison matrix for determining the relative performances of these four hydrogen production systems with respect to social acceptability.

	CG	SMR	BG	PVEL
CG	(0.5 0.5)	(0.2 0.6)	(0.1 0.8)	(0.1 0.9)
SMR	(0.6 0.2)	(0.5 0.5)	(0.2 0.7)	(0.1 0.8)
BG	(0.8 0.1)	(0.7 0.2)	(0.5 0.5)	(0.2 0.6)
PVEL	(0.9 0.1)	(0.8 0.1)	(0.6 0.2)	(0.5 0.5)

The optimum values of k and l can be determined, and they are 0.33 and 1.67, respectively. Then, the relative performances of these four hydrogen production systems can be determined, as presented in Table 16.3. Then, all the data with respect to the input indicators and the output indicators can be determined, as presented in Table 16.4.

TABLE 16.3 The data of the four hydrogen production systems with respect to social acceptability.

	CG	SMR	BG	PVEL
Intuitionistic fuzzy number	(0.0359 0.2941)	(0.0564 0.5027)	(0.0983 0.7146)	(0.1395 0.8185)
Crisp weights	0.3492	0.3350	0.3708	0.4128
Normalized crisp weights	0.2379	0.2282	0.2526	0.2812

TABLE 16.4 The data of the four hydrogen production systems with respect to the inputs and the outputs indicators in the.

			CG	SMR	BG	PVEL
Inputs	Production cost	US\$ day^{-1} kg^{-1}	22.37	32.75	23.78	17.36
	Global warming potential	g CO_2 eq. kg^{-1}	17,000	12,000	2992	2000
	Acidification potential	g SO_2 eq. kg^{-1}	30.69	14.516	29.03	8.07
Outputs	Hydrogen product	kg	1	1	1	1
	Energy efficiency	/	0.35	0.375	0.65	0.05
	Social acceptability	/	0.2379	0.2282	0.2526	0.2812

The data used in this study were derived from Pilavachi, P.A., Chatzipanagi, A.I., Spyropoulou, A.I., 2009. Evaluation of hydrogen production methods using the analytic hierarchy process. Int. J. Hydrog. Energy 34(13): 5294–5303; Acar, C., Dincer, I., 2014. Comparative assessment of hydrogen production methods from renewable and non-renewable sources. Int. J. Hydrog. Energy 39(1): 1–12; Ozbilen, A., Dincer, I., Rosen, M.A., 2011. A comparative life cycle analysis of hydrogen production via thermochemical water splitting using a Cu–Cl cycle. Int. J. Hydrog. Energy 36(17): 11321–11327.

After obtaining the data of these four alternative hydrogen production systems with respect to both the input and the output indicators, the DEA efficiency of each DMU (hydrogen production system) can be determined. Taking DMU_1 (coal gasification) as an example, the following programming model can be established for determining the DEA-efficiency of this DMU:

$$\begin{aligned}
&\min \partial - \varepsilon\left(\sum_{j=1}^{3} s_j^{+} + \sum_{t=1}^{3} s_t^{-}\right)\\
&22.37\Lambda_1 + 32.75\Lambda_2 + 23.78\Lambda_3 + 17.36\Lambda_4 + s_1^{-} = 22.37\partial\\
&17000\Lambda_1 + 12000\Lambda_2 + 2992\Lambda_3 + 2000\Lambda_4 + s_2^{-} = 17000\partial\\
&30.69\Lambda_1 + 14.516\Lambda_2 + 29.03\Lambda_3 + 8.07\Lambda_4 + s_2^{-} = 30.69\partial\\
&\Lambda_1 + \Lambda_2 + \Lambda_3 + \Lambda_4 - s_1^{+} = 1\\
&0.35\Lambda_1 + 0.375\Lambda_2 + 0.65\Lambda_3 + 0.05\Lambda_4 - s_2^{+} = 0.35\\
&0.2379\Lambda_1 + 0.2282\Lambda_2 + 0.2526\Lambda_3 + 0.2812\Lambda_4 - s_3^{-} = 0.2379\\
&\Lambda_1 \geq 0 \quad (i = 1, 2, 3, 4)\\
&s_j^{-} \geq 0 \quad (j = 1, 2, 3)\\
&s_t^{+} \geq 0 \quad (t = 1, 2, 3)
\end{aligned} \tag{16.32}$$

After solving programming (16.32), the results can be obtained, as presented in Table 16.5. In a similar way, the results with respect to the other three hydrogen production systems can also be determined (see Table 16.5).

According to Definitions 16.5 and 16.6, the hydrogen production systems including SMR, BG, and PVEL can be recognized as DEA-efficient, but CG for hydrogen production is non-DEA-efficient.

As for the data with respect to the first input (production cost), the projected value is:

$$0.9195 \times 22.37 - 0 = 20.5692 \tag{16.33}$$

As for the data with respect to the second input (global warming potential), the projected value is:

$$0.9195 \times 17000 - 13136.19 = 2495.3 \tag{16.34}$$

As for the data with respect to the third input (acidification potential), the projected value is:

$$0.9195 \times 30.69 - 9.6705 = 18.5490 \tag{16.35}$$

TABLE 16.5 The results of programming (16.32).

	∂	s_1^-	s_2^-	s_3^-	s_1^+	s_2^+	s_3^+
CG	0.9195	0	13136.10	9.6705	0	0	0.0290
SMR	1.0000	0	0	0	0	0	0
BG	1.0000	0	0	0	0	0	0
PVEL	1.0000	0	0	0	0	0	0

TABLE 16.6 the improved methods for the non-DEA-efficient hydrogen production system.

	IP_1	IP_2	IP_3	OP_1	OP_2	OP_3
Unit	US$ day^{-1} kg^{-1}	g CO_2 eq. kg^{-1}	g SO_2 eq. kg^{-1}	kg	/	/
Original	22.37	17,000	30.69	1	0.35	0.2379
Projected	20.5692	2495.3	18.5490	1	0.35	0.2669

As for the first and the second output (hydrogen product and energy efficiency), they do not need to be improved according to the obtained results in this case. As for the third output (social acceptability), the project value is:

$$0.2379 + 0.0290 = 0.2669 \tag{16.36}$$

The original values and the projected of CG with respect to the input indicators and the output indicators are summarized in Table 16.6.

In order to improve the CG for hydrogen production and make it DEA-efficient, the production cost of hydrogen should be decreased from 22.37 to 20.5692 US$ day^{-1} kg^{-1}. In addition, the global warming potential (17,000 g CO_2 eq. kg^{-1}) and acidification potential (30.69 g SO_2 eq. kg^{-1}) should be decreased to 2495.3 g CO_2 eq. kg^{-1} and 18.5490 g SO_2 eq. kg^{-1}, respectively. Moreover, the social acceptability of coal gasification should also be improved through various clean coal utilization technologies in hydrogen production.

16.5 Conclusion

Sustainability of energy and industry draws more and more attention with the increase of the environmental consciousness and social influences. This study aims to develop a methodological framework for life cycle sustainability improvement of energy and industrial systems by combining DEA, the intuitionistic fuzzy AHP and life cycle thinking. Life cycle tools including LCA, LCC, and SLCA were suggested to determine the data of the energy and industrial systems with respect to the hard criteria for sustainability assessment, and the intuitionistic fuzzy AHP method was suggested to determine the data with respect to the soft criteria for sustainability assessment. DEA was employed to determine the DEA-efficiency of each DMU (energy and industrial system). The criteria were divided into two types when using DEA: one is the input indicator, and another is the output indicator. DEA can not only be used for judging whether or not the energy and industrial system is consistent, but also for proposing the corresponding countermeasures for improving the sustainability of energy and industrial systems to make the non-DEA-efficient DMUs efficient.

Four typical hydrogen production systems including coal gasification, stream reforming of methane, biomass gasification, and photovoltaic electrolysis were studied by the developed methodological framework. Coal gasification was recognized as non-DEA-efficient, and the methods for improving the sustainability of hydrogen production system based on coal gasification were also obtained according to the obtained results. All in all, the developed method has the following two significant advantages:

(1) The data of the alternative energy and industrial systems with respect to the soft criteria for sustainability assessment can be quantified;
(2) The measures for improving the sustainability of the non-DEA-efficient energy and industrial systems can be obtained.

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References

Abdullah, L., Jaafar, S., Taib, I., 2013. Intuitionistic fuzzy analytic hierarchy process approach in ranking of human capital indicators. J. Appl. Sci. 13 (3), 423–429.
Acar, C., Dincer, I., 2014. Comparative assessment of hydrogen production methods from renewable and non-renewable sources. Int. J. Hydrog. Energy 39 (1), 1–12.
Atanassov, K.T., 1986. Intuitionistic fuzzy set. Fuzzy Sets Syst. 20, 87–96.
Banker, R.D., Charnes, A., Cooper, W.W., 1984. Some models for estimating technical and scale inefficiencies in data envelopment analysis. Manag. Sci. 30 (9), 1078–1092.
Charnes, A., Cooper, W.W., Rhodes, E., 1978. Measuring the efficiency of decision making units. Eur. J. Oper. Res. 2 (6), 429–444.
Charnes, A., Cooper, W.W., Lewin, A.Y., Seiford, L.M. (Eds.), 2013. Data Envelopment Analysis: Theory, Methodology, and Applications. Springer Science & Business Media.
Cooper, W.W., Seiford, L.M., Tone, K., 2000. Data envelopment analysis. In: Cooper, W.W., Seiford, L.M., Zhu, J. (Eds.), Handbook on Data Envelopment Analysis, first ed, pp. 1–40.
Hammer, J., Pivo, G., 2017. The triple bottom line and sustainable economic development theory and practice. Econ. Dev. Q. 31 (1), 25–36.
Heijungs, R., Huppes, G., Guinée, J.B., 2010. Life cycle assessment and sustainability analysis of products, materials and technologies. Toward a scientific framework for sustainability life cycle analysis. Polym. Degrad. Stab. 95 (3), 422–428.
Lim, C.I., Biswas, W.K., 2018. Development of triple bottom line indicators for sustainability assessment framework of Malaysian palm oil industry. Clean Techn. Environ. Policy 20 (3), 539–560.
Ozbilen, A., Dincer, I., Rosen, M.A., 2011. A comparative life cycle analysis of hydrogen production via thermochemical water splitting using a Cu–Cl cycle. Int. J. Hydrog. Energy 36 (17), 11321–11327.
Pilavachi, P.A., Chatzipanagi, A.I., Spyropoulou, A.I., 2009. Evaluation of hydrogen production methods using the analytic hierarchy process. Int. J. Hydrog. Energy 34 (13), 5294–5303.
Qian, G., Feng, X.Q., 2008. Intuitionistic weight generation approach from intuitionistic preference relations. In: 2008 International Conference on Machine Learning and Cybernetics (Vol. 1, pp. 536–541) in Kunming. IEEE, July 12–15.
Ren, J., Manzardo, A., Mazzi, A., Fedele, A., Scipioni, A., 2013. Emergy analysis and sustainability efficiency analysis of different crop-based biodiesel in life cycle perspective. Sci. World J. 2013.
Ren, J., Xu, D., Cao, H., Wei, S.A., Dong, L., Goodsite, M.E., 2016. Sustainability decision support framework for industrial system prioritization. AICHE J. 62 (1), 108–130.
Rezaei, J., 2015. Best-worst multi-criteria decision-making method. Omega 53, 49–57.
Saaty, T.L., 2004. Decision making—the analytic hierarchy and network processes (AHP/ANP). J. Syst. Sci. Syst. Eng. 13 (1), 1–35.
Valenti, W.C., Kimpara, J.M., Preto, B.D.L., Moraes-Valenti, P., 2018. Indicators of sustainability to assess aquaculture systems. Ecol. Indic. 88, 402–413.

Wang, H., Qian, G., Feng, X., 2011. An intuitionistic fuzzy AHP based on synthesis of eigenvectors and its application. Inf. Technol. J. 10 (10), 1850–1866.
Xu, Z., 2007. Intuitionistic fuzzy aggregation operators. IEEE Trans. Fuzzy Syst. 15 (6), 1179–1187.
Xu, Z., Liao, H., 2014. Intuitionistic fuzzy analytic hierarchy process. IEEE Trans. Fuzzy Syst. 22 (4), 749–761.
Ye, Y.C., Ke, H.L., Huang, D.Y., 2006. System Synthetical Evaluation Technology and Its Application, first ed. Matellurgical Industry Press, Beijing, China.

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Life Cycle Sustainability Assessment for Decision-Making

Methodologies and Case Studies

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