Biofuels for a More Sustainable Future

Life Cycle Sustainability Assessment and Multi-Criteria Decision Making

Edited by **Jingzheng Ren, Antonio Scipioni, Alessandro Manzardo and Hanwei Liang**

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CHAPTER 1

Biofuels technologies: An overview of feedstocks, processes, and technologies

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1 Introduction

Biofuels are defined as fuels produced from living plant matter or by-products of agricultural production; they are primarily grouped into biodiesel and ethanol. Biofuels can be divided and separated into several groups based on their technologies, processes, and feedstocks.

Biofuels technology can be defined as application of feedstocks in a sequence of processes leading to the production of different biofuels types. *Biofuels processes* are either natural or chemical stages of an industrial or pilot project development leading to the final production of biofuels. *Biofuels feedstocks* are any living, dead, or decomposed plant materials suitable for processing and conversion to biofuels by means of different processes.

From the perspective of the industrial development and market presence, biofuels feedstocks, processes, and technologies can be classified as "developed" (with well-established markets), "developing" (with newly created or progressing market shares), or in the "demonstration" stage

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(describing pilot projects or potential future developments) (compare: Lane, 2017). Due to a high feedstock variability accessible to be utilized for biofuels generation the existing biofuels technologies and processes have expanded over time thus creating a wide net of production opportunities and innovation potential in this field.

Generally, biofuels technologies can be divided into "conventional" and "advanced" biofuels (Fig. 1.1). Conventional biofuels (also called "first generation biofuels") designate ethanol and biodiesel generated from eatable crops. Advanced biofuels (encompassing the "second, third and fourth generation biofuels") are defined as liquid fuels from nonfood/nonfeed sustainably grown feedstocks and agricultural (municipal) wastes. The need for advanced biofuels originated from a concern about the competition for natural resources (e.g., water, energy, land) between fuel and food production (Rathmann et al., 2010; Harvey and Pilgrim, 2011, Ajanovic, 2011). Accordingly, advanced biofuels cannot create any competition with food crop production, while they need to meet higher sustainability requirements, that is, contribute to greenhouse gas (GHG) emission reduction by a larger percentage than conventional biofuels.

The designation of biofuels "generations" is directly linked and subject to the specific technology and feedstock used for biofuels production. It also relates to the temporal development trends over years and the complexity of

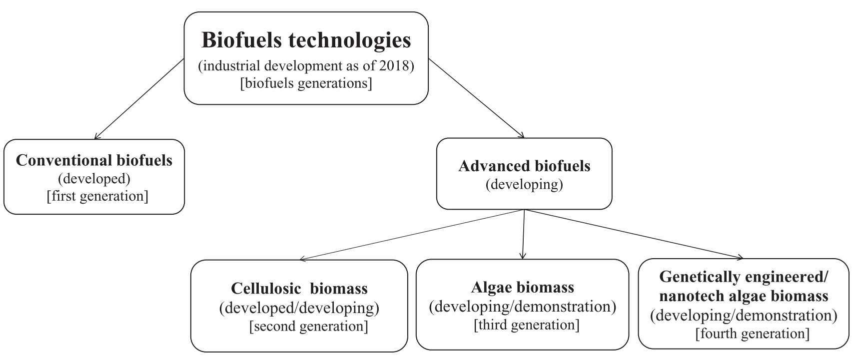

Fig. 1.1 Biofuels technologies with corresponding development stages. *(Authors' presentation modified from Ziolkowska, J.R., 2014. Prospective technologies, feedstocks and market innovations for ethanol and biodiesel production in the US. Biotechnol. Rep. 4, 94–98; Ziolkowska, J.R., 2018. Introduction to biofuels and potentials of nanotechnology. In: Srivastava, N., Srivastava, M., Pandey, H., Mishra, P.K., Ramteke, P.W. (Eds.), Green Nanotechnology for Biofuel Production. Biofuel and Biorefinery Technologies. Springer, Basel, pp. 1–15.)*

the biofuels market with a growing number of potential feedstocks to be used for biofuels production.

First generation biofuels are produced from food crops: (a) biodiesel extracted from oil plants/plant materials (in the chemical process of transesterification and esterification), and (b) ethanol extracted from sugar-containing plants/plant materials and converted to fuel in the process of fermentation. *Second generation biofuels* are produced from nonfood crops (e.g., crop waste, green waste, wood, and energy crops planted specifically for biofuels production). *Third generation biofuels* are based on improvements in biomass production, with algae being the main feedstock representing this group as of today. *Fourth generation biofuels* aim at providing more sustainable production options by combining biofuels production with capturing and storing CO_2 with the process of oxy-fuel combustion or by application of genetic engineering or nanotechnology.

Due to the wide range of feedstock application and process development the evaluation of different biofuels in terms of their sustainability will clearly depend on the combination of those factors. Thus in the face of the multitude of discussions in this field, a closer look at each of the biofuel types is needed for a holistic and science-based evaluation.

Although this chapter does not aim at investigating sustainability of the respective biofuels technologies, processes, and feedstocks per se, it will provide an overview for a better understanding of those issues to be addressed in the following chapters in this book.

2 Biofuels technologies and feedstocks

Globally, the total biofuels production has increased over time, with an estimated ethanol production at 160 billion liters (42.3 billion gallons) in 2019 and biodiesel production at 41 billion liters (11 billion gallons) (OECD, 2010) (Figs. 1.2 and 1.3). The feedstock composition in the global biofuels production has varied and changed considerably over time as well. According to OECD (2010) projections, on the ethanol market, coarse grains (including corn) have reached the peak in 2016, while ethanol production from sugar cane will increase throughout 2019. An increasing trend was also projected for biomass-based ethanol with 11 billion liters (2.9 billion gallons) on the market in 2019. On the biodiesel market, vegetable oils constitute the main feedstock that is expected to increase up to 30.7 billion liters (8.1 billion gallons) by 2019 (OECD, 2010). Also jatropha and other nonagricultural feedstocks (animal fats) make a smaller share in the biodiesel

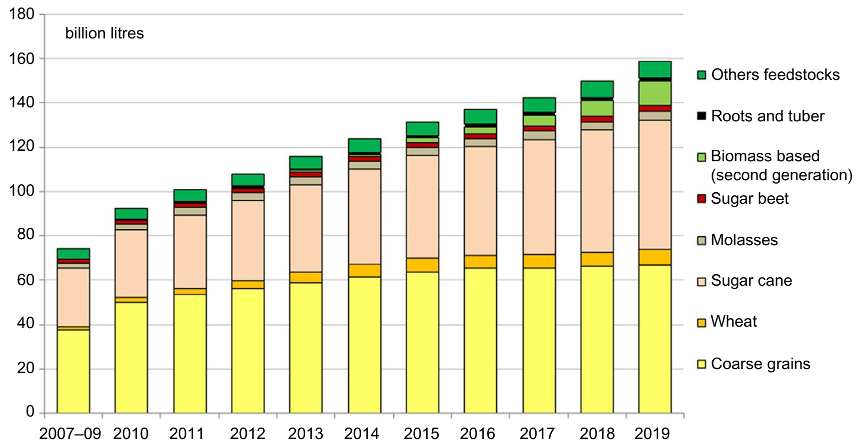

Fig. 1.2 Global ethanol production by feedstock—projections (2007–19). *(Modified from OECD-FAO, 2010. Agricultural Outlook 2010. Biofuel Production 2010–19; Ziolkowska, J.R., 2018. Introduction to biofuels and potentials of nanotechnology. In: Srivastava, N., Srivastava, M., Pandey, H., Mishra, P.K., Ramteke, P.W. (Eds.), Green Nanotechnology for Biofuel Production. Biofuel and Biorefinery Technologies. Springer, Basel, pp. 1–15.)*

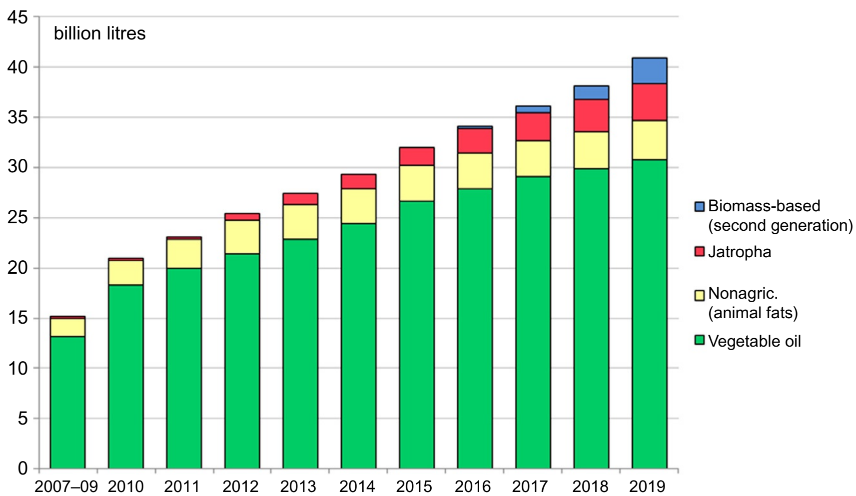

Fig. 1.3 Global biodiesel production by feedstock—projections (2007–19). *(Modified from OECD-FAO, 2010. Agricultural Outlook 2010. Biofuel Production 2010–19; Ziolkowska, J.R., 2018. Introduction to biofuels and potentials of nanotechnology. In: Srivastava, N., Srivastava, M., Pandey, H., Mishra, P.K., Ramteke, P.W. (Eds.), Green Nanotechnology for Biofuel Production. Biofuel and Biorefinery Technologies. Springer, Basel, pp. 1–15.)*

production. However, their use has increased over time and could even become of a higher importance in the future. Biodiesel production from animal fats, however, has remained rather stable over time in terms of the percentage share in the total biofuels production (Fig. 1.3).

2.1 Conventional (first generation) biofuels

The first attempts of biofuels engine operation (peanut oil engine run by Rudolf Diesel in 1900 and vegetable oil run engines in 1930s) as well the first industrial biofuels were based on food crops (Ziolkowska, 2018). In the past decades, food crop application for biofuels production has increasingly been criticized due to two major issues: "fuel vs. food" trade-off and concerns about the real CO_2 reduction potential of biofuels (some biofuels could release more carbon in their production process than sequester it in the feedstock growth process).

Because of these urgent issues, most studies in this area focus on competition for resources resulting from crop cultivation and their application either for food/feed or biofuels production. This trade-off situation for food, feed, fuel, and production factors can impact producers, distributors, and the related markets, and finally regional and national economies (Tomei and Helliwell, 2016; Baffes, 2013; Filip et al., 2017). Most attention in the literature has been given to land resources (Rathmann et al., 2010; Harvey and Pilgrim, 2011) and impacts of biofuels production on food market prices (Aké, 2017; Enciso et al., 2016; Tyner, 2013; Ajanovic, 2011).

Conventional biofuels encompass ethanol (produced from crops with high sugar contents, e.g., corn, cereals, sugar beet/sugar cane) and biodiesel (produced from high oleic plants, e.g., soybean, rapeseed, palm oil, animal fats, waste oils).

In the past decades, conventional biofuels have developed into flourishing fuel markets. In 2015 in the United States alone, the consumption of ethanol in BTU energy units (1 BTU $=$ 1055 J) amounted to 1.14 quadrillion BTU, while biodiesel consumption totaled 0.26 quadrillion BTU. The total capacity of ethanol consumption was estimated at 15 billion gallons (57 billion liters), while 2 billion gallons (7.6 billion liters) for biodiesel (US EIA, 2016).

Production of conventional biofuels has varied in different parts of the world, subject to feedstock availability. Global production of conventional biofuels for the transport sector reached 140 billion liters (37 billion gal) in 2017 (IEA, 2018). In 2015, Brazil and the United States accounted for ~70%

of the global biofuel supply of sugarcane- and corn-based ethanol (REN21, 2016; Araújo et al., 2017). Other suppliers, that is, European Union countries and Asia have entered the biofuels market in the last two decades. Biofuels production in the European Union is mainly based on biodiesel from waste, soybeans, rapeseed, and palm (Huenteler and Lee, 2015), while in the Americas and Asia ethanol production is prevailing with the following feedstocks: sugarcane, corn, wheat, and cassava. In Asia, additional efforts and investments in recent years have contributed to a growing biodiesel market utilizing palm, soybean, rapeseed, and Jatropha feedstocks. The regional and feedstock diversification has been recognized by several German associations and agencies (GTZ, 2006) as potentially conducive to the formation of an international biofuel commodities market.

2.2 Advanced biofuels

The development of advanced biofuels was propelled in response to concerns related to the "fuel-food tradeoff" as well as environmental and economic questions surrounding conventional biofuels (UN Report, 2007). By utilizing biomass (not suitable either for food or feed purposes) and in many cases grown on marginal lands, the problem of resource competition in food/fuel production could potentially be mitigated to some degree. At the same time, emerging recognitions and new knowledge about energy value of biofuels (compared to fossil fuels) spurred questions about economic efficiency of biofuels in general (Czekała et al., 2018). For instance, production of cellulosic biofuel is highly energy intensive meaning that energy contained in this type of biofuel is lower than the energy required for its production (Ge and Li, 2018).

Environmental questions about advanced biofuels relate directly to CO_2 emission reduction. Many studies provided evidence that biofuels contribute to CO_2 emission reductions in the fuel burning process (Mendiara et al., 2018; Kousoulidou and Lonza, 2016; Subramanian et al., 2018). However, it needs to be emphasized that the exact emission reduction levels strongly depend on the applied feedstock, with algae being acknowledged among the leading feedstocks (Shuba and Kifle, 2018; Su et al., 2017; Savakis and Hellingwerf, 2015) with carbon negative properties (Ziolkowska and Simon, 2014). However, concerns have been raised about other biomass feedstocks (e.g., timber) pointing out that forest bioenergy is not carbon neutral due to high CO_2 emissions released in the wood burning process (Moomaw, 2018). According to US EIA (2016), the consumption of wood/forestry biomass (including wood pellets, hog fuel, and wood chips) utilized for electricity and heat production in BTU energy units is larger than bioenergy from conventional biofuels. In the United States alone, wood biomass consumption amounted to 2.04 quadrillion BTU in 2015, while it totaled 11 million tons in wood pellet capacity.

2.2.1 Cellulosic ethanol (second generation biofuels)

Cellulosic ethanol can be produced from any material containing cellulose and lignocellulose. The main feedstock sources for cellulosic ethanol production can be divided as follows:

(a) Energy crops grown specifically for the purpose of conversion into biofuels (e.g., switchgrass, miscanthus, wheat straw, poplar, willow, jatropha).

(b) Green waste used as a by-product of other production processes (e.g., corn stover and other field residue, e.g., stalks and stubble (stems), leaves, seed pods, as well as forest/park residues).

According to Chen et al. (2010), 40%–70% of hemicellulose and 72%–90% of cellulose in corn cobs could be converted to ethanol using different bacteria and fungi. Also application of more unconventional feedstocks containing cellulose or lignin (e.g., kapok fiber, pineapple waste, waste papers, and coffee residue waste for bioethanol production) has recently been investigated (Dutta et al., 2014; Choi et al., 2012; Ruangviriyachai et al., 2010; Chen et al., 2010).

The question of economic efficiency of the second generation biofuels remains open due to high costs related to breaking down cellulose, making it a lesser competitive feedstock and biofuel in general compared to fossil fuels. Although many industrial and laboratory attempts have been undertaken in the past decade to lower the production costs of cellulosic ethanol, the experiments were not as successful as initially anticipated, with the average price for cellulosic ethanol still not being competitive enough with traditional gasoline. As of 2010, production costs of cellulosic ethanol equaled to \$2.65/gal of fuel (Coyle, 2010), which was ~\$1 more than costs of corn ethanol. The more recent research studies and scenarios by the National Renewable Energy Laboratory (NREL) have proven that cellulosic ethanol could be cost competitive at \$2.15/gal (NREL, 2013). Due to this economic limitation determining the market access, most studies in this area are focused on improving technological processes of cellulose decomposition and breakdown (Liu and Bao, 2017; Gao et al., 2018; Shadbahr et al., 2018; Song et al., 2018). Many studies attempted to provide solutions to high costs of second generation biofuels by introducing microbial or fungal systems facilitating more effective and faster cellulose breakdown and fermentation process (Bhatia et al., 2017; Ziolkowska, 2014). However, research and development in this field is ongoing and no wide-scale commercial solution has been introduced, which again, will depend on the respective feedstocks and their cellulose and lignin contents.

An advantage of advanced biofuels is that feedstocks used for their production generally generate greater greenhouse gas emissions savings, and thus are more sustainable and desirable from the environmental point of view. For this reason, in the United States, with the 2007 Energy Independence and Security Act, the Renewable Fuel Standards (RFS) were introduced as a mandate to expand the quantity of renewable fuels blended into transport fuel from 9 billion gallons (34.07 billion liters) in 2008 up to 36 billion gallons (136.27 billion liters) in 2022 (Ziolkowska, 2018; Ziolkowska et al., 2010). Within these totals, starting in 2015, only 15 billion gallons (56.78 billion liters) can be provided on the market from conventional ethanol, while the remaining annual mandated quantity needs to be supplied from advanced feedstocks. In April 2010, the RFS2 was enacted by the EPA as an extension of the original mandate specifying minimum quantities from different feedstocks or biofuel types needed to be blended toward the total mandate (FAPRI, 2010; Ziolkowska et al., 2010). Accordingly, the cellulosic ethanol production was mandated to increase each consecutive year with the goal of 16 billion gallons (60.5 billion liters) in 2022 (US EPA, 2010). Furthermore, cellulosic ethanol was assigned a Life Cycle Assessment requirement to be effective with reducing GHG emissions by at least 60% compared to the emission levels generated from combustion of traditional gasoline (i.e., fossil fuels used in transportation) (Table 1.1). Due to the 2007 Energy Independence and Security Act and renewable fuel standards established as mandates, production of cellulosic ethanol and compliance with its supply for blending has been mainly discussed in the United States. In Europe, where biofuels policy is based on voluntary targets rather than mandates, cellulosic ethanol production took off at a later time and has gained less attention in general.

It needs to be mentioned that in addition to bioethanol production from the second generation feedstocks, also other advanced biofuels (isopropanol, butanol, isobutanol, and farnesol) have been gaining on importance due to their high energy density as well as lower hygroscopic properties and lower corrosity to pipelines during transportation than other fuels (Chen et al., 2013; Yua et al., 2011). In addition, metabolic engineering of biosynthetic fuels can lead to even greater productivity of these alcohols.

Table 1.1 Requirements for new standards under RFS2

Type	Volume by 2022	Lifecycle GHG threshold	Comment
Biodiesel	1 billion gal (3.79 billion l)	50%	For 2012 and beyond[a]
Cellulosic biofuel	16 billion gal (60.57 billion l)	60%	Subject to annual assessments
Advanced biofuel	21 billion gal (79.49 billion l)	50%	Anything but corn starch, minimum of 4 billion gal additional
Renewable biofuel	36 billion gal (136.27 billion l)	20%[b]	Minimum of 15 billion gal additional

[a] Could be increased from 2013 onward.
[b] Only applies to fuel from new facilities. "Grandfathered" facilities are those (domestic and foreign) that commenced construction before 31 December 2007 and ethanol facilities that commenced construction prior to 31 December 2009 and usenatural gas and/or biomass for process heat.
Data from US Environmental Protection Agency (EPA), 2010. National Renewable Fuel Standard Program—Overview. Office of Transportation and Air Quality, US EPA, Washington, DC, April 14; Ziolkowska, J., Meyers, W.H., Meyer, S., Binfield, J., 2010. Targets and mandates: lessons learned from EU and US biofuel policy mechanisms. AgBioForum 13(4), 398–412.

2.2.2 Algae biofuels (third generation biofuels)

The third generation of biofuels aims at improving the production of biomass to make it a more viable (and sustainable) feedstock. Since the beginnings of this technology, the third generation biofuels have relied on algae as the main feedstock (grown either naturally or artificially). Many studies confirmed that the algae feedstock can be competitive with other biomass sources (Jones and Mayfield, 2012; Ziolkowska and Simon, 2014; Laurens et al., 2017; Adeniyi et al., 2018), thus making it, in many cases, more prospective for company investments than cellulosic ethanol. The advantages of algae as a feedstock relate to:

(a) Negative (carbon neutral) environmental footprint as by growing algae 2 g of CO_2 are consumed for every g of generated biomass (Pienkos and Darzins, 2009). At the same time, one ton of CO_2 can be converted into 60–70 gal of algae-based ethanol (Hon-Nami, 2006; Hirayama et al., 1998).

(b) Possibly no competition for fresh water as algae can grow in waste/saline water environment.

(c) No competition for fertile land (i.e., no direct food-fuel trade-off) as algae is grown in closed photobioreactors or open ponds (water environments) which can be located on any plot of land not suitable for other purposes, which thus eliminates potential opportunity costs (Ziolkowska and Simon, 2014).

(d) High oil contents in algae biomass make it suitable to produce 10–100 times more oil per acre than traditional oil crops (such as oil palm)

(e) Fast growing rate as algae can grow 20–30 times faster than food crops (Ziolkowska and Simon, 2014).

(f) High fuel diversity as algae biomass can be converted into a multitude of fuel types, such as diesel, petrol, and jet fuel (see also Jones and Mayfield, 2012).

(g) High nutritional diversity of the feedstock as it can be processed both through sugar and oil processing procedures to extract sugars/oils for biofuels production.

(h) High compatibility with traditional gasoline engines (thus eliminating the need of automobile engine adjustments) due to the same biochemical characteristics and composition (energy density, number of carbon atoms per molecule) as present in gasoline (Solazyme, 2012).

Despite the many advantages of algae biomass and algae-based fuels, its economic feasibility has been questioned and challenged many times (Doshi et al., 2016; Vassilev and Vassileva, 2016). Also, economic and policy issues have been pointed out as possible determinants of future developments (Doshi et al., 2016). In 2008, the price for algae-based fuels amounted to approximately \$8/gal (US DOE, 2008), while there is no uniform market estimate as the final price is determined by each producing company subject to the applied technology and production factors. For many decades, the industry has struggled with bringing down the production cost and thus the final price of algae-based fuels through reducing costs of systems infrastructure and integration, algae biomass production process, harvesting and dewatering techniques, extraction and fractionation, and finally biofuels conversion process (US DOE, 2010). Sustainable or market competitive solutions have not been found to date to make algae-based fuel a viable and desirable fuel due to the high fuel unit costs.

2.2.3 Future technology (fourth generation biofuels)

The fourth generation biofuels are in the development and experimental stages, thus they combine a diversity of different (potential) applications both on the technology, processing, and feedstock level.

The main feedstock for the fourth generation biofuels production is genetically engineered, highly yielding biomass with low lignin and cellulose contents (thus eliminating the issues present in the second generation biofuels production line) or metabolically engineered algae (with high oil contents, increased carbon entrapment ability, and improved cultivation, harvesting, and fermentation processes) (thus improving the third generation production) (Dutta et al., 2014). While algae have commonly been recognized for its high oil contents, the exact parameters depend on the respective algae strains. *Botryococcus braunii, Chaetoceros calcitrans, Chlorella* species, *Isochrysis galbana, Nannochloropsis, Schizochytrium limacinum and Scenedesmus* species have been analyzed in the literature so far for their applicability and suitability for biofuels production (Chisti, 2007; Rodolfi et al., 2008; Singh and Gu, 2010). It has been found that the fast growing algae (e.g., Spirulina) have low oil content, while algae strains high in lipid contents are characterized by slower growth rates. Thus introducing new technologies like metabolic engineering for accelerated growth of algae biomass or increased lipid contents can result in faster commercialization and improved economic feasibility of fourth generation biofuels (Singh and Gu, 2010). Nanotechnology could also be applied in algae fuel production to increase efficiency of algae biomass and decrease production costs, thus making it a cost-competitive addition to the biofuel market (Ziolkowska, 2018).

The fourth generation biofuels is distinguished from other biofuels production technologies also by the fact that in most cases they represent a combination of different technologies, for example, sustainable energy production (biofuels) and capturing and storing CO_2 emissions. Biomass absorbing CO_2 during its growth is manufactured into biofuel by means of the same or similar processes as second generation biofuels. The difference between the fourth generation biofuels compared to the second and third generation production is that the former captures CO_2 emissions at all stages of the biofuels production process by means of oxy-fuel combustion (Oh et al., 2018; Sher et al., 2018). Oxy-fuel combustion is a process utilizing oxygen (rather than air) for combustion yielding flue gas CO_2 and water (Markewitz et al., 2012). While the process is more effective in generating CO_2 stream of a higher concentration (the mass and volume are reduced by about 75%), making it more suitable for carbon sequestration, the economic problem occurs mainly at the initial stage of separating oxygen from the air and using it for combustion. The process requires high energy inputs; nearly 15% of production of a coal-fired power station can be consumed for this process (University of Edinburgh, n.d.), which can ultimately drive up production costs and make the final process economically infeasible. Even though currently still not competitive, oxy-fuel combustion has been studied as a potential alternative in combination with biofuels production. For this reason, this technology is in the developing stage as of today. However, if successfully validated in the future, it could be used to geosequester CO_2 by storing it in old oil and gas fields or saline aquifers. In this way, through carbon capturing and storage, the fourth generation biofuels production could be called carbon negative rather than carbon neutral. Thus environmental advantages arise both from carbon storage and from replacing fossil fuels with biofuels (University of Edinburgh, n.d.).

The remaining fuel from oxy-fuel combustion is cleaned and liquefied and yields ultraclean biohydrogen, biomethane or synthetic biofuels that can be used in the transport sector as well as for electricity generation.

Another potential technological combination for biofuels production has been proposed by the Joule company with their renewable solar fuel generation (Fig. 1.4).

The company developed a process for hydrocarbon-based fuel generation through the application of nonfresh water, nutrients, cyanobacteria, carbon dioxide, and sunlight. The process is based on helioculture using photosynthetic organisms; however, it is distinct from the traditional algae-based fuel in that the latter need to be refined into fuel, while helioculture directly produces fuel (either ethanol or hydrocarbons) not requiring

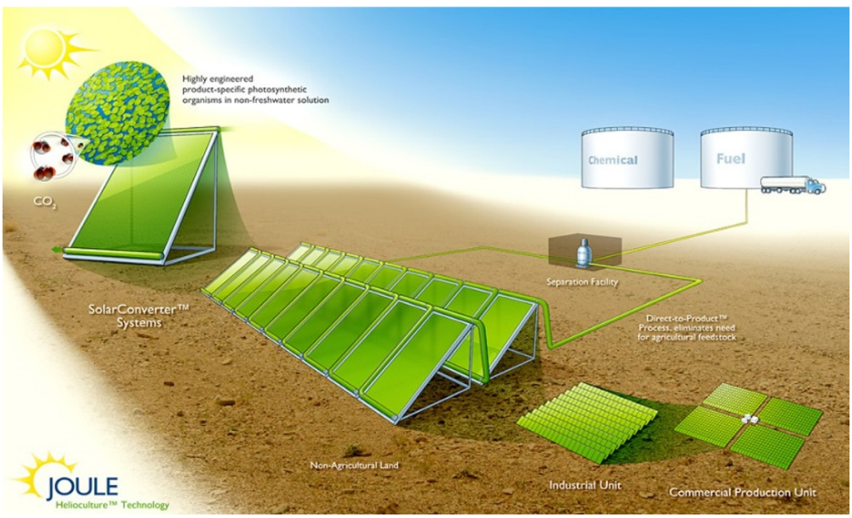

Fig. 1.4 Joule helioculture renewable solar fuel. *(From St. John, J., 2010. Joule Patents Secret Sauce for Diesel-Excreting Organisms. 2010. GigaOm, September 14. https://gigaom.com/2010/09/14/joule-patents-secret-sauce-for-diesel-excreting-organisms (24 November 2018).)*

any refinement. The process does not produce biomass either, thus making the technology easier to apply in practice. Although the company was discontinued its operation in August 2017 due to difficulties with raising additional funds for future developments, the suggested innovation based on helioculture presents an attractive technological attempt. The company claimed to be able to produce more than 20,000 gal of fuel per acre per year (19,000 m^3/km^2). The economic estimates by Joule Unlimited claimed its product to be cost competitive with crude oil at \$50 a barrel (\$310/m^3) (St. John, 2010).

Moreover, nanotechnology has also been considered as a technological solution to alleviate challenges related to algal biomass growth and cultivation (Sekoai et al., 2019; Gavrilescu and Chisti, 2005), mainly high costs of algae harvesting and production as well as energy-intensive lipid extraction (Pattarkine and Pattarkine, 2012). A new form of "nanofarming" technology is currently in the pilot stage and could find wide commercial application. It facilitates oil extraction from algae even more efficiently as it relies on a process of "milking algae," thus using biomass continually (up to 70 days) rather than destroying it as is the common case with conventional material science processes (Vinayak et al., 2015; Chaudry et al., 2016; Ziolkowska, 2018).

3 Biofuels processes

From the technological perspective, four main processes can be distinguished for biofuels production:

(a) *Mechanical processes* involving traditional processing of wood materials through mechanical treatment, for example, chipping or grinding, and potentially the following densification of the material by pelletizing the biomass.

(b) *Thermochemical processes* converting biomass into energy through combustion, followed by pyrolysis. This process is more efficient than mechanical processes due to greater energy density as well as chemical and physical fuel properties being more similar to fossil fuels. Another possible process is gasification generating syngas for the production of different liquid biofuels, through the Fischer-Tropsch (FT) process.

(c) *Chemical processes* are used mainly for the production of transportation fuels, such as biodiesel and cellulosic ethanol.

(d) *Biochemical processes* applied to produce fuel ethanol, for example, sugar/starch fermentation leading to biogas (methane) generation in anaerobic conditions.

In the production process of first generation biofuels, mainly sugar fermentation followed by the distillation process is applied to produce dry ethanol. Biodiesel is produced through transesterification of oils with a chemical catalyst (acid/alkali) or enzyme (Dutta et al., 2009; van Gerpen et al., 2002), followed by a two-step distillation to remove by-products (e.g., glycerol).

In regard to the second generation biofuels production (e.g., cellulosic ethanol, butanol), the following processes are applied: pretreatment to separate cellulose and hemicellulose from lignin, enzymatic hydrolysis (i.e., saccharification) to extract simple sugars, fermentation, and finally distillation. Also biogass can be produced through the same sequence of the biochemical processes up to distillation. Accordingly, gasification or pyrolysis (thermochemical processes) is applied to convert biomass at higher temperatures and pressures than those applied in biochemical processes. The process of biomass gasification and direct liquefaction is commonly called "biomass-to-liquid." Gasification is more cost intensive than other processes; however, it produces cleaner fuel that can be directly used in engines (Larson, 2007). Through the gasification process, a variety of biofuels can be produced, such as Fischer-Tropsch liquids (FTL), dimethyl ether (DME), and other alcohols (FitzPatrick et al., 2010; Dutta et al., 2014).

The third and fourth generation biofuels can be extracted with the same processes, while the difference relates only to the feedstock used at the input stage (traditionally cultivated algae for the third generation fuels, and genetically modified algae biomass for the fourth generation fuels). Thus different final industrial fuels can be extracted in different processes of the third and fourth generation biofuels production, as follows:

(a) Biodiesel through oil extraction, transesterification, and distillation.

(b) Ethanol, biomethane, and biobuthanol through biochemical processes, while biomethane and biobuthanol are processed in addition through anaerobic digestion.

(c) Syngas, synthetic diesel (aviation fuel), and bioenergy through the thermochemical process of gasification (Dutta et al., 2014).

4 Summary and conclusions

Biofuels production has faced many sustainability challenges over the decades of technological, feedstock, and process developments. High costs of feedstock processing and environmental concerns have been among the major issues discussed in scientific debates and policy considerations. While conventional biofuels were introduced with an effort to increase energy independency from fossil fuels (and foreign fuel imports), advanced biofuels emerged as innovation spillovers and expanded at a progressive pace. Accordingly a variety of new feedstocks has been explored and experimented with to alleviate economic limitations to those new technologies. The main purpose of these investments was to bring down the production costs and the final price of advanced biofuels, with the aim of making them competitive with fossil fuels as transportation fuel and bioenergy. Although a considerable success has been achieved in these areas, challenges still exist. New technological inventions, such as nanotechnology and genetic engineering of biofuel feedstocks could prove as a viable solution. However, environmental and social issues of these technologies (including unexplored consequences of their application and potential impacts on water resources, soil, and the following influence on humans) have not been fully explored yet. Social resistance to these technologies (as currently also analyzed in the food sector) might be decisive in the mid- and long-term also for biofuels production.

Governmental and private funding for research and development of new biofuels technologies can bring about prospective solutions to those questions and make new generation biofuels more economically feasible, environment friendly, and socially acceptable.

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CHAPTER 2

Biofuel transitions: An overview of regulations and standards for a more sustainable framework

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1 Introduction

Climate change, environmental degradation, air pollution, and the expected world population growth make the transition out of a fossil-based economy into one based on biomasses of outmost importance and urgency.

In the latest report of Intergovernmental Panel on Climate Change it is stated that to achieve the global warming target of 1.5°C, the CO_2 emissions must be reduced by about 45% of the 2010 levels by 2030 and reaching zero by 2050 (IPCC SPM, 2018, p. 15). The situation is also extremely challenging when it goes to resources scarcity. Although material productivity

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increases in the majority of countries, relative improvements are overcompensated by economic and population growth. These lead to an overall increase in terms of absolute levels of material consumption, associated with environmental problems including climate change and materials and water scarcity (Giljum et al., 2014).

As it seems, this transition involves both the energy sector as well the material and goods production sector and should combine new production models with new consumption models oriented toward reduction, reuse, and sharing practices.

If realized, such a wide-ranging transition will bring the world into the bioeconomy era, one where well-being is decupled from environmental unsustainable exploitation, and economic growth is not the only dominant paradigm. Therefore the foreseen transition is not about doing the same things in a different way, but rather doing different things in a different way to get a better quality of life for the most and not for the few.

For this transition to happen, efforts are needed from all involved players: researchers and industries which have to develop new production models, consumers which have to adopt new consumption practices, and policy-makers which have to develop a favorable regulatory and legislative environment for such changes to occur and deliver the aimed for environmental, economic, and social gains.

In this chapter we will look closely at this latter element of the transition, specifically assessing the impact that standards and regulation played on the biofuel sector. We will identify pros and cons associated with such intervention, aiming at drawing some lessons applicable broadly to the bioeconomy transition.

2 Defining and mapping biofuels and their markets

Biofuels include any liquid or gaseous fuels made wholly or in part from biomass (Johnson et al., 2012, p. 2); as such, the development of biofuels as an alternative resource is driven by a decrease in the availability of oil resources per capita, the increasing costs and difficulties to reach oil reserves that are located in geopolitically unstable regions (Pfau et al., 2014), and their socio-environmental benefits associated with green jobs, energy security, and decarbonization of the economy.

Based on the biomass used for their production, biofuels can be classified into three groups: first-generation biofuels, second-generation biofuels, and algae-based biofuels.

First-generation biofuels include bioethanol or butanol, which utilizes the sugar and starch portion of plants (e.g., cereals, sugarcane, sugar beet) as biomass, and biodiesel that is produced with oilseed crops (e.g., rapeseed, soybeans, sunflowers, coconut oil, palm oil, jatropha, recycled cooking oil, and animal fat) as biomass (UNCTAD, 2016). For ethanol or butanol produced with sugar, simple sugars are extracted from a variety of sugar crops and then fermented; while for starch ethanol or butanol the process is more complex since starch is converted into simple sugars through a high heat enzymatic process that requires additional energy. Bioethanol instead is produced by mixing lipids present in the oilseed crops with an alcohol, for example, methanol, ethanol, through the chemical process of transesterification. Whereas ethanol or butanol generates 70% less energy than gasoline, bioethanol produces 88% to 95% of the energy content of conventional diesel (Timilsina and Shrestha, 2010).

Bioethanol production reached 115.6 billion liters in 2015, more than double since 2005, and is expected to grow to nearly 128.4 billion liters by 2025 (Purohit and Dhar, 2018). The United States and Brazil have been the leaders of bioethanol production and exports, with 57% of global bioethanol production in the United States and 29% in Brazil. The EU is a net importer of bioethanol, producing only 4% of total bioethanol (EC et al., 2015); while other countries (China with 3%, Canada with 2%, Thailand with 1%, and the rest of the world with the remaining 4%) produce bioethanol mainly for domestic use. Different countries use different feedstock for the production of bioethanol: Brazil and India use mainly sugarcane, North America and China starch crops (mainly corn), and the EU sugar beet and grains (EC et al., 2015).

As far as biodiesel is concerned, its production reached 31 billion liters in 2015, compared to the 3.9 billion liters produced in 2005, and is expected to grow to 41.4 billion liters by 2025 (Purohit and Dhar, 2018). The EU produces 38% of global biodiesel production but is also the main importer from Argentina, Indonesia, and Malaysia. The United States contributes with 16% to the global biodiesel production and imports most of the biodiesel from Canada and exports mainly to Canada and Norway (EC et al., 2015). Brazil produces 14% of the global biodiesel production, Argentina produces 7%, Indonesia contributes with 6% of the production, Malaysia with 2%, and the rest of the world with 17% of the total. India accounts for less than 1% of global biodiesel production (Purohit and Dhar, 2018). The feedstocks mainly used for biodiesel are soybean in the United States and Argentina, rapeseed and sunflower in the EU, and palm and coconut oil in Indonesia and Malaysia (EC et al., 2015). In 2015 almost all biodiesel in China was produced from waste cooking oil (WCO), whether in India from WCO, animal fats, and other oils (Beckman et al., 2018, pp. 259–260).

Second-generation biofuels are produced from lignocellulosic biomass and waste-based materials, for example, agricultural and forest residues, municipal solid wastes (UNCTAD, 2016). The lignocellulosic biomass is considered advanced feedstock; instead of using sugar or starch fractions of plants, the lignocellulosic conversion process utilizes entirely the lignocellulosic material contained in residues and waste. The lignocellulosic biomass is composed of polysaccharides that are converted into sugars through hydrolysis and/or chemical processes and then fermented into ethanol (Timilsina and Shrestha, 2010).

Although the residues and nonfood biomass conversion technologies have been available since the beginning of the 21st century, no industrial production of second-generation bioethanol was attainable until 2008. By that time, there were only 15–20 companies, located mainly in the United States, involved in pilot-scale plants using different biotechnological and thermochemical biomass conversion processes (Timilsina and Shrestha, 2010). Second-generation bioethanol production technologies are more complex and expensive than the first-generation ones; however, they are considered more sustainable since there is no need for dedicated crops, thus undermining indirect land use change (iLUC) and related social and environmental issues (see session 3 for further discussion). The production of second-generation biofuels from lignocellulosic biomass has increased significantly since 2012, although the overall volumes remain below the production capacity and the environmental needs. The United States produced only 2 million liters, China reached 3 million liters of ethanol from maize cobs for use in blends with petrol, and the EU had only some demonstration plants available by that time (EC et al., 2015).

The algae-based biofuels, also called third generation biofuels, are fuels produced with algae-based biomass. The latter can be cultivated specifically for biofuel production or are collected from polluted aquifers, generating little pressure on arable land (Timilsina and Shrestha, 2010). The utilization of microalgae for biofuels production is based on the lipid content of the microorganisms (reaching 60% to 70%) and has high productivity (7.4 g/L/day) (EC et al., 2015). The conversion of algae biomass into biofuels follows the transesterification process of lipids as for the biodiesel production.

Although biofuels production from algae biomass is at an early stage, thus uncertainties and costs of production are high, the potential yields for algae are considerably higher (45,000 liters of biodiesel/ha) compared to yields of oilseed crops (only 1500 liters of biodiesel/ha from rapeseed and 2500 liters of bioethanol/ha from maize) (EC et al., 2015).

Countries, both from developed and developing regions, have engaged dynamically in the development and deployment of biofuels. In particular, the United States, the EU, Brazil, China, and Canada contribute to the majority of biofuels world trade (UNCTAD, 2016). Yet, the trade share of each country is highly dependent on tariff barriers, for example, India has established high imports tariffs to protect domestic agriculture and biofuel industries, while OECD countries have imposed low import tariffs (Timilsina and Shrestha, 2010). In addition, also sustainability regulatory measures have limited the trade of biofuels, for example, the EU sustainability criteria for imported palm oil from Malaysia and Indonesia. All in all, biofuels global trade is expected to increase in the near future due to an increase in production and a decrease in production costs. Biofuels production has the potential to cover the rapidly increasing demand for transport biofuels, which will reach more than a quarter of the total transport fuel by 2050 (EC et al., 2015). This overview on biofuels production shows a supremacy in the production and trade of first-generation biofuels, which are however characterized by relevant sustainability issues (to be discussed in session 3), and call for policies, regulations, and standards (to be discussed in session 4).

3 Economic, social, and environmental issues associated with biofuels (production and consumption)

While the need for a sustainable economic growth that does not rely heavily on fossil-based resources remains the main driver for the development of biofuels, and of bioeconomy in general, their large (industrial) scale diffusion is not exempt from socioeconomic and environmental issues.

On the one hand, biofuels contribute to economic, social, and environmental sustainability. Indeed, biofuels guarantee energy security by replacing the scarce resources of fossil fuels and providing domestic sources of fuel instead of imported ones with increasing costs of exploitation. In addition, biofuels have a great potential to reduce lifecycle GHG emissions with respect to conventional fuels (Moioli et al., 2018); this happens particularly for second and third generation biofuels which use feedstock produced in marginal lands or waste biomass (Huang et al., 2013).

Moreover, the development of biofuels could generate important socioeconomic benefits for rural and local economies. Since biomass availability is geographically widespread, and new power plants have to be constructed for their treatment, the production of biofuels can potentially create new employment opportunities, increase income, and supply local goods and services (Demirbas, 2017). In addition, economic benefits of biofuels production range from value added to feedstock and market opportunities for exceeding crops, to investments in new plants and equipment, new jobs and income opportunities for rural communities, and reduced dependence of local and national economy from conventional fuels imports.

In spite of these advantages (in terms of environmental sustainability), the current demand for biofuels relies heavily on policy support, such as obligatory quotas (Horvat et al., 2017; further explained in Section 4), due to their low competitiveness (with respect to conventional fuels) related mainly to production and processing costs, and technological infrastructures for second and third generation biofuels.

Biofuels production costs are determined by several factors including the type of feedstock, agricultural practices in terms of land and labor costs, processing technologies, and plant size. The latter is particularly important with regard to economies of scale; due to uncertain availability of biomass resources, power plants are typically built for small-scale operations. Moreover, biofuels have low energy density and high costs of collection and transportation of biomass.

Overall, the costs of producing biofuels in OECD countries are three times higher than conventional fuels; unlike developing countries, where the production costs of biofuels are close to those of conventional fuels (Demirbas, 2017) due to the large availability of domestic biomass, low levels of per capita consumption, and low cost of labor (Gomiero, 2018).

To reach the 6% reduction target for GHG emissions for fuels and expand the market for biofuels, fuel-blending obligations and subsidies have been introduced. However, the maximum percentage of biofuel content in petrol and diesel compatible with engines and vehicles is 10% (COM, 2017, 284 final). For higher blends and other marketing options, new vehicle standards, adaptation of engines and vehicles, and new fuel distribution infrastructures are needed.

In the United States, biofuels are not competitive even when a barrel of oil costs more than US$100 (Gomiero, 2018). Additionally, conventional fuels price fluctuations strongly influence the development and marketability of biofuels (Morone and Cottoni, 2016). Therefore economic subsidies and environmental obligations remain fundamental supporters for biofuels development. All in all, the demand for biofuels is expected to increase up to 37% by 2040 (Joshi et al., 2017; IEA, 2014).

On the other hand, the production of biofuels has revealed controversial socioenvironmental issues. Several scholars (see among others, Basili and Rossi, 2018; Gomiero, 2018; Harris et al., 2015; Mosnier et al., 2013; Melillo et al., 2009; Searchinger et al., 2008) claim that biofuels have a limited contribution to GHG emissions reduction, and socioenvironmental sustainability in general. These claims are grounded on the effects generated by deforestation, loss of biodiversity, application of fertilizers for fuel crops, and indirect land use change.

Deforestation is a critical issue in countries that are converting forests into plantation of fuel crops, such as soybean in the Amazon and oil palm in Southeast Asia (Fargione et al., 2008). Forest conversion into plantation has a twofold negative impact on the environment; on the one hand, reduced forest surfaces help capturing less carbon from the atmosphere, and on the other hand, deforestation together with monoculture plantations may be harmful for indigenous people, biodiversity and rare species, and soil erosion (Jefferson, 2018; Obidzinski et al., 2012).

Additional negative environmental impacts of biofuels production concern water pollution either from nutrients, pesticides, and sediments, or from crop lands irrigation and biofuels refining (NRC, 2011). Also, fertilizer application, by releasing nitrous oxide, increases GHG emissions.

Key socioeconomic and environmental aspects negatively influenced by the production of (mainly first generation) biofuels include: (i) indirect land use change (iLUC), food crop prices, and food security (to be discussed in Section 3.1); and (ii) equality and gender issues due to lack of access to resources associated with increasing land pressure (to be discuss in Section 3.2).

3.1 Sustainability issues: From food security to nonfood resource biorefineries

The competition for land between food crops and fuel crops is one of the main sustainability issues associated with biofuels production. The demand for food, water, and energy is expected to increase continuously with the world's population growth. This coupled with a growing demand of biofuels and limited agricultural land has established the controversy food vs. fuel with potential impacts on iLUC and food security. Because of higher and safer incomes and employment opportunities, farmers are more likely to convert their food crop production into fuel crops. As a result, food availability (associated with food security) decreases and food prices increase (Joshi et al., 2017; Demirbas, 2017; Luthra et al., 2015; Obidzinski et al., 2012).

Although the link between the market expansion of biofuels and food prices seems to be blurred, Paris (2018) has shown a long-term relation between biofuel production and the prices of several agricultural commodities. The author has found evidence of a long-term effect of rising oil prices on agricultural commodities price used for biofuels. Hence, this effect is transmitted to other agricultural commodities through the substitution effect. As claimed by the author, in the absence of biofuels production, changes in oil prices do not affect agricultural commodities in the long run, with the exception of rapeseed price that is affected by oil prices in the long run (Paris, 2018).

Indeed, several other studies estimated the effect of biofuels production on crop prices. Zhang et al. (2013) projected an increase of 5% to 53% on corn prices in 2015. In addition, evidence was found for on average 2% to 3% increase in long-term corn prices, for each billion gallon of corn ethanol produced across 19 studies (Condon et al., 2013).

The impact of biofuels production on food prices affects mostly developing countries where higher crop prices may lead to malnutrition and starvation (Raman et al., 2015; Obidzinski et al., 2012). Furthermore, incentives favoring the production of feedstocks in these countries and their exportation in richer ones for biofuels production may generate location-specific environmental risks and energy insecurity (Raman et al., 2015).

These sustainability issues are less stringent for second and third generation biomasses, based on residues, nonfood crops, and algae (Paris, 2018). Moreover, the development of technologies for using waste as biomass can contribute to the advancement of circular bioeconomy, minimizing the social and environmental risks.

3.2 Rural development

Biofuels production has many benefits on rural development. Growing and harvesting biomass, transportation, and new plant operations have potential for new employment opportunities (Demirbas, 2017). In addition, farmers have the opportunity to increase their income and return as the market for agricultural and forest residues expands and the production of biofuels increases. The latter can have a positive impact on traditional industries, rural diversification, rural electricity supply, and soil conservation. Also, environmental and landscape benefits can derive from a revitalization of degraded forests and the utilization of excessive waste streams for producing energy.

The production of biofuels can contribute to local energy security by revitalizing cultural traditions. "In the boreal forest, many remote communities have no year-round road or connections to electricity grids and are dependent on diesel generators supplied by fuel flown or barged in at high cost. These communities are often surrounded by forest that could provide the necessary biomass for energy generation, making the community more self-sufficient, reducing costs, providing employment, keeping wages and benefits within the community, and generally integrating well with a forest-based culture" (Demirbas, 2017, p. 50).

Looking at the other side of the coin, however, the production of crops for biofuels could also be detrimental to local communities (Obidzinski et al., 2012). Plantations of fuel crops may affect negatively social relations and land ownership in rural areas. Large companies involved in fuel crop plantations abuse local communities' human rights, particularly during the process of land acquisition and plantation development. Other conflicts involving plantation developers and local communities concern lack of recognition of customary rights, breached agreements, and disregard for the environment. "In 2010 no fewer than 630 land disputes between palm oil companies and local communities had taken place in Indonesia" (Obidzinski et al., 2012).

As it seems, pros and cons are associated with biofuel production. This calls for an effective regulatory framework able to maximize positive impacts minimizing, at the same time, potential negative effects. In what follows, the regulatory framework will be scrutinized in a comparative perspective looking both at EU and US regulatory frameworks. The EU/US comparative approach will be complemented by an overview of Chinese, Indian, and Brazilian regulatory frameworks. This study will allow pointing out lessons learned from alternative experiences and drawing conclusions on future developments.

4 The role of policy: Regulation and standards

4.1 The Europe framework

4.1.1 Source of law and motivation

As for the EU biofuels framework the sources of law can be divided into EU's law and member states' law. At the EU level biofuels are regulated mostly by directives which establish general goals and rules but simultaneously have to be implemented by member states in their own legal and administrative systems. At the states' level biofuels related legal acts can take the form of detailed statutes and executive acts holistically regulating biofuels production, distribution, and use. Such acts have to be consistent with EU directives and effectively realize their goals.

At Union level it is fundamental the Directive 2009/28/EC of the European Parliament and of the Council of 23 April 2009 "on the promotion of the use of energy from renewable sources and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC" (hereinafter referred to as "RED Directive").[1] The RED Directive regulates promotion of renewable energy sources in general, but the biofuels regulations are especially large.

The states level is constituted by different legal acts of each of 28 EU member states. For the purpose of this article the RES Legal database of biofuels support schemes in various member states was analyzed.

The RED Directive has a large preamble in which there are described general context, problems, and motivations for passing the act. The main motivation included greenhouse gases emission reduction, energy security, technological development and innovation, regional and local development, employment, rural development, social cohesion, SMEs, and independent producers' development. Issues concerning potential negative impact of biofuels on food production and prices, biodiversity loss, indirect land use changes, or raise of greenhouse gases emission are extensively discussed. Simultaneously, the directive proposes appropriate solutions like the introduction of sustainability criteria for biofuels production as well as the development of second and third generations biofuels.

4.1.2 Support schemes

The RED Directive established the binding goals of biofuels for member states; however, it did not introduce any specific support scheme leaving the choice in this matter to single states. As a binding goal, the Directive sets a minimum 10% level of final energy consumption in transport in each member state to be generated from renewable sources (not necessary biofuels). Simultaneously, in each member state "the share of energy from biofuels produced from cereal and other starch-rich crops, sugars and oil crops and from crops grown as main crops primarily for energy purposes on agricultural land shall be no more than 7% of the final consumption of energy in transport."[2]

[1] Consolidated version of the Directive from 2015, EUR-Lex, https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:02009L0028-20151005 (accessed 25 July 2018).

[2] Article 3, paragraph 4, point d of RED Directive.

Specific support schemes introduced at country level vary by different member states. Dominant support is provided through the introduction of obligatory quotas of biofuels in total amount of sold fuels, imposed on fuels manufacturers, distributors, and sellers (e.g., Austria, Belgium, Czech Republic, Hungary, Denmark, Finland, Germany, Great Britain, France, Italy, Poland). Alternative schemes include subsidies to production of raw materials and biofuels (e.g., Austria, Estonia, Greece, Lithuania), and tax reduction or exemptions on biofuels (e.g., Austria, Belgium, Czech Republic, Greece, Finland, Hungary, Netherlands).[3]

4.1.3 Standards of biofuels

The RED Directive, taking into account the problematic character of biofuels and other bioliquids, laid down five "sustainability criteria." Fulfillment of these criteria is necessary for biofuels to be credited toward national targets and received financial support.

First, the greenhouse gases emission reduction should reach at least 60% for installations started operating after 5 October 2015. The installations started on or before 5 October should reach 35% reduction until 31 December 2017 and 50% from 1 January 2018 (for comparison, the general reduction goal for traditional fuels is 10% in 2020[4]). The biofuels and bioliquids produced from waste and residues, other than agricultural, aquaculture, fisheries, and forestry residues have to fulfill only this criterion.

Second, the biofuels or bioliquids should not be produced from materials obtained from land with high biodiversity value. The Directive defined such lands by enumeration included, for example, primary forests, areas designated for nature, ecosystems or rare species protection, and highly diverse grasslands. However, the production from designated areas is allowed if evidence that such activities do not interfere with protection purposes is provided. Similarly, for nonnatural biodiverse grassland evidence that harvesting of raw material is necessary to preserve their status should be provided. Moreover, it is important to stress that status of the land is taken into account in January 2008 or after this date, whether or not the land will still have this status in the future. It seems that such regulations should prevent

[3] RES Legal, http://www.res-legal.eu/ (accessed 16 September 2018).

[4] Article 7a, paragraph 2 of Directive 98/70/EC of the European Parliament and of the Council of 13 October 1998 relating to the quality of petrol and diesel fuels and amending Council Directive 93/12/EEC, Consolidated version from 2015, EUR Lex, https://eur-lex.europa.eu/legal-content/EN/TXT/?qid=1537450042687&uri=CELEX:01998L0070-20151005 (accessed 20 September 2018).

deliberate actions of changing the status of protected land for pure economic motivation.

The third criterion states that biofuels and bioliquids should not be made from the materials obtained from land with high carbon stock. The RED Directive details the notion of such lands as lands which in January 2008 were wetlands or continuously forester areas (more than hectare, with trees higher than 5 m or able to reach in situ and canopy covered more than 30%) and do not have this status anymore. In the case of forested areas covered by canopies in 10%–30% exceptions are foreseen if evidence that use of the land is able to fulfill the first criterion (i.e., GHSs emission reduction) is provided.

The fourth criterion provides that the raw material should not be obtained from peatlands unless there is evidence that cultivation and harvesting will not lead to drainage of previously undrained soil.

The last criterion statutes that raw material production should be compliant with agricultural and environmental requirements of "Council Regulation (EC) No 73/2009 of 19 January 2009 establishing common rules for direct support schemes for farmers under the common agricultural policy and establishing certain support schemes for farmers."[5] Besides these, member states should not introduce additional sustainability criteria.

Every 2 years the Commission is required to report to the European Parliament and Council on the respect of sustainability criteria, influence of increased demand for biofuels on social sustainability, including "the availability of foodstuffs at affordable prices, in particular for people living in developing countries, and wider development issues," and provide information about ratification of third countries (exporters of biofuels materials) significant international conventions of, for example, labor laws.

When it goes to blending standards for individual fuels selling to consumers, the amount of ethanol in traditional petrol should not exceed 10% from 2013 and amount of FAME (fatty acid methyl ester) in traditional diesel fuel should not exceed 7% (e.g., E10 and B7).[6] However, the member states can require the 5% ethanol level for longer time than to 2013 if they consider it as necessary and can permit introduction of diesel with higher level of FAME than 7% but these actions should be taken providing appropriate information to consumers. Moreover, there are also other bioethanol

[5] The regulation itself is no longer in force. See the consolidated version from 2014, EUR-Lex, https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:02009R0073-20140925 (accessed 20 September 2018).

[6] Respectively Annexes I and II of the Fuels Quality Directive.

and biodiesel blends (but labeled as biofuels not simply petrol or diesel), like for example petrol containing from 70% to 85% bioethanol, B20, or B100 in Polish Regulation of Minister of Energy from 25 May 2016.[7]

4.2 The US framework

4.2.1 Sources of law and motivation

The main acts dealing with biofuels are the Energy Policy Act from 2005[8] and the Energy Independence and Security Act from 2007,[9] with later amendments. Both acts have established the Renewable Fuel Standard (see in the official compilation of US law, that is title 42, § 7545, letter (o) of US Code)[10] as the crucial biofuels support scheme in the United States. More detailed executive acts are passed by the Environmental Protection Agency (EPA) and published in the Federal Register.[11]

The Energy Independence and Security Act has a very short preamble indicating the aims of the law. Main targets are energy independence and security, increase production of clean renewable fuels, protect consumers, increase energy efficiency, promote research on and deploy greenhouse gas capture and storage options, and improve energy performance of the Federal Government.

4.2.2 Support schemes

The main support mechanism, named The Renewable Fuel Standard (RFS), established the following goal: the total amount of US fuel should contain a certain share of renewable fuel. The share is rising year by year from 4 billion of gallons in 2006 to 36 billion of gallons in 2022, that equals nearly 20% of projected motor fuel consumption in 2022 (Stokes and Breetz, 2018, p. 82).

[7] Online System of Legal Acts, http://prawo.sejm.gov.pl/isap.nsf/DocDetails.xsp?id=WDU20160000771 (accessed 1 October 2018).

[8] http://uscode.house.gov/statviewer.htm?volume=119&page=594 (accessed 19 September 2018).

[9] http://uscode.house.gov/statviewer.htm?volume=121&page=1492 (accessed 19 September 2018).

[10] § 7545 of US Code, http://uscode.house.gov/view.xhtml?req=(title:42%20section:7545%20edition:prelim)%20OR%20(granuleid:USC-prelim-title42-section7545)&f=treesort&edition=prelim&num=0&jumpTo=true (accessed 17 September 2018).

[11] § 7545, letter (o), paragraph (3) of U.S. Code as well as Environmental Protection Agency official website, https://www.epa.gov/renewable-fuel-standard-program/regulations-and-volume-standards-renewable-fuel-standards (accessed 20 September 2018).

The general renewable fuel goal includes three additional targets, that is minimal amount of advanced biofuel (from 0.6 in 2009 to 21 billion of gallons in 2022), cellulosic biofuel (from 0.1 in 2010 to 16 billion of gallons in 2022), and biomass-based diesel (from 0.5 in 2009 to 1 billion of gallons in 2012).

General provisions of statutes are detailed in executive regulations of EPA, which provide every year (on November the 30th) obligatory percentage quotas of biofuels to be reached by refineries, blenders, and importers in the following year.

The RFS, unlike as RED Directive, established credit program. Based on this program, actors exceeding the minimum required amount of biofuels obtain a proportional amount of credits. Such credits can be traded with other actors unable to reach their minimal biofuels level. Such mechanism aims at reaching biofuels goals in the most economically efficient way, showing similar features to the UE cap-and-trade system (Thompson et al., 2018).

The RFS allows waiving the biofuels targets in whole or in part if "implementation of the requirement would severely harm the economy or environment of a State, a region, or the United States" or "there is an inadequate domestic supply." There are also additional waivers opportunities in the case of cellulosic biofuel (if the production is too small) and biomass-based diesel (in the case of "significant renewable feedstock disruption or other market circumstances").

4.2.3 Standards of biofuels

Standards of renewable fuels were based on the definition of renewable biomass. First, such biomass should take form of planted crops or trees, and their residue harvested from land cleared or cultivated before 19 December 2007 (and in the case of crops—nonforested). Such regulation seems to counteract deforestation activities. Second, it should not come from forests or forestlands that are "ecological communities with a global or State ranking of critically imperiled, imperiled, or rare pursuant to a State Natural Heritage Program, old growth forest, or late successional forest."[12]

Third, renewable biomass also can take form of animal waste material, animal by-products, biomass obtained from the immediate vicinity of buildings and other areas regularly occupied by people, or of public infrastructure, at risk from wildfire, separated yard waste or food waste, including recycled cooking and trap grease, and algae.

[12] § 7545, letter (o), letter (I), point (iv) of US Code.

Fourth, specific targets regarding fuels are limited to GHG emissions reduction criteria. For instance, advanced fuels are defined as renewable fuels (excluded ethanol derived from corn starch) when their lifecycle GHG emissions achieve at least 50% reduction compared to the baseline lifecycle emissions. The cellulosic biofuels are defined as "renewable fuel derived from any cellulose, hemicellulose, or lignin that is derived from renewable biomass" when their lifecycle GHG emissions achieve at least 60% reduction compared to the baseline lifecycle emissions. The biomass-based diesel is defined as biodiesel with at least 50% less lifecycle GHG emissions than the baseline lifecycle emissions.

A fifth criterion stems from the regulation providing that: "in the case of any such renewable fuel produced from new facilities that commence construction after December 19, 2007, achieves at least a 20 percent reduction in lifecycle greenhouse gas emissions compared to baseline lifecycle greenhouse gas emissions."

The amount of ethanol in traditional fuel should not exceed 10% but could be up to 15% when used in model year 2001 and newer light-duty motor vehicles (that means petrol, e.g., E10 and E15).[13] According to ASTM D975 standard, conventional diesel fuels should contain up to 5%[14] of biodiesel to be named simply "diesel"; however, there is possibility to register fuels with any amount of biodiesel up to 100%.[15]

[13] Regulation to Mitigate the Misfueling of Vehicles and Engines with Gasoline Containing Greater Than 10 Volume Percent Ethanol and Modifications to the Reformulated and Conventional Gasoline Programs, Environmental Protection Agency, https://www.gpo.gov/fdsys/pkg/FR-2011-07-25/pdf/2011-16459.pdf (accessed 27 IX 2018).

[14] US Department of Energy, https://www.afdc.energy.gov/fuels/biodiesel_blends.html (accessed 1 October 2018).

[15] Environmental Protection Agency, https://nepis.epa.gov/Exe/ZyNET.exe/P1001SRG.txt?ZyActionD=ZyDocument&Client=EPA&Index=2006%20Thru%202010&Docs=&Query=&Time=&EndTime=&SearchMethod=1&TocRestrict=n&Toc=&TocEntry=&QField=&QFieldYear=&QFieldMonth=&QFieldDay=&UseQField=&IntQFieldOp=0&ExtQFieldOp=0&XmlQuery=&File=D%3A%5CZYFILES%5CINDEX%20DATA%5C06THRU10%5CTXT%5C00000005%5CP1001SRG.txt&User=ANONYMOUS&Password=anonymous&SortMethod=h%7C-&MaximumDocuments=1&FuzzyDegree=0&ImageQuality=r75g8/r75g8/x150y150g16/i425&Display=hpfr&DefSeekPage=x&SearchBack=ZyActionL&Back=ZyActionS&BackDesc=Results%20page&MaximumPages=1&ZyEntry=5 (accessed 1 October 2018).

4.3 The China, India, and Brazil frameworks

4.3.1 China

Important reasons for promoting biofuels in China include the rapid growth of number of vehicles and air quality connected issues, energy security, and CO2 emissions problems (Hao et al., 2018, p. 645). China did not establish binding national targets (but there are official projections of 10 Mt. of ethanol and 2 Mt. of biodiesel consumption in 2020) for biofuels consumption or national mandatory quotas for producers, distributors, and sellers. However, it introduced other support schemes, especially demonstration projects, tax instruments, subsidies, and standards (Hao et al., 2018). Moreover, E10 blending zones were established in 10 provinces and two municipalities in 2016 (Beckman et al., 2018, p. 259).

A demonstration program started in 2002 involving two bioethanol producers that provided bioethanol to two cities, and which was expanded later. It included warranted price for bioethanol petrol equal to the price of pure petrol with the same grade and specified trade price between bioethanol producers and petroleum companies. Food security issues lead to hold back projects based on crop-based fuel, promoting the development of noncrop-based biofuels projects. Biodiesel projects were also developed, but in limited number of regions. This might be associated with a low utilization of biodiesel production capacity, which is in turn influenced by lack of demand and problems with feedstock. The big companies have not incentives for production of biodiesel because there is a lack of mandatory regulations and the regional regulations are not sufficiently implemented. At the same time, companies are afraid of decreasing benefits from selling conventional fuels. Moreover, waste oil, which is the main biodiesel feedstock, is mainly used as cooking oil. Such use is strictly prohibited by law, but the financial benefits of using waste oil as cooking oil than sell to fuels companies are higher (Hao et al., 2018, pp. 649–651).

Alike demonstration programs, also tax incentives and subsidy policies, originally introduced for all types of bioethanol production, were withdrawn for bioethanol produced from crop-based fuels and kept only for noncrop-based biofuels. Tax exemption mechanisms and VAT reimbursement were initially introduced for bioethanol production, such incentives were gradually withdrawn in 2015. Similarly, previously introduced subsidies were phased out in 2015, but not for noncrop-based biofuels. Tax exemption was also kept for waste oil-based biodiesel.

As far as standards are concerned, China established four standards since 2001 that are the Denatured Fuel Ethanol (GB 18350), Ethanol Gasoline for

Motor Vehicles (E10) (GB 18351), Biodiesel Blend Stock (BD100) for Diesel Engine Fuels (GBT 20828), and Biodiesel Fuel Blend (B5) (GBT 25199). These standards are frequently updated over time (Hao et al., 2018, p. 649).

4.3.2 India

The drivers for Indian biofuels policy include, among others, energy security, employment for rural population, reduction of poverty, rehabilitation of "waste lands," and tackling CO_2 emissions. At the same time, due to the limited area for energy crops, growing population and greatest number of undernourished people, and the threat to food security, the Government of India has promoted cultivations of nonedible energy crops (e.g., *Jatropha curcas*) on so-called wastelands, that is degraded and underutilized land (Chaliganti and Müller, 2016; Murali et al., 2016).

The National Biofuel Policy established the indicative target of 20% blending for biofuels (both bioethanol and biodiesel; Kataki et al., 2017). In 2013 after introducing mandatory blending quotas for bioethanol in a limited number of states and territories, the Government of India mandated a 5% quota of blending for ethanol in the whole country. Two additional mechanisms named "Minimum Support Price of non-edible oil seeds" and "Minimum Purchase Price for bioethanol and biodiesel" were also adopted (Murali et al., 2016, p. 449). However, as argued by Beckman et al. (2018, p. 260) "actual blending has never reached the targeted rate because of inadequate domestic supplies, inadequate price incentives for ethanol producers and blenders, and a requirement that fuel ethanol, as opposed to ethanol destined for industrial or chemical use, be supplied from domestic sources."

4.3.3 Brazil

Brazilian biofuels support scheme goes back to interwar period when a mandatory blending level of bioethanol and petrol, as well as financial instruments was introduced. Challenging the excess of sugar stock and economic depression, an obligatory level of bioethanol blending for petrol importers was introduced at 5% level by federal decree in 1931. Moreover, tax exemption for blended ethanol, duty exemption for anhydrous ethanol production and distillation machineries, and higher taxes on vehicles with low-compression internal combustion engines were also introduced by the same decree. In the same year, the "Study Commission on the Ethanol Engine" and the "Commission for Defense of Sugar Production" were created and in 1933 were merged into the "Sugar and Ethanol Institute" aimed to promote production and use of ethanol as a fuel. For security reasons in 1938 blending obligations were imposed upon domestic producers, and guaranteed prices for ethanol and sugar were introduced (Rutherford, 2016).

In order to face the 1973 oil crisis, a National Alcohol Programme (PROALCOOL) introducing low-rate credits from public banks and prices parity between ethanol and sugar was launched in 1975. Moreover, automotive policy aimed at promoting the development of pure bioethanol motors was introduced, and cassava (mandioca) and other fuel-agriculture products were incentivized. After the second oil crisis PROALCOOL was further developed through the introduction of loans in line with the inflation rate, as well as tax reductions for bioethanol cars and bioethanol itself.

Most supporting interventions in the bioethanol sector were progressively reduced over the period 1989 and 1999.[16] Despite today bioethanol in Brazil being largely driven by market forces (Rutherford, 2016, p. 216), new supporting instruments (mainly financing innovations in the sector and bioethanol storage) were introduced to face the bioethanol supply crisis in 2011. Mandatory blending level was also maintained in the range between 18% and 27.5%—this in order to match bioethanol prices with petrol price.

When it comes to biodiesel, despite the first supporting programs were launched in 1980s, policy support accelerated starting from 2002. Since November 2014 the blending level of biodiesel has reached 7% and tax exemptions and other incentives are also available. The biodiesel legislation is driven not only by economic and security reasons but also by social inclusion and environmental (e.g., climate policies) concerns. To this aim, the Social Fuel Seal (SFS), which promotes the inclusion of small-scale agriculture in diesel production, was created. To obtain the SFS, biodiesel producers must purchase feedstock from family-based farmers included in the National Program for the Strengthening of Family Farming. SFS holders benefit from tax exemption, better conditions of financing, preferential allocation in auctions, and exclusive supply for biodiesel stocks (Rutherford, 2016).

5 Lessons learned and future perspectives for the bio-based economy

The analysis developed so far shows that biofuels development is associated with several problems, including competition with food production,

[16] However, in the face of dropped oil prices and increased prices for sugar, the mandatory blending level was established at 22% in 1993.

indirect land use change, and loss of biodiversity. These problems are partially addressed through regulations, standards, and policy schemes. At the same time, biofuels and other bioenergy policies, together with policies regarding other bioeconomy sectors (e.g., waste, packaging, forest policies), are directly influencing the industry, using measurable goals and effective instruments for the development of bioeconomy in the EU (STAR-ProBio, 2018, pp. 21–26).

The competition of food vs. fuel is directly addressed in many regulations, including 7% limit of biofuels produced with crops from agriculture land, for example, cereals, oil crops, and so on, in the EU; targets for lignocellulosic biomass in the United States; and support for nonfood crops for biomass in China and India. However, the development of second-generation biofuels is very problematic and many regulations, especially in United States, can be treated as overoptimistic. For example, despite China started to support nonfood crops for biofuels, corn and wheat accounted for 80% of feedstock used in the production of bioethanol in 2015. Moreover, although bioethanol in India is produced almost exclusively from molasses (coproduct of cane sugar production), the achievement of 5% to 10% of blending seems unrealistic because of the strict connection with sugar production and prices. Biodiesel production from waste cooking oil in China and from jatropha in India has not been successful yet (Beckman et al., 2018, pp. 259–260). In turn, in the United States the commercialization of cellulosic ethanol has been slow, and the 5% limit for cereals and other related biomass (today 7%) in the European Union was considered to make it difficult to achieve the target of 10% renewable fuels in transport (HLPE, 2013, p. 40; Stokes and Breetz, 2018, p. 82).

For these reasons, technological progress on its own is not enough and it should be combined with additional political and economic instruments and experts view for a transition to a sustainable bioeconomy. In particular, The High Level Panel of Experts on Food Security and Nutrition (HLPE) of the United Nations Committee on World Food Security (CFS) stated that there is a need to foster the transition from pure biofuels policies to comprehensive food-energy policies. The latter are very important since they can be "an effective development strategy to provide high-value products, electricity and alternative power for cooking, power for water management and local productive facilities, in addition to transport fuel" (HLPE, 2013, p. 18). Moreover, transport and climate policies should consider other measures despite biofuels, including increasing fuel efficiency, developing collective transport, and alternative renewable fuels.

Comprehensive food-energy policies can help achieving the targets of renewable energy for transport (including solar, wind, and biofuels) and, at the same time, relieve the pressure from food production. Based on HLPE recommendations, on the one hand, there is the need to support research and development (R&D) for advanced biofuels, and to identify ways in which biofuels could contribute to the restoration of degraded land and better management of watersheds. Simultaneously, "research partners should devise solutions adapted to the needs of the least developed countries and of smallholders who are most in need of access to energy" (HLPE, 2013, p. 18).

On the other hand, regardless of the success in the development of advanced biofuels, food security should be prioritized in any biofuel policy. Since biofuels sector is more and more globalized and market driven, countries should create international cooperation mechanisms, including regular notification of biofuels policies by country, to protect food security threatened by biofuels production (HLPE, 2013, p. 13).

Therefore policies should integrate land and water assessment of biofuels development, before and after the concessions of land. This assessment should include in the same measure also nonfood crops because there are not "magic non-food crops that can ensure more harmonious biofuel production on marginal lands. Therefore, non-food/feed crops should be assessed with the same rigour as food/feed crops for their direct and indirect food security impacts" (HLPE, 2013, p. 18). For example, "it has become clear, however, that, while jatropha might have some of the agronomic advantages initially identified, its economic viability demands high productivity levels, which in turn require better varieties, better quality soils and greater water inputs. It provides no ready solution, therefore, to the competition for resources that has been the main source of criticism of first-generation biofuels" (HLPE, 2013, p. 46).

Moreover, World Health Organization and other relevant stakeholders should develop appropriate methodologies for assessing international and national biofuels policies, while Global Bioenergy Partnership (GBEP) should ensure only the use of certification schemes which are "multi-stakeholder, fully participative and transparent" (HLPE, 2013, p. 19). The transactions cost of such schemes should be limited to include also small stakeholders.

A holistic perspective, not limited to biofuels, could also help avoiding the threat of maintaining unsustainable status quo that is stopped at the stage of weak ecological modernization. Such threat could lead, for example, to a dominant position of individual car transport fuelled with biofuels on the detriment of collective transport or cycling. At the same time, the biomass which was not used for producing biofuels could be used for other bioeconomy sectors, including packaging or most crucial food production, following the cascading approach adopted in integrated biorefineries.

Although food problems, as well as problems related to indirect land use changes, GHGs emissions, and biodiversity loss are directly addressed through regulations, standards, and policies, for example, in the form of obligatory emissions reductions and bans on production of biomass in certain natural land in the EU and the United States, the development of holistic food-energy or bioeconomy policies seems to be still insufficient. In the remainder of this section we will broaden the concepts of comprehensive bioeconomy, climate, and sustainability policies, focusing on their relations to economic growth and global challenges.

The analysis developed so far seems to confirm the suggestion of Malik et al. (2016) concerning the insufficient role of technological improvements to combat CO_2 emissions. As a consequence, policy actions should focus also on the demand side. This is indeed the case for biofuels, in particular, and bioeconomy policy, in general. One way to address environmental problems could be the adoption of "sufficiency" principle in resource consumption and thus turning to economic models that go beyond GDP growth, such as degrowth or steady-state economy (O'Neill et al., 2018, p. 92).

However, as observed by Raftery et al. (2017, p. 3) "policies to reduce GDP per capita seem unlikely." In addition, Malik et al. (2016, p. G) stated that "it is not only difficult but also impossible to implement policies for interfering in people's freedom of choice, and restraining their consumption, particularly in liberal-democratic societies of most developed nations." Nevertheless, justifying inappropriate or insufficient policies and regulations with the concept of "freedom of choice" could lead to over-blame consumers for environmental degradation, releasing at the same time policy-makers and other important stakeholders from their responsibilities in maintaining the unsustainable status quo (Walker, 2015, pp. 55–56; Shove et al., 2012, p. 164; Evans, 2011). This clash of responsibilities increases the risk for behaviors that undermine human well-being, while favoring business models that promote environmentally and socially unsustainable activities.

Therefore we argue that scientists and policy-makers should promote deeper—rather than purely technological—societal changes. For instance, in the case of biofuels, along with the development of sustainability standards, action should be undertaken to reduce unsustainable transport behaviors, especially car driving and air travels, and to support a shift to modal transportation for passengers. The number of cars is expected to increase to 2.9 billion by 2050, most of which in emerging economies, especially China and India (Doucette and McCulloch, 2011, p. 803). The future of aviation is also posing some serious threats. According to ICAO projections (2016), despite technological and operational improvements, the CO2 emissions from aviation through 2050 will increase. Therefore "additional measures will be needed to achieve carbon neutral growth relative to 2020" (ICAO, 2016, p. 22).

As it seems, the positive environmental effects of biofuels, and bioeconomy in general, might be offset by mounting social issues and intensified unsustainable consumption patterns. To avoid this, a twofold approach to the bioeconomy is needed. On the one hand, technological developments, including second and third generation biofuels, are extremely relevant to complete the transition. On the other hand, sustainable transport behaviors (e.g., public transport, less air travels) and conscious consumption patterns, in general, can generate positive environmental effects causing less pressure on food production, and land with special environmental value or high carbon stock. Therefore the few examples of effective policy intervention or voluntary engagement already implemented, for example, the EU ban on single-use plastics[17] or the reduction of packaging materials, should be diffused as good practices for all the sectors of bioeconomy.

6 Conclusions

This chapter has addressed the biofuel transition, analyzing first the main economic, social, and environmental issues associated with biofuel production and consumption, and subsequently assessing the relevance and effectiveness of standards and regulations in addressing those issues. Building on this, in Section 5 we extend our discourse to the broader context of bioeconomy transition, drawing lessons and insights from the biofuel experience.

Key problems associated with the biofuel transitions include indirect land use change (iLUC), food crop prices and associated food security issues, as

[17] Initially accepted by the European Parliament but not yet adopted as law—European Parliament Legislative Observatory, https://oeil.secure.europarl.europa.eu/oeil/popups/ficheprocedure.do?lang=en&reference=2018/0172(OLP) (accessed 29 October 2018r.).

well as equality and gender issues stemming from lack of access to resources deriving from increasing land pressure.

As observed, second and third generation biofuels, which use feedstock produced in marginal lands or waste biomass, address some of these issues in a coherent way, providing technologically driven solutions. However, a holistic approach able to take into consideration production and consumption elements is advocated as vital in promoting a long-term sustainable transition. In this regard, regulations and standards can play a key role, starting from the definition of supporting schemes and sustainability criteria which take into consideration the whole lifecycle of the product as well as the three pillars of sustainability (social, economic, and environmental). Going beyond, regulation can promote the emergence of responsible production and consumption behavior in many ways including, for instance, investments in R&D, public and private partnerships, information and education campaign, as well as new consumption models associated with circular and sharing practices.

In this sense the adoption of an overarching strategy for the bioeconomy by the European Union (as well by several member states) is welcomed as it sets a wide-ranging roadmap, including adequate and proportionate measures, to address the main challenges associated with the ongoing sustainability transition out of a fossil-based economy.

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CHAPTER 3

Triple bottom line, sustainability and sustainability assessment, an overview

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1 Introduction

Over the years, the concept and practice of sustainable development has been continuously evolving. Indeed, the greatest challenge that humanity faces today is to plan and carry out human activities in a manner compatible with the Earth's limits.

Despite the fact that the concept of sustainable development has been developed and studied for several decades, its practical application is still limited and many targets have not been met. This means humanity is still operating unsustainably, and there are clear and ample implications of continued unsustainable production and consumption.

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Environmental policies have been evolving recognizing the leading role of sustainable production and consumption (SCP) to ensure an absolute decoupling of environmental impacts from socioeconomic well-being. A significant decoupling of environmental impacts from socioeconomic well-being requires the definition of specific policies aiming at reducing burdens associated to production and consumption of goods and services.

A new paradigm for economic growth, social equality, and environmental sustainability was set in 1987 when the "Brundtland's report" (WCED, 1987) introduced the concept of "sustainable development" to the international community. Sustainable development is the human development that meets the needs of the present generations without compromising the ability of future ones to meet their own needs. The definition builds on: (i) the concepts of "needs," in particular the essential needs of human population and (ii) the idea of limitations, which is imposed by environment's ability to meet both present and future needs, as well as by the level of technological advancement and social organization.

The transition toward sustainable production and consumption is recognized as one of the major challenges for sustainability and specific methodologies are needed in order to analyze the current situation; to define future scenarios; and to assess the capability of policies, plans, and actions to provide adequate solutions. Increasing demand for energy, food, water, and other resources has resulted in resource depletion, pollution, environmental degradation, and climate change, pushing the earth toward its environmental limits. With humans now consuming more resources than ever before, the current patterns of development across the world are not sustainable.

SCP is about fulfilling the needs of all while using fewer resources, including energy and water, and producing less waste and pollution. It can contribute to poverty alleviation and the transition toward a low carbon, green economy. SCP is as well essential for improving the lives of the world's poorest people, who depend so closely on the natural resources provided by their environment (UN, 2015).

The United Nations Secretary-General's High-level Panel on Global Sustainability in the report "Resilient People, Resilient Planet: A future worth choosing" (UN, 2012) has defined a vision for a sustainable planet. This entails a just society and a growing economy which aim at: (i) eradicating poverty, reducing inequality, and making growth inclusive; (ii) making production and consumption more sustainable, while combating climate change and respecting a range of other planetary boundaries;

(iii) enabling consumers to make sustainable choices and to advance responsible behavior individually and collectively; (iv) managing resources and enabling a 21st century green revolution in the fields of agriculture, oceans and coastal systems, energy and technology.

These objectives are nowadays reflected in the sustainable development goals (SDGs) (UN, 2015), in which an ethical connotation is at the basis of the goal 12, on responsible consumption and production.

Moreover, the EU sustainable development strategy (CEC, 2001, 2008, 2009) depicted the EU vision on sustainable development, highlighting key topic to be mainstreamed within the EU policy context, with a clear focus on sustainable production and consumption. Moreover, the SDGs are taken as reference for the EU sustainability agenda (CEC, 2016).

At European level, the new Bioeconomy strategy is advocating a transition to a sustainable bioeconomy, clearly stated from the title of the main Communication "A sustainable Bioeconomy for Europe: Strengthening the connection between economy, society and the environment" (CEC, 2018).

Natural resources provided by the Earth, both biotic and abiotic (i.e., raw materials, energy, water, air, land, and soil as well as biodiversity and ecosystems) represent crucial economy and life-support elements for human societies worldwide. Indeed, natural resources are a building block in the supply chain, thus pushing the economic growth, and providing global functions, as in climate regulation. In a globalized world where population is in continuous expansion and the demand for finite resources continues to rocket, the current production and consumption patterns in both developed and developing countries are generating great concerns. Particular concerns are related to the potential repercussion on the environment and climate. On such a background, a transition toward bio-based economy represents an opportunity to comprehensively address interconnected societal challenges such as food security, natural resource scarcity, fossil resource dependence, and climate change, while achieving sustainable economic growth (CEC, 2012; Ronzon et al., 2017). However, not only fossil-based products carry an environmental burden, but also bio-based ones. Furthermore, the use of bio-based resources may raise issues such as those on land competition for food production. Hence, to be effective, bioeconomy products strategies should be founded on resource efficiency and eco-innovation principles, as well as should be interconnected with circular economy to ensure the least dissipative use of resources (Corrado and Sala, 2018).

2 State of the art in sustainability assessment

Over the years, the world has gained a deeper understanding of the interconnected challenges we face and has recognized that sustainable development has to embrace several sustainability pillars: from the three fundamental pillars of the environmental protection, economic growth, and social equity to pillars concerning, for example, institutional (O'Connor, 2006), cultural (Nurse, 2006), and technological (Vos, 2007) aspects. The concept of the environmental sustainability is central in the sustainability discourse and it is rooted in the ecology domain, in relation to the carrying capacity of the earth system.

However, the definition of sustainability and "what should be sustained" (e.g., what might constitute critical natural capital) is by no means agreed within the scientific community. The definition is subject to value judgments (Bell and Morse, 2008; Bond et al., 2011), and it could be interpreted as a shared ethical belief (Seager et al., 2004). Patterson et al. (2017) identified four main interpretations of the concept of sustainability: (i) ecological, (ii) economic, (iii) thermodynamic and ecological-economic, (iv) public policy and planning theory. The ecological interpretation focuses on a vision of the socioeconomic system embedded in the global biophysical system; the economic emphasizes the idea of social welfare; the thermodynamic interpretation poses ecological sustainability in the context of the entropic nature of economic-environmental interactions; the public policy and planning interpretation seeks to achieve a balance of the different aforementioned factors.

Giving the theoretical underpinning complexity of the sustainability concept, sustainability assessment (SA) is one of the most complex types of appraisal. Not only SA does entail multidisciplinary aspects (environmental, economic, and social), but also cultural and value-based dimensions. Besides, SA is usually conducted for supporting decision-making and policy development in a broad context. Indeed, assessing sustainability is increasingly becoming common practice in product, policy, and institutional appraisals. Concepts such as "Integrated Assessment" and "Sustainability Assessment"[1] are introduced to offer "new" perspectives to impact assessment geared toward planning and decision-making on sustainable development (SD) (Hacking and Guthrie, 2008). However, sustainability assessment

[1] Other synonyms adopted are "triple bottom line assessment," "3E impact assessment" (environmental, economic, equity), "extended impact assessment," and "sustainability appraisal."

is a methodology "that can help decision-makers and policy-makers decide what actions they should take and should not take in an attempt to make society more sustainable" (Devuyst, 2001, p. 9). In this context, the so-called triple bottom line is an accounting framework with three parts: social, environmental (or ecological), and economic.

However, as mentioned before, other dimension of sustainability may be considered (such as institutional and cultural) and different approaches to sustainability assessment have been developed over time.

One of the main consequences of having multiple perspectives in sustainability affects the definition and the assessment of the different capitals (natural, social, and economic). This implies two basic approaches to sustainability: strong and weak. Strong sustainability is based on the condition that some natural capital provides functions that are not substitutable by man-made capital: each capital needs to be preserved for future generation. Weak sustainability reflects a view whereby natural and man-made capital together comprise total capital; natural capital is considered to be substitutable for man-made capital and weak sustainability occurs whereby the level of total capital passed onto future generations does not decrease (the inference being that man-made capital has replaced natural capital to maintain total capital) (Pearce et al., 1994).

Besides, in recent years also the categorization of capitals has been extended. For example, Porritt (2007) has developed a five capitals framework (natural, human, social, manufactured, and financial) in which the capitals are not purely of instrumental value but they represent an appropriate framework within which particular endpoints of intrinsic value can be identified.

Furthermore, there is another level of complexity when addressing different capitals and associated values: some of them may be globally recognized (such as the thermodynamics underpinning chemical and physical process), other with very specific, local/regional values and meanings (such as the concept of well-being, if developed and developing countries, with different context and culture, are compared) (Sala et al., 2013a). This leads not only to ontological but also methodological challenges in capital's evaluation.

The debate over sustainable development has led to defining a new discipline: Sustainability Science (SS). SS is considered an emerging discipline, applicative and solution-oriented, whose aim is to handle environmental, social, and economic issues in light of cultural, historic, and institutional perspectives. The challenges of the discipline are not only related to better identifying the problems affecting sustainability but to the actual transition toward solutions adopting an integrated, comprehensive, and participatory approach. This implies the coexistence of a scientific and a social paradigm as basis to define any kind of assessment of sustainability as well as to design of solutions and interventions (Sala et al., 2013a,b).

In order to offer guidance, principles for sustainability assessment have been proposed over time. For example, the Bellagio STAMP (Sustainability Assessment and Measurement Principles), first developed in 1996 and then revisited in 2012 (Pinter et al., 2012) represents an attempt of delineating the principles and requirements of robust Sustainability Assessment. Nowadays, most case studies assessing sustainability and adopting the common triple bottom-line approach (TBL) still end up comparing different alternatives on the basis of indicators (more or less) randomly chosen from among various alternatives in the three pillars of sustainability (namely, economy, environment, and society), without deepening the analysis of potential interconnections between the pillars. Namely, selecting the indicators depending more on information availability, rather than by the necessity to represent one of the three pillars.

In the literature, a broad range of different appraisal processes is described under the heading of sustainability assessment (SA). Nevertheless, current SA practices need a robust framework to overcome concerns recognized in the scientific community regarding whether the various available examples of assessment are really comprehensive and robust, moving from integrated assessment toward an SA. Increasing comprehensiveness and robustness of assessment may fulfill the "transformational" role request to sustainability science. Hence, SA could be seen as leverage for effectively promoting sustainability and not only for evaluating its progress and/or comparing options (Sala et al., 2015a).

Fig. 3.1 illustrate the key life cycle stages of biofuels production and consumption, which requires sustainability assessment.

Biofuels sustainability pertains to the so-called wicked problems for which a sustainability assessment is particularly challenging. Currently, decision-making is facing multiple energy, development, and climate objectives (Bhardwaj et al., 2019). Biofuels assessment is multifaceted. Specific bioenergy options (such as biofuels produced from edible vs. nonedible feedstocks) are not positive or negative per se. In fact, sustainability impacts are context specific, both in terms of the location and management of feedstock production systems and the socioeconomic systems where their

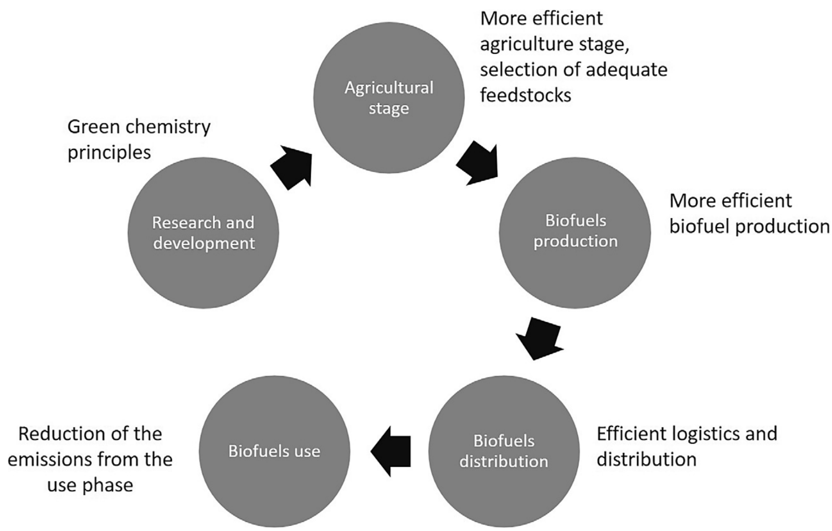

Fig. 3.1 Typical life cycle stages of biofuels design, production, and use to be evaluated within sustainability assessment and complemented with key concept for their improvement.

production occurs. A wide spectrum of knowledge and competences are needed to govern bioenergy expansion to harness opportunities and minimize risks of negative impacts.

Assessment of biofuels from environmental, economic, and political point of view has been growing over time (Demirbas, 2009) highlighting the multidimensional nature of the problems and the existence of multiple, often conflicting, objectives (Lovett et al., 2011). Indeed, biofuels are promoted as replacements for transport fuels, but biofuel policies are also asked to fulfill socioeconomic goals and strategic goals such as security of energy supply. Notwithstanding biofuel has been evolved from first to fourth generation along with the assessment of different feedstock and production technologies, sustainability concerns still exist. Liew et al. (2014) reviewed the state of the art of technologies and assessment methods on economic performance, safety, health, and environment (SHE) as well as social impact for biofuel production, starting from the early process design phase of biofuel production.

The following section illustrates key aspects of the sustainability assessment, which are essential for a robust assessment, especially considering the bioeconomy context.

3 Biofuels and sustainable development goals

In order to perform a comprehensive sustainability assessment, a theoretical reference framework is needed. This framework may then enable the selection of appropriate methods and tools for assessing benefits or burdens of human intervention.

The sustainable development goals (UN, 2015) are crucial to define the objective and the key challenges in biofuel assessment.

Biofuels are considered as one of the pivotal solutions for sustainability, as they may contribute to global climate change mitigation as well as other environmental and social objectives. Bioenergy typically reduces reliance on fossil fuels and may enhance regional energy access. This has clear implications to the forestry and agriculture sectors, including the potential increase use of renewable resources as feedstocks for a range of industrial processes. However, trade-offs related to bioenergy and biofuels exist if they are not produced properly. Among key concerns, there are food security, land use competition, direct and indirect impacts due to land use and land use change, biodiversity decline, challenges in economic competitiveness, and limitation in high quality and affordable energy services.

To given an overview of the complexity of the SA of biofuels, Fig. 3.2 illustrates how the different SDGs are interlinked with biofuels, from their design up to production and use.

To support the international dimension, since 2005, the United Nation Conference on Trade and Development (UNCTAD) has launched the "Biofuels Initiative" (UNCTAD, 2018). The initiative is focusing mainly on Sustainable Development Goal 7 (Affordable and Clean Energy) and Goal 9 (Industry, Innovation and Infrastructure) and collaborate with other international organizations, NGOs, and academia, aiming at supporting countries for what concerns bioenergy and biorefining.

In literature, several studies have addressed the role of biofuels toward SDGs, considering that biofuels might be relevant in reaching these goals, as one of the most advanced alternative energy sources. Acheampong et al. (2017) reviewed the literature, assessing the potential of biofuels to contribute to the SDGs by presenting an appraisal of their development over the years. They concluded that, notwithstanding the existence of potential negative trade-off, a combination of plant biology, carbon capture techniques, and novel bioconversion processes for third and fourth generation biofuel might reach the goal of providing fuels that will be abundant, energy efficient, and environmentally sustainable.

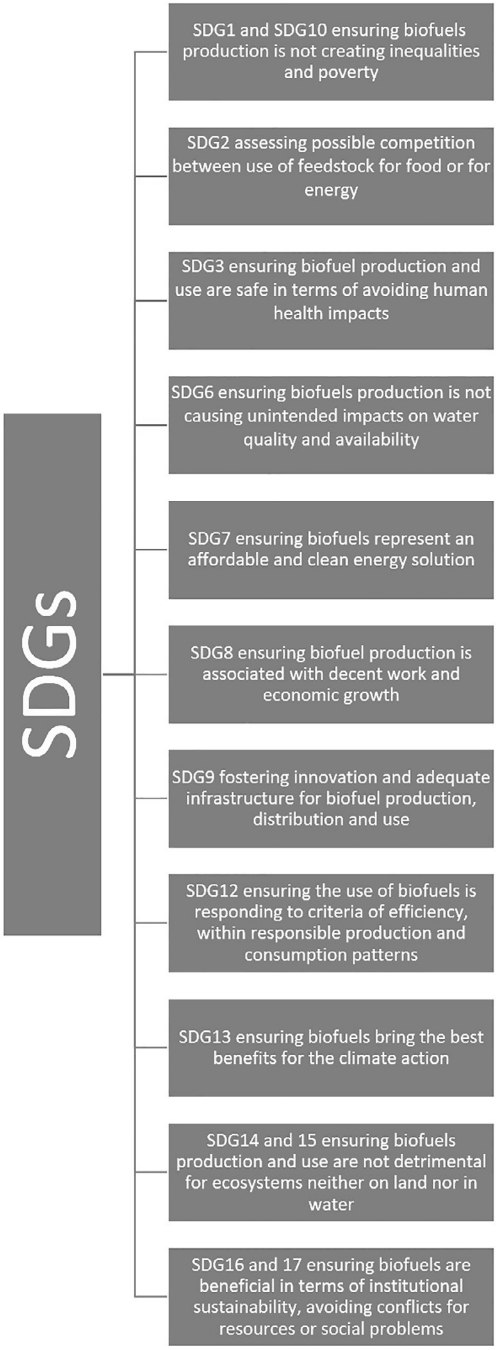

Fig. 3.2 Relationship between sustainable development goals and biofuels production and consumption.

Another recent review by McCollum et al. (2018), unveiled that there are knowledge gaps regarding the interactions between the energy SDG targets and those of the non energy-focused SDGs, including their context dependencies (relating to time, geography, governance, technology, and directionality). This requires further efforts to promote policy coherence and integrated assessments to assess potential policy spillovers across sectors, different sustainability domains, and issues associated to geographic and temporal boundaries. Indeed, the debate on the importance of interlinkages between SDGs and the interactions due to geographical context, resource endowments, time horizon, and governance is open (Nilsson et al., 2018).

4 Environmental sustainability

The economic, social, and economic pillars of the sustainability should be assessed in a quantitative or semiquantitative manner. This requires the use of specific methods and models to allow the comparison of alternative solutions and the appraisal of the absolute impacts and performance associated to a studied system.

Sala et al. (2015a) compared several approaches in order to highlight those that may be considered more suitable for conducting sustainability assessment. From this assessment, it emerged that life cycle thinking and life cycle assessment (LCA) are vital elements of sustainability assessment and increasingly mentioned as being essential for informing decisions in a comprehensive and holistic manner, in both business and policy contexts (Sala et al., 2013a,b).

While LCA focuses primarily on burdens linked to emissions into the environment and resources, life cycle costing (LCC) aims at assessing cost along the supply chain and the emerging social life cycle assessment (SLCA) complements this in relation to working hours/conditions and social domains to complete the environment and socioeconomic analysis. Aiming to cover the different pillars of sustainability, life cycle sustainability assessment (LCSA) methodologies and applications are under development aiming at integrating better the sustainability pillars, while assessing the mutual interaction among them. From the literature and the LCA practice, it is clear that LCA is a methodology, which may complement other methodologies and insights, for assessing the performance of goods/ services/ systems/ technologies/ innovations/ infrastructures/ waste management options/ regions. While the application of LCA in the context of business has a longer tradition (starting in the 1970s), the array of options for the use of LCA in other decision context, including in policy making, is not yet completely deployed.

The European Commission has released a Communication on Better regulation (CEC, 2015a) in order to improve the policy making process. The Communication is complemented with a Better Regulation toolbox (CEC, 2015b) which lists models and methods to be used for assessing impacts and benefits of policies, in the so-called policy impact assessment step. Within the toolbox, life cycle assessment is listed among the models, which may support the environmental assessment of impact and benefits associated to different policy options (Sala et al., 2016).

4.1 Life cycle assessment

Life cycle assessment is a standardized methodology (ISO, 2006) for assessing potential environmental impacts associated to a product, a process, or a system, along its life cycle, namely, from the extraction of raw material to the end of life. The main steps of LCA are reported in Fig. 3.3. The goal and scope of the study define the system to be assessed. Based on the goal and scope, the inventory of all resource and emissions—happening during the different life cycle stage—is collected in the life cycle inventory. The inventory is subsequently characterized by means of environmental model in the life cycle impact assessment step. Finally, the interpretation of the results is a critical step to ensure that all the elements of the study are properly captured.

By accounting for inputs and outputs (respectively, materials, energy, and emissions) at each step of the product life cycle, LCA supports the identification of hotspot of impacts and allows the comparison of options. The LCA is a multicriteria assessment methodology as it covers a wide variety of pressures and impacts associated with human health, ecosystem health, and resources. The LCA is one of the methodologies that makes the Life Cycle Thinking (LCT) operational; in particular, LCA is widely recognized the state of the art relating to the environmental dimension of sustainability (Sala et al., 2013a,b; Finnveden and Moberg, 2005).

Life cycle assessment may play a relevant role all along the decision making process. Indeed, the life cycle perspective and the systemic approach to the evaluation of options is a crucial added value. However, when the scope of the assessment changes from the product (micro) scale to the system (meso-macro) scale, several improvements are required to benefit the most from the LCA methodology. Suitable frameworks, methods, and tools for system analysis are needed to properly develop sustainable policies on, for

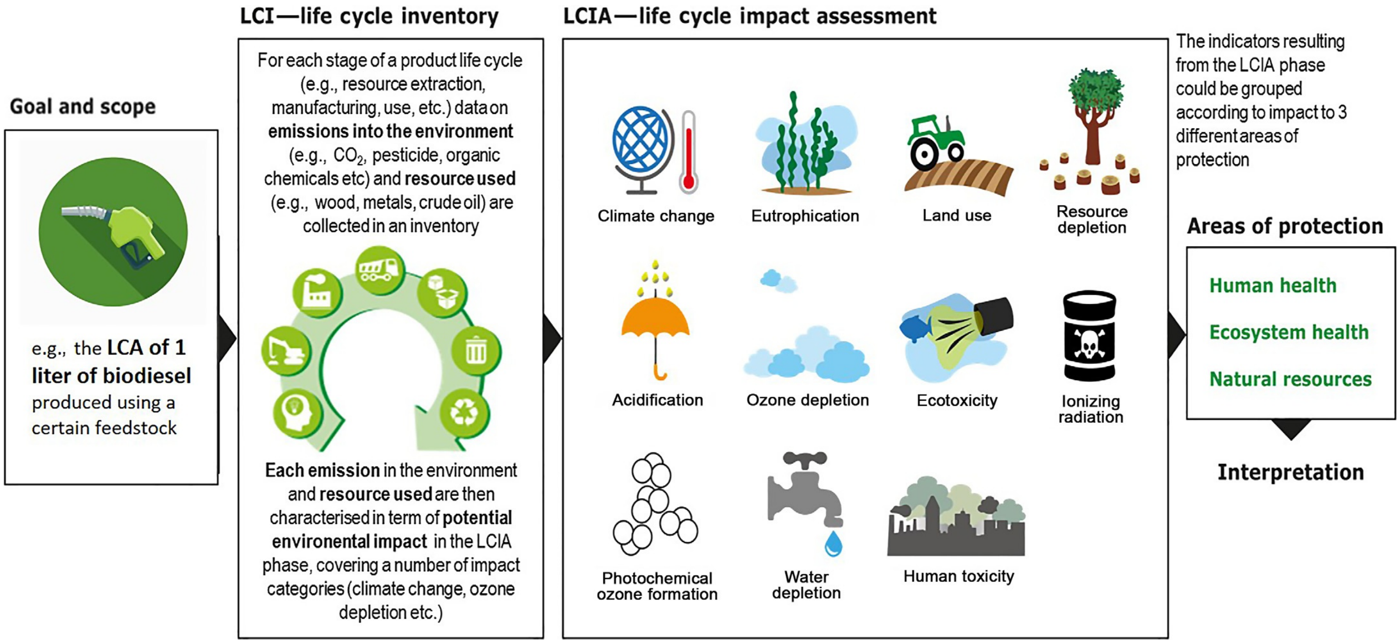

Fig. 3.3 The main methodological steps of an environmental LCA goal and scope definition, life cycle inventory, life cycle impact assessment, and interpretation. *(Modified from Sala, S., Reale, F., Cristobal-Garcia, J., Marelli, L., Pant, R., 2016. Life Cycle Assessment for the Impact Assessment of Policies. EUR 28380 EN. https://doi.org/10.2788/318544.)*

example, bioeconomy, circular economy, resources efficiency, eco-innovation, and sustainable production and consumption. This calls for reflecting upon current and future challenges of the application of LCA as decision support tool.

LCT and LCA have a strong link with the sustainable development goals (SDGs) mentioned before. In fact, LCT and LCA may play a role in assessing impacts and benefits associated to several goals, both environmental and socioeconomic ones. For example, through LCA it is possible to account for climate change-related drivers of impact and the associated potential damage to ecosystems due to production and consumption patterns. Similarly, the assessment framework may cover impact on water, land, resources, and so on. When life cycle thinking is applied to social issues (social LCA), the supply chains related impact could be assessed, for example, those related to poverty or inequalities (see section on social sustainability). Moreover, at European level, LCA is considered the best framework for assessing the potential environmental impacts of products, process, and systems, for example, in the context of the European Environmental Footprint for products (PEF) and organizations (OEF) (CEC, 2013a,b).

The literature on the application of LCA to biofuels has been thriving over the last 10 years (see, e.g., Martin et al., 2015; Malça and Freire, 2011) basically highlighting the main trade-offs and the need of specific methodological improvements to be able to comprehensively assess the environmental sustainability of biofuels production and use.

Often the LCA studies are coupled with models and concepts coming from other disciplines, for example, with economic modeling (Panichelli and Gnansounou, 2017), cost-benefit analysis (Møller et al., 2014), process optimization (van Boxtel et al., 2015), and so on.

Some of the key and most challenging aspects to be further developed for improving environmental sustainability assessment are reported in the following sections.

4.2 Comprehensive evaluation of environmental impacts

The Life cycle impact assessment step seeks to comprehensively address environmental impacts, unveiling trade-offs among impact categories. Usually, in LCA practice, at least 16 different impact categories are taken into account, such as: climate change; acidification; eutrophication, terrestrial; eutrophication, marine; eutrophication, freshwater; particulate matter; photochemical ozone formation; human toxicity, cancer; human toxicity, non-cancer; ecotoxicity, freshwater; land use; water use; resource use, minerals and metals; and resource use, fossils, ionizing radiation, ozone depletion (EC, 2017).

However, some limitations still exist in the models used for assessing the impacts and some impacts are still not completely captured.

Regarding biofuels assessment, a number of improvements will be needed in support to current models and methods.

In the land use modeling there is the need both of assessing more comprehensively impact on soil quality properties (Vidal Legaz et al., 2017; De Laurentiis et al., 2019) and to rely on a better basis for assessing biodiversity loss due to habitat changes (Curran et al., 2016; Chaudhary et al., 2016). Biodiversity is indeed considered a pivotal impact which assessment should be more systematically addressed (Koh and Ghazoul, 2008), including the role of deforestation and global changes related to biofuels expansion (Keles et al., 2018).

In the water use modeling, the impact assessment model recommended by the United Nation for Environment and Setac (UNEP-SETAC) life cycle initiative is the AWARE model (Boulay et al., 2018). The model requires data with spatial and temporal details which are very often not available in life cycle inventories, basically hampering a more specific assessment of water scarcity-related impacts. The development of archetypes may help to overcome the need of very detailed spatially and temporally resolved input data. Scenarios could be run to explore the domain of variability of results and to highlight in which context (as combination of spatial and temporal dimensions) they provide the worst or the best conditions.

Regarding the ecotoxicity potentially associated with the agricultural stage of feedstock, toxicity models are considered still in need of improvements in terms of substance coverage and comprehensiveness of impacts covered (Saouter et al., 2017a,b). A recent work by the European Commission's Joint Research Centre has focused on improving the toxicity evaluation, in terms of number of substances that could be assessed and updating the calculation principles (Saouter et al., 2018). For example, there are increasing concerns related to the loss of ecosystem services, such as pollination. Currently, attempts are ongoing to ensure pollinators are included in the LCA framework (Crenna et al., 2017). In general, more ecological consideration on feedstock production and biofuels production is needed, including the assessment of the carrying capacity of ecosystems (Martire et al., 2015) as well as the accounting of the environmental impacts associated to biotic resources use (Crenna et al., 2018).

Other challenges are present in both the inventory (Corrado et al., 2018) and at impact assessment stage and are affecting in general the modelling of the agricultural stage (Sala et al., 2017).

Moving from the production of the feedstock to the production biofuels, the lack of data related to biochemical transformation and the frequent need of defining proxy processes may reduce the discriminating power of LCA when comparing options between chemical production processes (Piccinno et al., 2016).

4.3 Absolute sustainability: Assessing biofuels in light of planetary boundaries

To illustrate the complexity and multidimensionality of the earth carrying capacity, the concept of planetary boundaries has been put forward and a number of thresholds identified for environmental pressures such as climate change, nutrient load, and so on (Rockström et al., 2009; Steffen et al., 2015). Several boundaries are not yet defined and pose serious challenges in their assessment, for example, those related to chemical pollution (Sala and Goralczyk, 2013). Over the years, studies focusing on the operationalization of the planetary boundary concept within LCA have been published (Ryberg et al., 2016; Clift et al., 2017) aiming at integrating absolute considerations into LCA, overcoming the mere use of LCA for comparative assessment. A set of factors to be used in LCA covering 16 impact categories has been presented in Sala et al. (2019). Beyond this, another crucial challenge is related to the need of allocating the boundaries. Notwithstanding, in theory, each human being on the earth should have the same allocation of the boundary, the debate is open. To operationalize the planetary boundaries concept, there are schools of thoughts for which there is the need of translating boundaries into and aligned with targets that are relevant at these decision-making scales (Häyhä et al., 2016).

4.4 The nexus challenges: Assessing interplays and interdependencies between food, energy, land, water, and ecosystems

Biofuels pose clear challenges to sustainability assessment as they are often related to competing use of resources which are needed for food, materials, and energy production.

The nexus approach and assessment starts for the consideration that the changes in the availability of water, land, and energy supply would strongly affect production of food, including the secure access thereof, with severe implications for human health (Gerbens-Leenes et al., 2009). Therefore, to answer to the need of assessing the interplay between the different sectoral demands and identifying win-win strategy of global resources management, the nexus concept among food, energy, water, land use, and ecosystems has been proposed (Ringler et al., 2013).

Karabulut et al. (2018) suggested a matrix for the operationalization of nexus assessment toward the identification of the main interlinkages between the different resources and proposed a theoretical framework for integrating nexus and LCA.

Studies applying the nexus concept to biofuels are often focusing on emerging economies, where the magnitude of impact will vary significantly across regions and countries depending on the size of the biofuel targets adopted, the identified technologies and feedstock, and especially the water availability and scarcity level (see, e.g., Silalertruksa and Gheewala, 2019). Optimization of biofuels production systems may profit from a nexus approach. For example, a biorefining system could be designed ensuring that the interactions with the surrounding watershed are taken into account, and the supply chain for the production and distribution of feedstocks, grains, and biofuels is compatible with local water and land requirements (López-Díaz et al., 2018). Recently, the assessment of nexus has been coupled with input-output matrixes (Bellezoni et al., 2018) for assessing possible consequences of future scenarios of biofuels expansion.

From the modeling point of view, several challenges are in common to food and biofuel assessment (Sala et al., 2017) whereas others are specific of feedstocks for bio-based products (Mirabella et al., 2013).

4.5 Closing the loop: A circular bioeconomy to foster the use of sustainable feedstocks

Both international and European policies are advocating a transition toward "bioeconomy," an economy aiming at reducing the dependence from fossil-based resources, limiting greenhouse gas emissions, safeguarding food security, and ensuring a sustainable economic growth. Besides, circular economy policies are aiming at closing loop of resources as much as possible. Increasingly, studies are performed to assess sustainability of bio-based alternatives, starting from energy applications up to materials and products (e.g., Mirabella et al., 2013). In fact, the application of circular economy principles to bioeconomy could represent a valuable contribution to bioeconomy performance optimization (Corrado and Sala, 2018). However, both bioeconomy and circular economy may imply environmental burdens if an integrated assessment encompassing all life cycle stages of production and consumption is note performed to unveil trade-offs. From the trade-offs analysis, solutions to maximize benefits could be designed.

5 Social sustainability

Social welfare is considered one of the main development goals of modern society. Understanding and assessing what could improve or undermine well-being is a key element in public policies, aiming at ensuring social and economic benefits while reducing both social and environmental impacts. The appraisal of social impacts and benefit is very difficult and controversial as cultural elements, different values, and lifestyles may affect the way social issues are perceived. Regarding product policies, social impacts along supply chains are increasingly assessed by different stakeholders, such as governments, businesses, and NGOs. To assess impact along supply chains, life cycle-based methodologies have been developed over time. Social life cycle assessment (S-LCA) integrates traditional life cycle assessment methodological steps while having social impacts as focus. Coupling the assessment of environmental and socioeconomic issues may support more comprehensive sustainability assessment of impacts, benefits, and related trade-offs. Social sustainability may be assessed using a variety of methods and indicators, such as the social footprint, social impact assessment, or well-being indices.

Compared to environmental LCA, social LCA is still in an infant stage (Sala et al., 2015b). However, the basic methodological principles and steps are the same and are illustrated in Fig. 3.4.

The UNEP guidelines on social life cycle assessment (S-LCA) (UNEP, 2009) present key elements to consider for product-level, life cycle-based social sustainability assessment. This includes guidance for the goal and scope definition, inventory, impact assessment, and interpretation phases of S-LCA. Methods for and studies of the broader scale, life cycle social dimensions of production and consumption are largely unavailable to date. Pelletier et al. (2018), for example, assessed social risks associated with trade-based consumption in EU Member States using a life cycle-based approach compared to a non life cycle-based approach in order to assess the value added of life cycle thinking and assessment in this context.

Social LCA studies on biofuels exist (e.g., Macombe et al., 2013; Ekener-Petersen et al., 2014; Interlenghi et al., 2017).

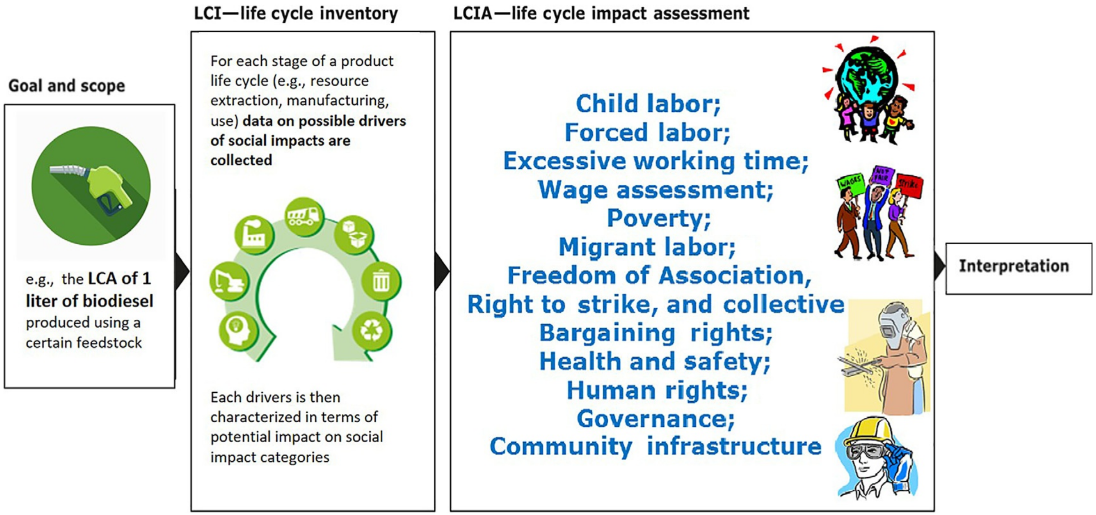

Fig. 3.4 The main methodological steps of social LCA, including goal and scope definition, life cycle inventory, life cycle impact assessment and interpretation. *(Modified from Sala, S., Reale, F., Cristobal-Garcia, J., Marelli, L., Pant, R., 2016. Life Cycle Assessment for the Impact Assessment of Policies. EUR 28380 EN. https://doi.org/10.2788/318544.)*

The studies unveil several challenges for a complete social impact assessment (which encompass data availability, granularity of the data at the product level, completeness of indicators, etc.). Moreover, it is important to assess not only negative impacts but as well positive ones (Di Cesare et al., 2018). In fact, biofuels may bring social benefits. However, positive social impacts are still evaluated to a limited extent (Di Cesare et al., 2018; Ekener et al., 2018) in literature.

5.1 Governance-related challenges in the biofuels domain

Measuring biofuel sustainability implies dealing with a wide array of complex and conflicting values at stake. Consequently, the biofuel capacity to contribute to one specific value cannot lead to any absolute conclusion about the overall sustainability of biofuel (Baudry et al., 2017). Some authors have worked on identifying the different sustainability criteria adopting a stakeholder-based approach in which the different stakes are explicit and transparently reported (see the example for France in Baudry et al., 2017).

In the biofuels domain stakeholders are very different (e.g., government and NGOs, feedstock producers, biofuel producers, refining industry, fuel distributors, users/consumers) and seeking for criteria fulfillment is very challenging.

Evidence- and science-based decision-making in this field need a robust and transparent integrated assessment of policy options. Scientific findings do not lead straight to political conclusions, and the relationship between science and decision-making is a debated issue. Barriers still exist and the effective interaction and communication between scientific enquiry and decision making is complex.

6 Economic sustainability

The need for decoupling economic growth from resource consumption and from environmental impacts is considered one of the pivotal aspects to be addressed by sustainable development. Decoupling takes place "when resource use or some environmental pressure either grows at a slower rate than the economic activity that is causing it (relative decoupling) or declines while the economic activity continues to grow (absolute decoupling)" (IRP, 2017, p. 7). Indeed, decoupling can take place at two levels: resource decoupling and environmental impact decoupling. Quantitative measures of decoupling result from comparing the economic output (e.g., Gross Domestic Product, GDP) with indicators of resource use (e.g., Domestic Material

Consumption, DMC), environmental pressures (e.g., CO_2 emissions), or environmental impacts (e.g., global warming potential).

However, aiming at the decoupling may be not sufficient if the absolute pressure generated on the environment is overcoming the earth carrying capacity, surpassing the abovementioned planetary boundaries. Hence, economic sustainability should aim at ensuring both economic growth and development within the limits of the planet.

This implies evaluating not only direct cost or gains, but as well the so-called externalities, namely, cost or benefit that affects a party who did not choose to incur that cost or benefit (Buchanan and Stubblebine, 1962).

Besides, energy policies, including shift to new feedstock may be vulnerable to rebound effects, namely, when the results of an intervention stimulate a behavioral and systemic responses. Yet, the implications of policy-induced rebounds are mostly unknown since most studies have focused on costless and exogenous efficiency improvements that are not linked to any specific policy intervention (Vivanco et al., 2018). The rebound effect concept is based on the reinforcing relationship between resource efficiency and resource use, where efficiency changes are met with behavioral and systemic responses, such as consumer and market responses to price changes, which result in additional demand and resource use. Rebound effects are typically described as the benefits that are offset once considering such additional resource use.

While the existence of rebound effects is widely accepted, there is disagreement about their magnitude. Estimates range from a moderate offsetting of environmental gains, to a complete elimination of such gains, depending on the boundaries, methods, scope adopted.

7 Conclusions

Sustainability science is a growing discipline, which integrates natural science and social science to identify solutions and assess options toward a more sustainable present and future. Regarding the role of biofuel, they have been considered particularly controversial for what concern sustainability. Despite being based on renewable resources, their production and consumption could be associated with unintended burdens, requiring a systematic assessment toward optimizing their production and reducing their impacts. Among methods for impact assessment, Life cycle assessment and life cycle-related approaches (such as social LCA) are considered particularly promising in order to assess biofuel sustainability in a holistic manner, considering the entire biofuel life cycle and a multiplicity of environmental and social indicators. However, to ensure that even a more comprehensive and strategic assessment is performed key concepts should be taken into account. Assessing biofuels against the carrying capacity of ecosystems, the planetary boundaries, the nexus concept, potential of using more secondary materials for biofuels production in a circular bioeconomy are essential for achieving sustainable development goals for the environmental dimension. Similarly, a full supply chain approach is needed for assessing socioeconomic benefits and burdens, including externalities, spillovers, and rebound effects.

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CHAPTER 4

Indicators for sustainability assessment of biofuels: Economic, environmental, social, and technological dimensions

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1 Introduction

With the growth of the world population and the progressive increase in living standards, the consumption of goods and energy has also increased, along with changes in soil use and deforestation, intensive agriculture practices, industrialization, and consumption of fossil fuel-based energy. All this has contributed to a gradual increase in the concentration of greenhouse gases (GHG) in the atmosphere and decrease in the available fossil fuel reserves.

The Paris Conference on Climate Change confirmed the international commitment of maintaining the increase in global average temperature "well below 2°C with efforts directed to limiting temperature increases in 1.5°C above preindustrial levels." However, the current trajectory of emissions is more aligned with a 4.0°C increase until 2100, and even if the promises made before the Paris Agreement and the intended contributions are effectively carried out, there is 66% probability of not fulfilling the objective established (Sharmina et al., 2017).

Among the 17 objectives published by the United Nations to transform our world, the 7th states that until 2030, international cooperation must be reinforced to facilitate access to clean energy, including renewable energy, energy efficiency, and advanced and cleaner fossil fuel technologies, promoting investments in energy infrastructure and in clean energy technologies (ONU, 2015).

Bioethanol and biodiesel are the most important liquid biofuels employed in the transportation sector in the world (REN21, 2018). Obviously, there are several studies that have evaluated different types of biomass for energy purposes (Bergmann et al., 2013; Ho et al., 2014; Chacartegui et al., 2015; Manochio et al., 2017; Veljković et al., 2018; Ambat et al., 2018; Delgado et al., 2018; Araujo et al., 2018; Neves et al., 2018; Coelho Jr et al., 2018). However, some studies (Seabra et al., 2011; Yanez Angarita et al., 2009) have demonstrated that the best biomass types for the production of ethanol and biodiesel are sugarcane and African palm (dendê), as these cultures present high biofuel yield, $7.6\,m^3\,ha^{-1}\,year^{-1}$ ethanol (Leal et al., 2013a) and $5\,t\,ha^{-1}\,year^{-1}$ oil (Bergmann et al., 2013). Besides, these cultures are highly available and therefore can be utilized for energy purposes.

In fact, there are several possibilities for the utilization of high amounts of biomass that are more efficient than current practice. In biorefineries, biomass can produce not only electricity and biofuels, but also chemical products and food. This represents the creation of a new chain of added value to the biofuel industry, capable of contributing to make it less vulnerable to market variations (IEA, 2014; Vaz, 2011).

Within this context we can highlight the proposals of integrated production models, also referred to as biorefineries, which have been highlighted as alternatives to improve interaction in the production of bioenergy, chemicals, and food from the sustainable processing of biomass (Fatih Demirbas, 2009; Ghatak, 2011; Garcia-Nunez et al., 2016; Ali et al., 2015).

2 Technological aspects

2.1 Biofuels

The term biofuel refers to solid, liquid, or gaseous fuel that is predominantly produced from renewable sources (biomass) (Fatih Demirbas, 2009). There are different sources of biomass for biofuel production, such as oleaginous plants (macaúba, sunflower, colza, dendê, pinhão-manso, algae, buriti), cereals (maize, wheat, barley), agricultural and forestry products, industrial and domestic organic residues, and animal fat and used frying oil (Hoekman, 2009).

According to Directive 2003/30/EC of the European Parliament, which was transposed to Brazilian legislation by decree-law no. 62/2006, the following are considered biofuels:

- *Biodiesel:* methyl ester produced from vegetal or animal oils, which presents fuel properties for diesel engines. It is mainly obtained from oleaginous plants such as palm, dendê, colza, soy, mamona, sunflower, and so on, through a chemical transesterification process.
- *Biomethanol:* Methanol produced from biomass through gasification.
- *Bioethanol:* Ethanol produced from biomass or from the biodegradable fraction of residues. It is produced from the fermentation of sugar, found sugarcane, wheat, maize, potato, and so on.
- *Biogas:* Fuel gas obtained from biomass or the biodegradable fraction of residues (agricultural or livestock, agroindustry and urban effluents) that can be purified to reach the quality of natural gas. It is a result of the anaerobic biological degradation of the organic matter contained in the residues.
- *Dimethyl bioether:* dimethyl bioether produced from biomass.
- *Bio-ETBE (ethyl tertiary-butyl ether):* ETBE produced from bioethanol. In France, it is utilized as oxygenate additive to lead-free gasoline formulations.
- *Bio-MTBE (methyl tertiary-butyl ether):* fuel produced from biomethanol.
- *Synthetic biofuels:* Synthetic hydrocarbons or mixtures produced from biomass.

- *Biohydrogen:* Hydrogen produced from biomass and/or the biodegradable fraction of residues.
- *Pure vegetable oil produced from oleaginous plants:* Crude or refined oil produced by pressure, extraction, or comparable methods from oleaginous plants.

3 Production of biofuel in biorefineries

Biorefineries can be defined as industrial installations that convert biomass and other biological raw materials into products capable of being utilized in the transformation industry such as chemical resources, biofuels, energy (heat and power), among others (Kamm and Kamm, 2007). These installations do not encompass only one process or technology, as different conversion routes can be utilized in function of the resource employed and products to be obtained, according to the type of biomass (Bio2Value, 2015) (Fig. 4.1).

Biorefineries are part of the research, development, and innovation agenda of most developed and developing countries, such as Brazil, mobilizing public and private efforts and high amounts of resources directed toward the optimized utilization of resources, to add value to the productive chain of biomass and reduce possible environmental impacts associated (Fava et al., 2015; Vaz, 2011).

Therefore it is necessary to know the possibilities of implementing a biorefinery, regarding the availability and type of biomass. Besides, it is necessary to select the basket of products, taking into consideration some sustainability aspects. Knowledge on the inherent characteristics of conversion processes, along with the technological development degree and its limitations is paramount for a viable and sustainable utilization of resources, besides helping in the decision-making process (Cardona and Moncada, 2016).

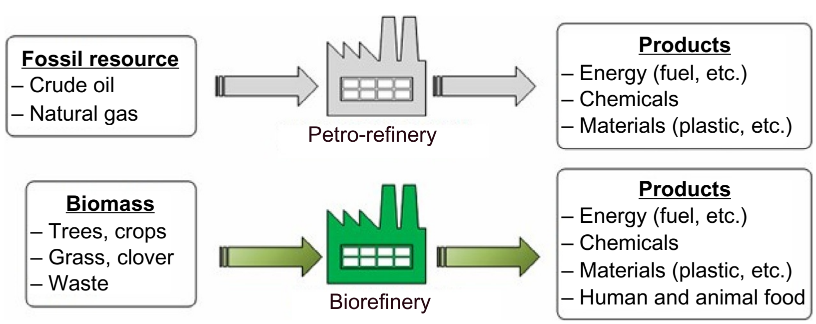

Fig. 4.1 Biorefinery versus petro-refinery.

An important aspect in the development of biorefineries is finding the route that provides the best economic and energy gains, with the least social and environmental associated impacts (i.e., the most sustainable route) (Ishiyama and Paterson, 2011).

Knowledge on the conversion process enables the optimization of the available processes and obtainment of more sustainable products in comparison with those produced from conventional resources. Fig. 4.2 depicts all the factors to be considered in the implementation of an optimized biorefinery.

The following indicators must be considered when determining the best option for the biorefinery:

- *Raw material utilized:* characteristics, resources employed in its production, productivity, environmental impacts, among others.
- *Energy conversion technologies utilized:* efficiency of processes, economic costs, development stage, and so on.
- *Products:* Energy, food, fuels, chemical products, and so on.

Evaluation of the current stage of development of conversion technologies is very important. Some conversion mechanisms are already consolidated at a

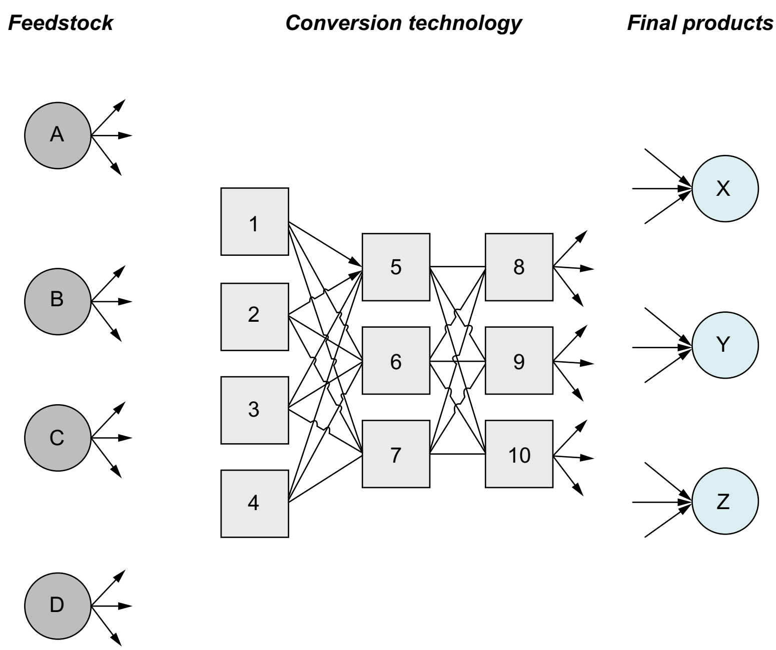

Fig. 4.2 Relevant factors for optimizing of biorefineries. *(Modified from Murillo-Alvarado, P.E., Ponce-Ortega, J.M., Serna-González, M., El-Halwagi, M.M., 2013. Optimization of pathways for biorefineries involving the selection of feedstocks, products, and processing steps. Ind. Eng. Chem. Res. 52, 5177–5190.)*

commercial scale, but others are still under development or being studied. Many promising routes are still at research stages. Fig. 4.3 presents the main biomass conversion routes and indicates the technological development stage of each one (Lora and Venturini, 2012).

There are several possible products and conversion processes to be considered in a biomass conversion process, as depicted in Fig. 4.3. These products can be obtained in integration with existing infrastructures.

4 Biochemical routes

Biochemical conversion processes have the objective of hemicellulose and cellulose polymers fractionation into sugar molecules. Once extracted, the sugar can be fermented by microorganisms, and as a result the desired products are obtained. These processes are called second-generation processes (E2G), and the extracted sugar is fermented similarly to conventional ethanol.

Lignocellulose ethanol is an alternative that enables an increase in plant productivity for ethanol production, without the need to increase planted area. Considering that the productivity of a sugarcane field is 80 tonnes per hectare and that an annexed distillery produces 85 L of ethanol per tonne of sugarcane, the productivity of ethanol is 6800 L per hectare of sugarcane. Considering that 1 tonne of bagasse generates 149.3 L of lignocellulose ethanol, for each tonne of bagasse employed in this process, the productivity per area increases approximately 2.2%, without any additional area planted (Walter and Ensinas, 2010; BNDES, 2008; Leal et al., 2013b).

In addition, as practically all the ethanol production in the world is first generation, E2G could contribute to minimize questions related to land competition between crops for food production and biofuels (PETROBRAS, 2013; BNDES, 2008). Cellulosic ethanol production in a biorefinery would also contribute in several ways to sustainability, such as increase in production without the need to expand cultivated area, reducing GHG emissions and production costs, favoring higher food security and reducing land competition. Another product that can be manufactured in biorefineries is biobutanol, which has received attention from the academic field. Due to its possibilities as fuel and industrial feedstock, biobutanol has the potential to become a renewable chemical commodity (Ndaba et al., 2015). Average annual butanol production growth is 4.7%, with the United States, Europe, and China being the largest global consumers. This increase corresponds to 2.9 million tonnes per year, and most of this alcohol is produced by petrochemical route (Mariano et al., 2014). Butanol is miscible in several solvents such as alcohols, ketones, aldehydes, ethers, glycols, aromatic hydrocarbons, having very limited water miscibility. When used as

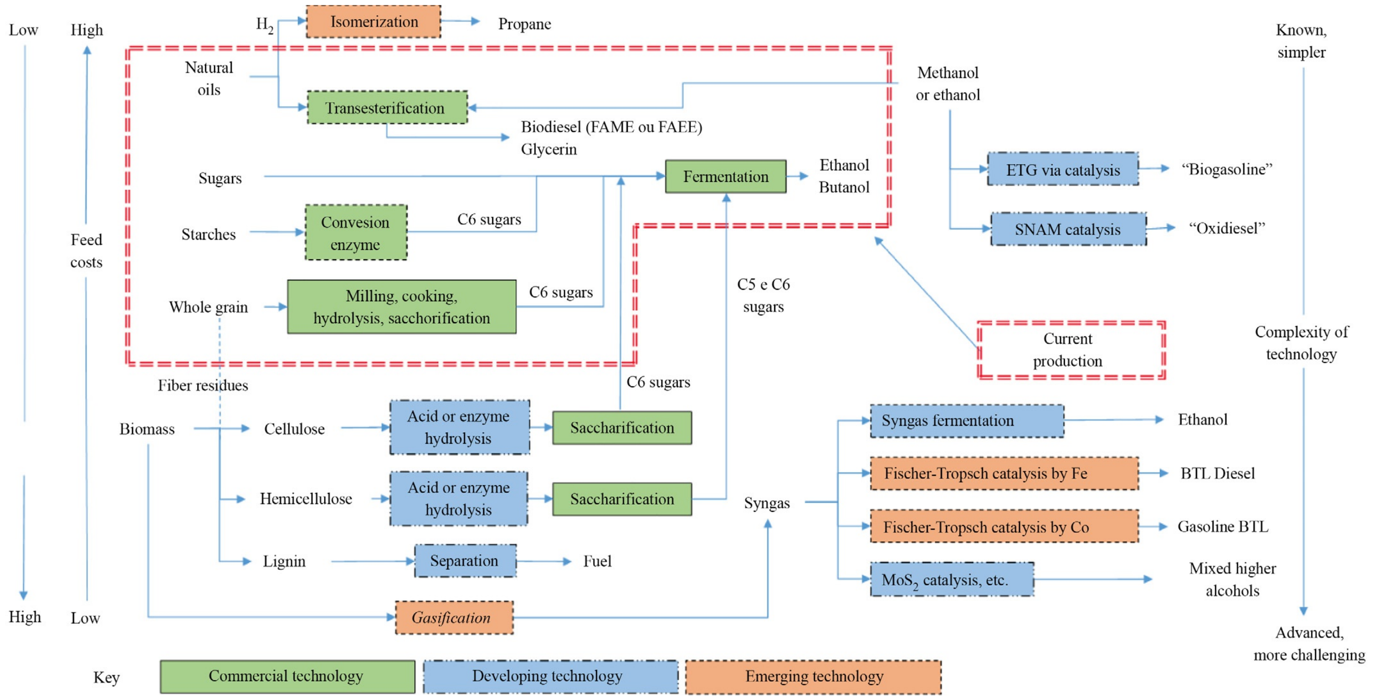

Fig. 4.3 Technological routes and development stages of different biomass conversion technologies. *(Modified from Lora, E.E.S., Venturini, O.J., 2012. Biocombustíveis, vol. 1. first ed. Editora Interciência, Rio de Janeiro, 1149 p.)*

a diesel or gasoline additive, it provides better fuel properties (Ndaba et al., 2015). In addition, butanol can be used as a solvent for paints and varnishes, or as a raw material for production of other chemical inputs, such as *n*-butyl acrylate, also an important monomer for production of polymers and emulsions used in paints. Other uses include *n*-butyl acetate as solvent, as well as glycol, plasticizers, extraction of active pharmaceutical and cosmetics production (Natalense and Zouain, 2013; Mascal, 2012). Nowadays, butanol production is accomplished by three different processes: Acetone-butanol-ethanol (ABE) fermentation, acetaldehyde condensation, and hydroformylation (OXO) synthesis, the latter being the most employed process in the world. The ABE process uses carbohydrate-rich raw materials, such as molasses, which in recent years have been increasing in price, and being a source of food, this can interfere with food safety. Limitations of the ABE process include its low yield, high fermentation times, and problems related to product inhibition. However, it is possible to use renewable raw materials that do not compete with food crops, such as residual crop biomass (Singh and Singh, 2011).

Table 4.1 presents a summary of the main steps utilized in the biochemical route for production of biofuels.

Table 4.1 Summary of the processes contemplated in the biochemical route

Step	Objective	Characteristics
Pretreatment	Disrupt the vegetal tissue, making the cellulose polymers accessible	Instantaneous compressions and decompressions by means of steam injection at high pressures in acid environment
Hydrolysis	Convert cellulose and hemicellulose polymers into fermentable sugars	Enzymes act in the rupture of chemical bonds, breaking the polymers into glucose molecules
Separation of solids	Separates the hydrolyzed solution from components that interfere with fermentation	Takes place in centrifuges that separate the sugared solution, fibrous parts, and other components that interfere in fermentation. Enables the recovery of the lignin
Fermentation	Convert sugar into the desired products, through the action of bacteria	Under adequate conditions, bacteria metabolize sugar and the products are obtained
Distillation and dehydration	Separates and purifies the final products for final destination	Due to the difference in the volatility of the fermented solution, the most volatile compounds evaporate

The production of biobutanol and lignocellulose bioethanol presents almost the same productive steps, and the main difference is the fermentation process that utilizes the Clostridium microorganism, which transforms the sugars of cellulose and hemicellulose into butanol, acetone, and ethanol (ABE fermentation).

5 Thermochemical routes

Besides the biochemical routes, there are thermochemical routes that convert biomass in several industrial products, through physical processes. In thermochemical routes, there is transformation of the solid biomass into liquid products, through pyrolysis, or into gaseous products, through gasification. Pyrolysis is a process that occurs without any oxidizing agents, in which biomass is submitted to high heat transfer rates at temperatures that reach 700–1000°C (temperatures can also be above these values). Due to the high heat transference rates, the product is heated in a very reduced time. This process is not fully commercially established (Basu, 2013).

Gasification is a consolidated process for raw materials based on carbon and oil, but it is still not commercially available for biomass (Higman, 2015; Dahlquist, 2013). In this process, partial oxidation of biomass occurs with oxygen amounts corresponding to 20%–30% of the stoichiometric molar amount necessary for combustion. This gas is a very versatile product that can be a resource for the production of several chemical products, biofuels, electricity, and so on (Rezayan and Cheremisinoff, 2005).

Syngas is a gas mixture composed mostly of hydrogen (H_2), carbon monoxide (CO), methane (CH_4), carbon dioxide (CO_2), also having other gases in lower concentrations. This gas can be used as raw material for the manufacture of chemical inputs, in addition of serving as fuel (Dahlquist, 2013). In theory, biomass can replace most products derived from the petrochemical chain, through platforms that include sugar conversion, or through processes that use syngas as input.

Syngas can be used for the manufacture of so-called building blocks, which in turn can be used for the production of fumaric acid, methanol, hydrogen, and glycerol, of great importance in transportation, textiles, food, pharmaceuticals, and cosmetics (Basu, 2013; Werpy and Petersen, 2004). Fig. 4.4 depicts the possibilities of syngas applications.

In recent years, there has been an increase in the use of syngas for the production of chemical products from coal residues, being more intense in China (Higman, 2015).

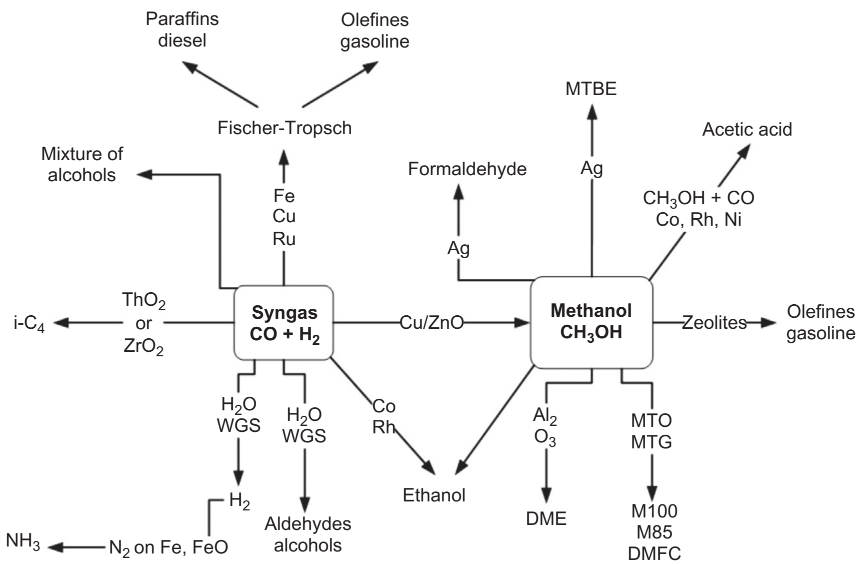

Fig. 4.4 Possibilities and applications of syngas (CGEE, 2010).

Fischer-Tropsch synthesis enables the exploitation of biomass by converting syngas into carbonic chains, finally resulting in liquid and solid hydrocarbons. Such process could represent an alternative to the use of crude oil, as the manufactured products are similar to those produced by petrochemical industries. Commercially consolidated production plants based on the Fischer-Tropsch synthesis process are already distributed worldwide, but present high production costs. These would become more economically feasible as oil prices rise. Additionally, the majority of these plants use fossil-based raw materials such as coal and natural gas. This process can be an opportunity for countries that do not have oil reserves, to manufacture products for the petrochemical industry chain without depending on market price fluctuations and demands (Vliet et al., 2009; Takeshita and Yamaji, 2008). However, depending on the location and the available energy utilities, it is interesting to carry out economic analyses and consider purpose-grown biomass and transportation. Peat fuel—a biomass energy source—is abundant in Northern Ontario, and in this case, the biomass procurement cost would be much reduced as the resource is close to, or on-site (Carvalho and Millar, 2012).

It is important to consider that the energy value of biomass differs according to the type, and that the ash content is also different, which determine the characteristics of the biofuels produced by each selected route. Therefore fuels that are produced from biomass have different properties and characteristics, depending on the type of biomass employed.

According to Jong and Jungmeier (2015), biorefineries could be classified in accordance with the selected route:
- Biorefineries that utilize oleaginous crops for the production of biodiesel, glycerin, and food.
- Biorefineries that utilize C6 sugars and amylaceous crops for the production of bioethanol and livestock food.
- Biorefineries of synthesis gas for the production of diesel, gasoline, naphtha, methanol, and other FT fuels.
- Lignin biorefineries that use wood and C5 and C6 sugars for the production of ethanol, electricity, heat, and phenols.

6 Sustainability in biorefineries

It is necessary to establish the sustainability degree of a biorefinery, in relation to conventional production techniques, and then compare the sustainability of different biorefineries as well as the environmental impacts generated by biomass conversion.

However, sustainability is a complex aspect to analyze, as the concept encompasses environmental, economic, and social variables. Currently, quantitative assessment of sustainability comprehends the calculation of economic, social, and environmental indicators, separately. The most commonly employed indicators are as follows:
- Economic indicators (production costs).
- Energy and exergy efficiencies.
- *Net energy ratio*—NER, relationship between the net input and output of energy.
- Fossil fuel substituted per hectare.
- Carbon emissions: avoided GHG emissions (reduction in the CO_2-equivalent emissions).
- Environmental impacts (different impact categories).
- Carbon emissions due to land use.
- Renewability indicators (exergy).
- Social indicators.

With these indicators, it is possible to compare different production schemes, providing decision tools for the implementation of a project.

Regarding the evaluation of environmental impacts, life cycle assessment (LCA) is a key methodology that enables the quantification of potential environmental and social impacts for products, processes, and activities, providing an indication of sustainability (Araujo et al., 2018; Neves et al., 2018; Coelho Jr et al., 2018; Carvalho et al., 2019).

Recent studies related to biofuel LCA, generally, compare different fuel production routes from energy and environmental points of view. The main questions approached by a biofuel LCA are (Gnansounou, 2018) as follows:

– What biofuel production route causes the lower environmental damage?
– Are there differences in the selection of biomass, and which one is ecologically better for the different production processes?
– What is the attributable share of environmental impacts for each stage of production?
– Are there margins for improvement?
– What is the environmental behavior of the biofuel if there are changes in the studied scenario?

Jungmeier et al. (2013) and Sacramento-Rivero (2012) discuss the particularities of the evaluation of sustainability in biorefineries considering the existence of multiple energy, food, chemical, and biomaterial products.

In this sense, LCA can be applied in combination with optimization methods, as it has already been combined in other areas: with economic analyses (Carvalho and Abrahao, 2017), with thermoeconomics (Silva et al., 2017), and within optimal energy supply systems (Serra et al., 2014).

The work carried out by Gebreslassie et al. (2013), for example, utilized a multiobjective optimization process in an algae-based biorefinery, and maximized the NPV (net present value) while minimizing the GHG emissions (Fig. 4.5).

Based on the indicators of the processes employed within a biorefinery, we can obtain the material and energy flows associated with the inputs and outputs of the system, and then determine the CO_2 emissions (or any environmental indicator) associated, following with a comparison, as shown in Fig. 4.6.

Based on refinery data, it is possible to establish minimum, maximum, and ideal (optimal) values for different indicators, for the several types of biorefineries. By plotting these data in a radar diagram (Fig. 4.7), we can have an idea of how much each aspect contributes. The larger area is considered the most sustainable.

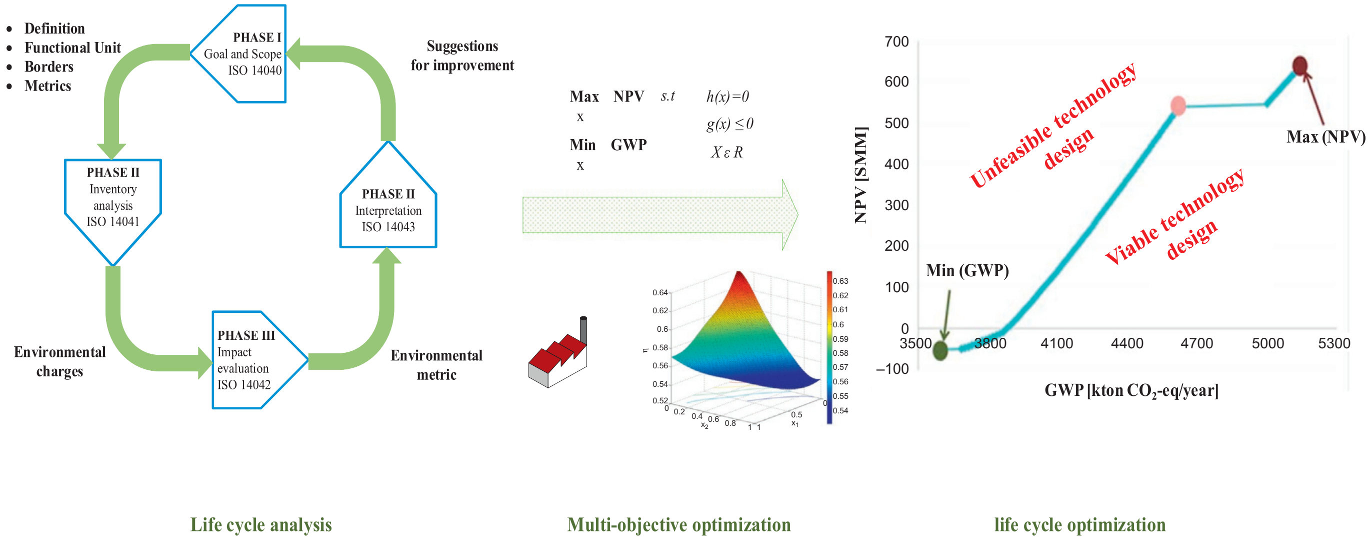

Fig. 4.5 Combination of LCA and multiobjective optimization at an algae-based biorefinery (Bergmann et al., 2013).

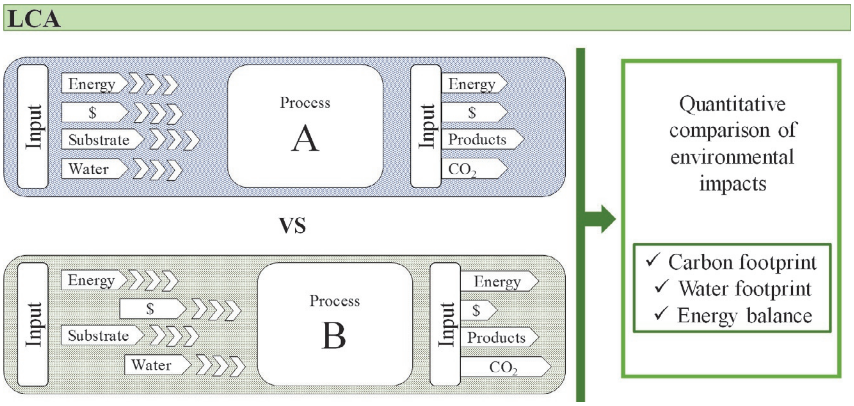

Fig. 4.6 Model of the indicators to be considered in the LCA comparison of alternative biorefineries (Bauzá, 2015).

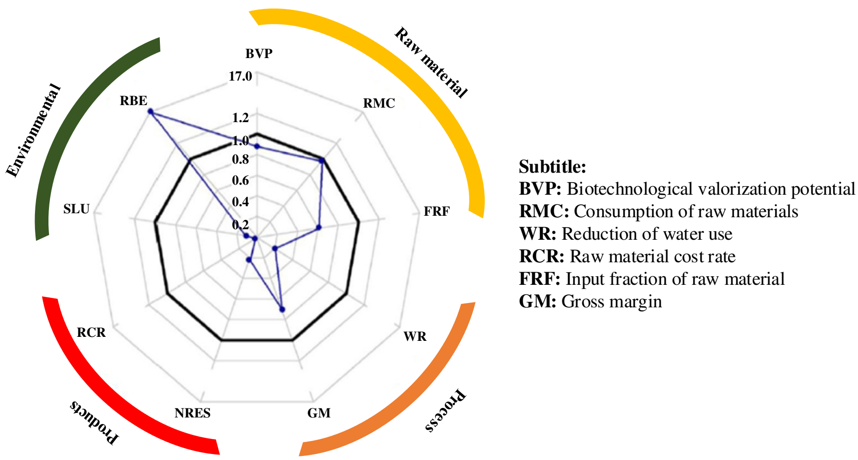

Fig. 4.7 Radar diagram for the evaluation of biorefinery sustainability. *(Modified from Navarro, F.S.P., Vilchis, L.E., Sacramento-Rivero, J.C., 2014. Aplicación de una nueva metodología para la evaluación de la sostenibilidad de biorefinerias. Memorias del XXV Encuentro Nacional de la AMIDIQ, pp. 3281–3286.)*

Other studies are based on the comparison of mass and energy balances between production processes that contemplate biorefineries and conventional processes. Both mass and energy balances must be as rigorous as possible. Fig. 4.8 depicts a comparison between energy balances and GHG emissions for a biorefinery and for a reference, conventional refinery.

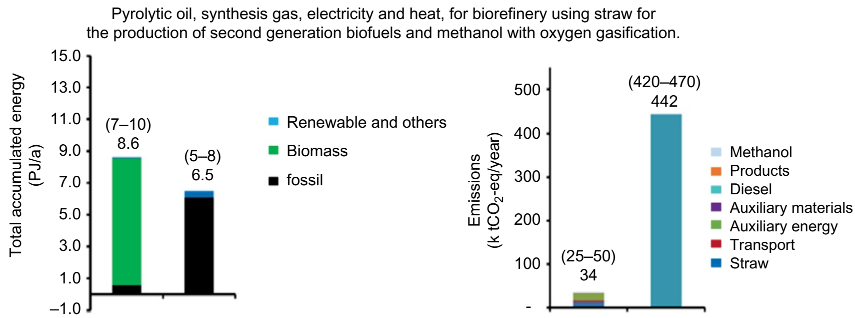

Fig. 4.8 Comparison between the energy balance and GHG emissions for a biorefinery and a reference case (Jungmeier et al., 2013).

7 Study case

With the objective of evaluating the impact of biofuel, food, and electricity production technologies on the energy and sustainability indicators of a specific biorefinery that uses sugarcane, six study cases will be considered. Each case utilizes different thermochemical or biochemical production routes, as follows:

- *Case zero (ZRO):* Based on a configuration currently available in the Brazilian sugar and ethanol sector, and considers that all straw and bagasse are burnt in the steam cogeneration cycle boilers for the production of electricity.
- *Case base (BSE):* Considers the incorporation of the thermochemical gasification route for the production of Fischer-Tropsch fuels and electricity from the BIG-GTCC cycle (Biomass Integrated Gasifier-Gas Turbine Combined Cycle). The biochemical route is employed for the production of lignocellulose ethanol and products of ABE fermentation. The amount of bagasse available is equally distributed to all biorefinery processes, and therefore each process receives a third of the available bagasse. Syngas produced via gasification is equally distributed between Fischer-Tropsch synthesis and BIG-GTCC cycle.
- *Case E2G (lignocellulose ethanol):* Considers the biochemical route for the utilization of all bagasse available at the plant, for the production of lignocellulose ethanol. The lignin generated in the process is used as fuel in the steam cogeneration boilers.
- *Case B2G (biobutanol):* Considers the biochemical route for the production of biobutanol, using all the available bagasse in ABE fermentation. As in case E2G, the lignin produced is used as fuel in the steam cogeneration boilers.

- *Cases FT (Fischer-Tropsch):* Considers the thermochemical route and the utilization of all the available bagasse available at the syngas production plant for Fischer-Tropsch synthesis. This process is characterized by the high demand of energy in the processes that constitute the plant. It was considered that a BIG-GTCC cycle would meet part of this demand, using 20% of the syngas produced.
- *Case BGT (BIG-GTCC):* Considers the thermochemical route for the production of electricity, using all the available bagasse in the gasification process. Therefore the produced syngas will be used as fuel for a gas turbine within a BIG-GTCC cycle, and does not require adjustments in its composition, as occurred indifferently from the FT case.

Each case assumes a sugar and ethanol industry that is energetically self-sufficient, with electric mills. The self-produced electricity operates equipment such as pumps, agitators, transporters, and lighting.

After the syrup extraction process, bagasse is obtained with a 50% humidity content. For each tonne of sugarcane processed, 280 kg on humid basis is obtained, which corresponds to the available bagasse (Marino, 2014; Olivério, 2014; Hassunai et al., 2005). Straw represents 14% of the sugarcane available in the field (Marino, 2014; Olivério, 2014; Hassunai et al., 2005), and therefore for each tonne of sugarcane cropped, there is 140 kg straw left in the field. Assuming a plant with a processing capacity of 1200 tonnes of sugarcane per hour, this value corresponds to 86% of the sugarcane available in the fields, and the remaining 14% refer to straw (equivalent to 195.35 tonnes/h).

It is recommended that the maximum amount of straw removed from the field is 50%, as half of the straw should be left on the soil to reduce erosion, allow the recirculation of nutrients and maintain the humidity levels of the soil (Marino, 2014; Hassunai et al., 2005). The amount of available straw (humidity content 15%) for utilization corresponds to 97.68 t/h, and herein the rounded value of 97.60 t/h will be considered. Regarding bagasse, part is mixed with straw and used as fuel for the steam boiler, in a 50%–50% proportion, as suggested by the manufacturer CALDEMA (Marino, 2014). It is also recommended that a 10% share of the bagasse generated after syrup extraction is stored (CGEE, 2010), which corresponds to 33.6 t/h.

After this separation, there is 204.8 t/h of bagasse that can be utilized as a resource in the biorefinery processes, which contemplates biochemical and thermochemical conversion routes.

Fig. 4.9 presents the general scheme of the biorefinery, in which the aforementioned processes are integrated within an existing plant. Table 4.2 presents the operational parameters adopted for the simulation, common to all study cases. LHV refers to the lower heating value.

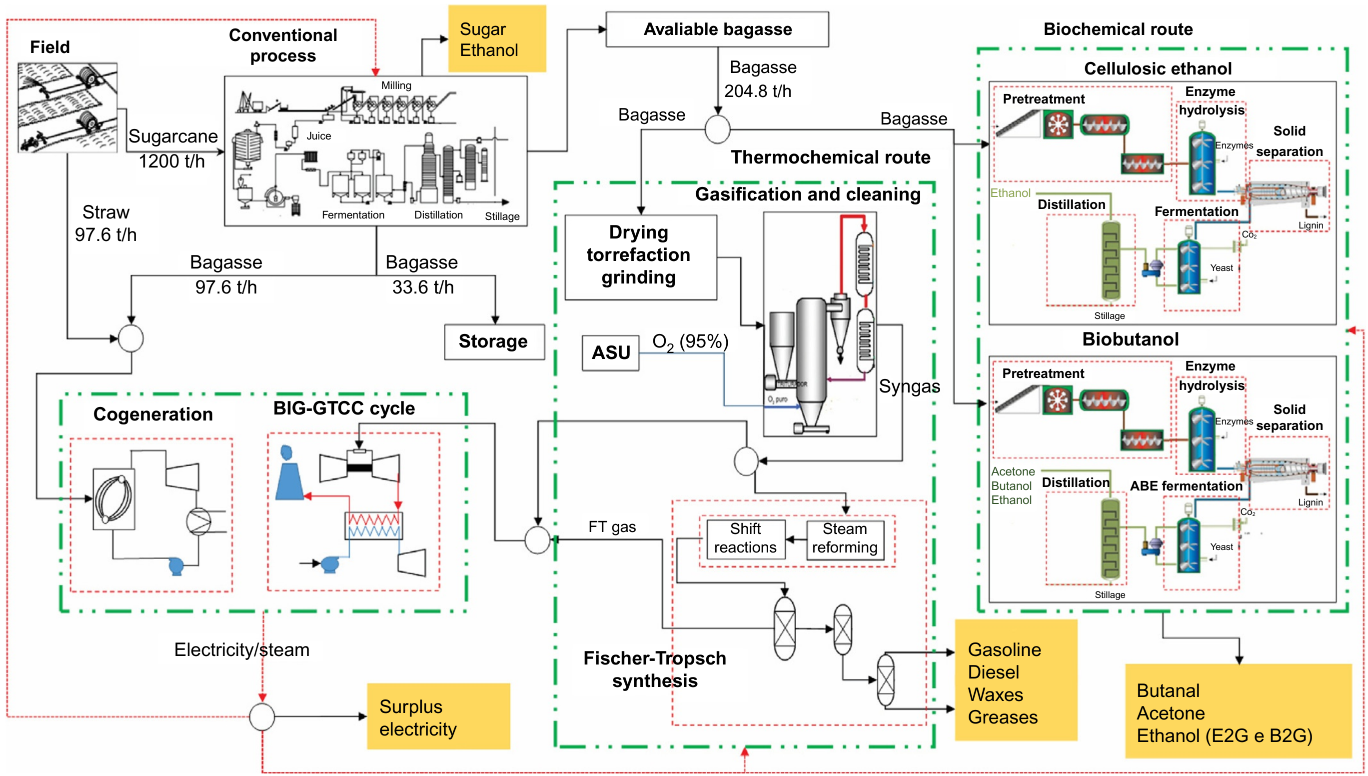

Fig. 4.9 General scheme of the sugarcane biorefinery.

Table 4.2 Operational parameters of the plant (BNDES, 2008; CGEE, 2010; Hassunai et al., 2005)

Parameters	Values	Unit
Milling	1200	t_c/h
Amount of bagasse[a]	336	t_b/h
Electrical consumption of the mills	16	kWh/t_c
Specific electrical consumption of the process	12	kWh/t_c
Steam consumption of the process	420	kg/t_c
Ethanol productivity[b]	42	L/t_c
Sugar productivity[b]	67	kg/t_c
Amount of bagasse	97.60	t/h
Humidity content of bagasse	50	%
Bagasse LHV (50% humidity)	7650	kJ/kg
Amount of straw	97.60	t/h
Straw LHV (15% humidity)	12,900	kJ/kg
Amount of fuel	195.2	t/h
LHV of the mixture	10,275	kJ/kg
Lignin productivity	590	kg_{lignin}/t_b
Lignin LHV	11,740	kJ/kg

t_c, tonne of sugarcane; t_b, tonne of bagasse.
[a]Considering a fiber content of 14% on dry basis.
[b]Assuming that 50% of syrup is destined to ethanol production and 50% to sugar production.

8 Main parameters adopted in the biochemical conversion processes

8.1 Second-generation ethanol

The operational parameters utilized by the second-generation ethanol plant are presented in Table 4.3.

Based on the parameters presented in Table 4.3, and considering the indications of Ojeda et al. (2011), the mass balance of the plant is obtained, and shown in Fig. 4.10.

8.2 Biobutanol

The same operation parameters presented by Nuncira (2013) were adopted (Table 4.4), who carried out a simulation for a cellulosic butanol plant utilizing software Hysys.

Fig. 4.11 presents the mass and energy balances for the cellulosic butanol plant.

Table 4.3 Operational parameters of the cellulosic ethanol plant (Walter and Ensinas, 2010)

Parameter	Value
Steam consumption (37 bar)[a]	1.14 $kg_v/L_{ethanol}$
Steam consumption (2,5 bar)[a]	4.60 $kg_v/L_{ethanol}$
Electricity consumption	92.5 $kWh/t_{bagasse}$
Productivity	149.3 $L_{ethanol}/ton_{bagasse}$

[a]Saturated steam.

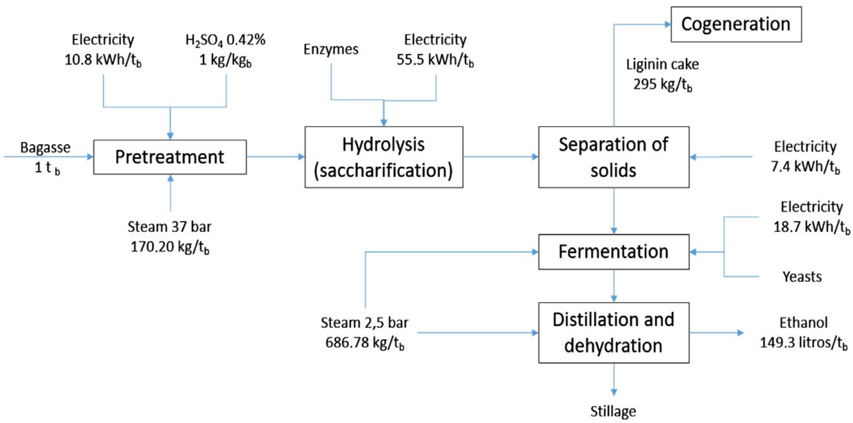

Fig. 4.10 Main mass and energy flows associated with the cellulosic ethanol production process.

Table 4.4 Operational parameters of a cellulosic butanol plant (Nuncira, 2013)

Parameter		Value
Steam consumption (12 bar)[a]		650 kg $_v/t_{bagasse}$
Steam consumption (2,5 bar)[a]		802 kg $_v/t_{bagasse}$
Electricity consumption		18 $kWh/t_{bagasse}$
Productivity	Butanol	57.2 $L_{butanol}/t_{bagasse}$
	Acetone	15.7 $L_{acetone}/t_{bagasse}$
	Ethanol	4 $L_{ethanol}/t_{bagasse}$
	Vinasse	3600 $L_{vinasse}/t_{bagasse}$

[a]Saturated steam.

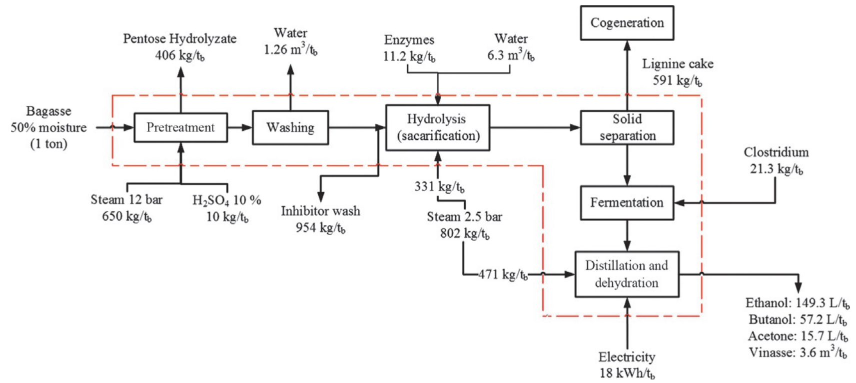

Fig. 4.11 Mass and energy balances for the cellulosic butanol production process.

9 Thermochemical conversion processes

For the sake of clarity and simplicity, the thermochemical step will be subdivided into pretreatment, syngas production, syngas conditioning, and Fischer-Tropsch synthesis. Fig. 4.12 shows the different steps in accordance with the processes. Mass and energy balances are also presented for the processing of 1 tonne of bagasse. These balances consider the utilization of syngas according to the Case Base (BSE), where 50% of the syngas is sent to the BIG-GTCC cycle and the remaining 50% is sent to FT synthesis.

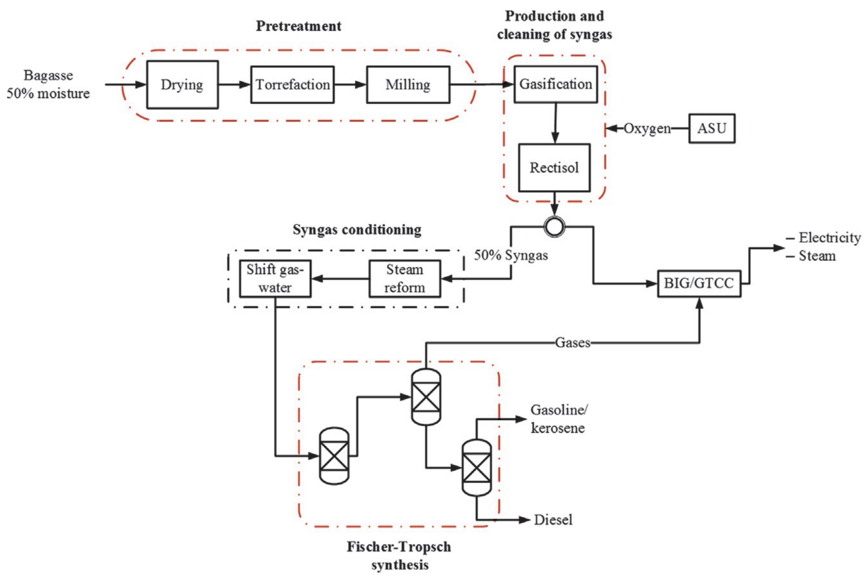

Fig. 4.12 Processes employed in the thermochemical route.

10 Pretreatment of bagasse

The pretreatment step comprehends drying, roasting, and crushing, and prepares biomass for the fluidized bed gasification process, providing adequate granulometry and humidity, and improving energy density.

11 Production and cleaning of syngas

Production and cleaning of syngas encompasses the gasification steps in fluidized bed and cleaning of the obtained gas. Gasification is a complex process, sensitive to several factors such as gasification temperature, equivalence ratio, biomass granulometry, and demands oxygen with elevated purity.

Therefore the production step of syngas includes a unit for air separation, which produces the oxidizing agent for gasification. The quality of syngas limits its utilization in other processes, and therefore cleaning has the objective of removing unwanted compounds, enabling its utilization in BIG-GTCC cycles and in FT synthesis.

Fig. 4.13 presents the mass balance of the syngas production step. It considers the energy flows associated with the processing of 1 tonne of sugarcane for the Base Case (BSE), where 50% of syngas goes to the FT synthesis and 50% to the BIG-GTCC.

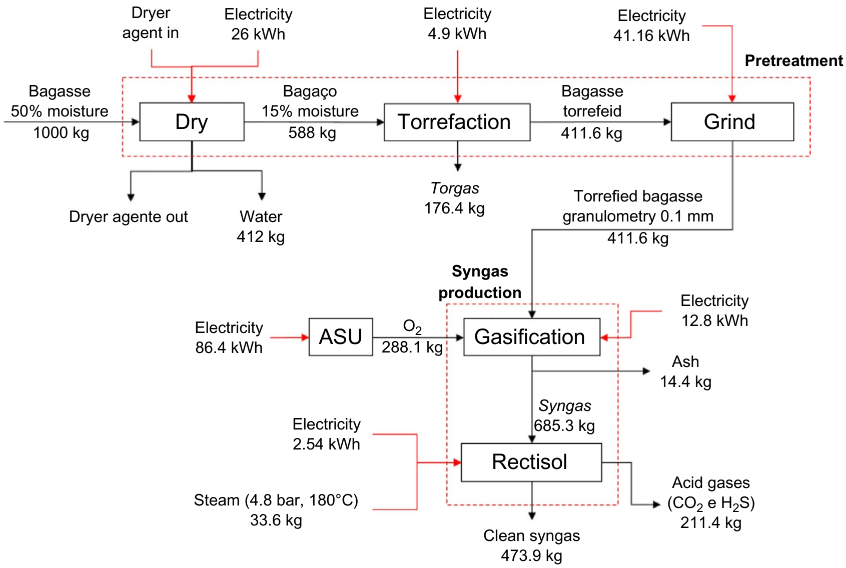

Fig. 4.13 Mass and energy balances for the production and cleaning of syngas.

12 Syngas conditioning

After the removal of unwanted compounds, syngas presents elevated purity, but adjustments are still necessary in its composition before its utilization in the production of FT fuels. These adjustments consist of the conversion of the methane present in the gas into CO and H_2, and of the increase of the ratio between the concentrations of hydrogen and carbon monoxide (ratio H_2/CO) closer to a 2.01 value in accordance with the requirements of the FT synthesis (Tijmensen et al., 2002).

13 Fischer-Tropsch synthesis

In the FT synthesis step, fuels are produced from syngas, through the conversion of carbon monoxide and hydrogen into water and the CH2 monomer, which will form the hydrocarbon chain, as shown by reaction (4.1), called polymerization reaction. Iron-based catalysts are utilized, as these do not suffer poisoning as occurs with CO_2 (Tijmensen et al., 2002).

$$CO + 2H_2 \rightarrow -CH_2- + H_2O \ \Delta H^{\circ}_{298} - 165.0\,\mathrm{MJ/kmol} \tag{4.1}$$

Fig. 4.14 presents the mass balance of the adjustment of syngas conditioning and FT synthesis.

Table 4.5 summarizes the processes contemplated by gasification and FT synthesis.

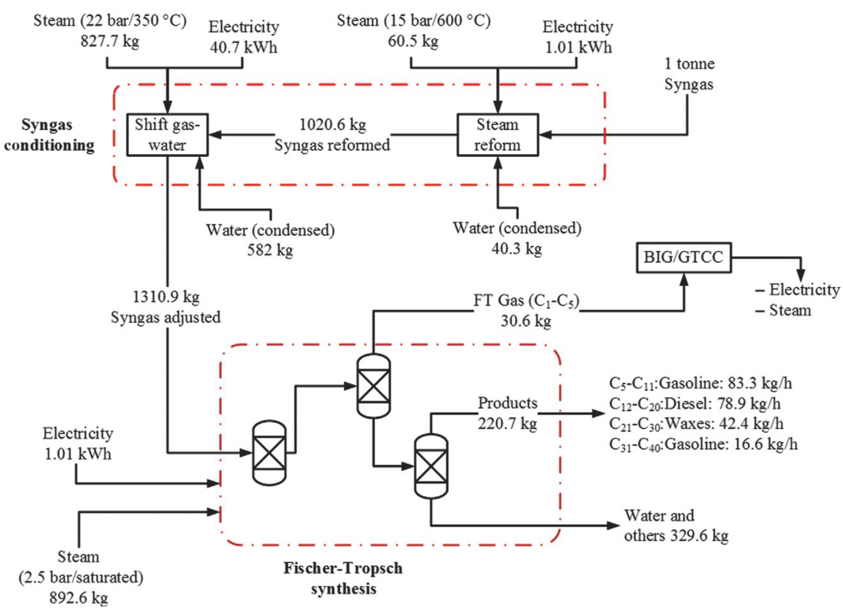

Fig. 4.14 Mass and energy balance for the FT synthesis.

Table 4.5 Summary of the processes involved in the thermochemical route

Step	Process	Operational objectives and characteristics	Unit consumption	Consumption for 1 t_b
Pretreatment	Drying	Reduce the humidity content from 50% to 15%, through the injection of exhaust gases from the cogeneration boiler at 260°C	Electrical: 26 kWh/t_b	26.00 kWh
	Roasting	Provide characteristics similar to charcoal, through heating at 280°C for 2 h	Electrical: 30 kJ/kg	4.90 kWh
	Crushing	Reduction of the biomass granulometry to 0.1 mm. No losses were considered in the crusher and humidity at outlet is 15%	Electrical: 70 $kWh/t_{b\ torr.}$	41.16 kWh
Production of syngas	Gasification	Convert biomass in syngas. Gasification occurs in a fluidized bed at 1150°C and 22 bar, with ER = 0.3	Electrical: 8.1 kWh/MW_{th}	12.80 kWh
	ASU	Produce oxygen with 95% purity for gasification	Electrical: −0.3 kWh/kg_{O2}	86.43 kWh
	RECTISOL	Remove unwanted compounds of syngas (CO_2 and H_2S), which could form acid gases. After gasification, syngas temperature is reduced from 1250°C to 30°C and is compressed from 22 to 40 bar. Syngas is reheated to 800°C	Electrical: 1900 kJ/kmol[a] Steam (180°C and 4.8 bar): 5.97 $kg_{st}/kmol$[a]	2.54 kWh 33.60 kg

Continued

Table 4.5 Summary of the processes involved in the thermochemical route—cont'd

Step	Process	Operational objectives and characteristics	Unit consumption	Consumption for 1 t_b
Syngas conditioning	Steam reforming	Convert all the CH4 into syngas into H_2 and CO. Clean syngas is injected in the reforming reactor at 15 bar and 800°C	Electrical: 3 kJ/kg_{syngas} Steam (15 bar e 600°C): 0.06 kg_{steam}/kg_{syngas}	0.24 kWh 14.31 g
	Shift gas-water	Adjust the relationship between the concentrations of gases, so that the H_2/CO ratio is 2.01. Occurs in two steps: the first is processed in a reactor at 350°C and 20 bar where steam is injected, and the second occurs in a reactor at 200°C and 20 bar	Electrical: 140 kJ/kg_{syngas} Steam (22 bar e 350°C)	9.40 kWh 206.80 kg
Production of liquid FT	FT synthesis	Convert syngas into FT liquids. The synthesis reaction occurs at 25 bar, 220°C with iron-based catalysts to maximize the production of diesel ($\alpha = 0.88$)	Electrical: 31 kJ/kg_{syngas}	2.66 kWh
	Distillation of the FT liquids	Separate the hydrocarbons produced for utilization. Distillation is accomplished through the provision of thermal energy in distillation columns.	Thermal: 1.95 MJ/$kg_{syngas\ reformed}$	31.0 kJ/$kg_{syngas,\ reformed}$

t_b, tonne of bagasse.
[a]kmol refers to the amount of compounds removed (CO_2 and H_2S).

14 Generation of electricity and steam in the BIG-GTCC cycle

The BIG-GTCC has the objective of producing thermal energy (steam) and electricity, complementing the conventional stem cogeneration cycle and sharing the responsibility of meeting the energy demands of the plant. Fig. 4.15 presents a scheme of the cogeneration system that operates with the BIG-GTCC cycle, meeting the thermal and power demands of the processes.

The main parameters utilized in the simulation of the BIG-GTCC cycle are shown in Table 4.6.

Evaluation of the performance of the different study cases considered specific indicators that related the energy consumed and produced, comparing the processes with others on the basis of avoided emissions (substitution

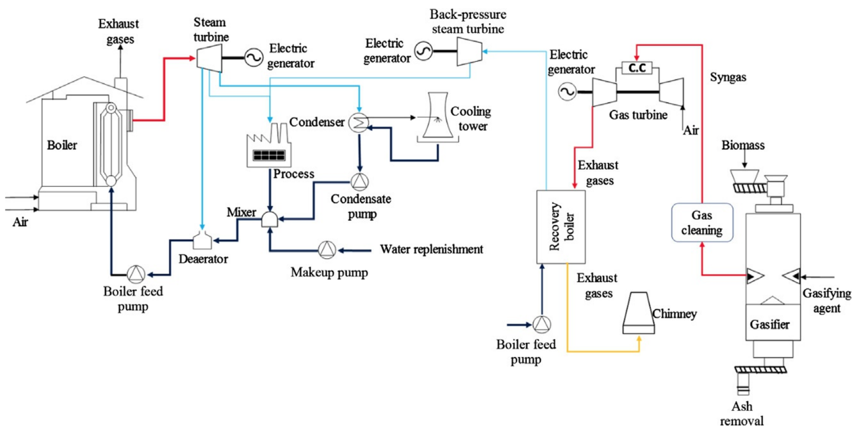

Fig. 4.15 Scheme of a sugarcane processing plant with the BIG-GTCC system.

Table 4.6 Parameters adopted in the simulations and calculations (BNDES, 2008; CGEE, 2010; Escobar et al., 2009; Hassunai et al., 2005)

Parameter	Value
LHV of syngas (after cleaning)	14.52 MJ/kg
LHV of the FT gases	46.90 MJ/kg
Pressure of the recovery boiler	85 bar
Isentropic efficiency of the steam turbines	78%
Pump efficiency	85%
Boiler efficiency	80%
Pressure of syngas at the turbine inlet	30 bar

of fossil fuels by the products produced by the different configurations studied). Therefore the indicators considered herein for the sustainability analysis were global efficiency, net productivity per hectare, and avoided CO_2-eq emissions. These indicators are presented in details as follows.

15 Sustainability indicators

Global efficiency of the plant (η_{global}): Defined as the ratio between the useful energy of the products (ethanol, sugar, cellulosic ethanol, cellulosic butanol, acetone, diesel, gasoline, waxes, and electricity) and the energy of the inputs consumed (syrup, bagasse, and straw). η_{global} is calculated by Eq. (4.2).

$$\eta_{\text{Global}} = \frac{\sum\left(\dot{m}_{\text{product}}\text{LHV}_{\text{product}}\right) + \sum \text{Surplus electricity}}{\sum\left(\dot{m}_{\text{juice}}\text{LHV}_{\text{juice}} + \dot{m}_{\text{bagasse}}\text{LHV}_{\text{bagasse}} + \dot{m}_{\text{straw}}\text{LHV}_{\text{straw}}\right)} \tag{4.2}$$

Energy productivity per hectare ($\sigma_{sugarcane}$): Indicates the utilization of sugarcane energy per hectare. This parameter indicates the amount of sugarcane energy that is actually converted into products, such as ethanol, butanol, acetone, sugar, FT liquids, and electricity per hectare. It is determined as a function of the production of 1 ha of sugarcane crop, as presented in Eq. (4.3).

$$\sigma_{\text{sugarcane}} = \frac{\sum\left(\dot{m}_{\text{product}}LHV_{\text{product}}\right) + \sum \text{Surplus electricity}}{\dot{m}_{\text{sugarcane}}}\Psi \tag{4.3}$$

ψ represents the sugarcane productivity per hectare and corresponds to 80 $t_{\text{sugarcane}}$/ha (Leal et al., 2013b; Rocha, 2015), considering a new sugarcane field. The value of $\sigma_{\text{sugarcane}}$ is expressed in GJ/ha, and the values of the physical properties of the products, such as LHV, density, and so on, are presented in Table 4.7.

Avoided $CO_{2\text{-}eq}$ emissions ($CO_{2\ eq.\ ev.}$): Indicates the amount of CO_2 that is not emitted due to fuel substitution. Ethanol, for example, is a biofuel that can substitute other oil-originated fuels such as gasoline. Therefore the emissions of gasoline, when substituted by ethanol, would be avoided, as ethanol is carbon neutral considering its use as fuel only.

The indicator considers the tonnes of $CO_{2\text{-eq}}$ emissions avoided per crop. Table 4.8 presents the values of the LHV of these products and the emissions avoided.

Table 4.7 Lower heating values considered for the products

Product	Value	References
Ethanol	28.23 MJ/kg	BNDES (2008)
Sugar	16.73 MJ/kg	Renó et al. (2011)
Syngas	14.52 MJ/kg	Calculated
Bagasse (50%)	7.65 MJ/kg	Renó et al. (2011)
Butanol	26.60 MJ/L	Beckwith (2011)
Acetone	28.80 MJ/L	Beckwith (2011)
FT gases$_{C1\text{-}C4}$	46.90 MJ/kg	Calculated
Gasoline$_{C5\text{-}C11}$	44.50 MJ/kg	Gebreslassie et al. (2013)
Diesel$_{C12\text{-}C20}$	44.21 MJ/kg	Gebreslassie et al. (2013)
Waxes$_{C21\text{-}C30}$	46.50 MJ/kg	Gebreslassie et al. (2013)
Fat$_{C31\text{-}C40}$	47.00 MJ/kg	Gebreslassie et al. (2013)
Syrup	2.51 MJ/kg	Renó et al. (2011)
Straw (15%)	12.90 MJ/kg	Walter and Ensinas (2010)

Table 4.8 CO2-eq avoided emissions per product

Product	Emissions avoided	References
Ethanol	0.086 kg $CO_{2\ eq}$/MJ	Väisänen et al. (2016)
Butanol	0.056 kg $CO_{2\ eq}$/MJ	Michailos et al. (2016)
Acetone ABE	0.057 kg $CO_{2\ eq}$/MJ	Väisänen et al. (2016)
Gasoline FT	85 g $CO_{2\ eq}$/MJ gasoline	CGEE (2010)
Diesel FT	3917 g $CO_{2\ eq}$/liter of diesel	Vliet et al. (2009)
Waxes	4210 g $CO_{2\ eq}$/liter of diesel	Vliet et al. (2009)
Fat	4210 g $CO_{2\ eq}$/liter of diesel	Vliet et al. (2009)
Electricity	0.1396 kg $CO_{2\ eq}$/kWh	EPE (2016)

16 Results

Table 4.9 presents the products of each study case, which were utilized for the determination of the sustainability indicators.

It is important to mention that all cases accounted for the ethanol and sugar produced in the conventional process, with energy contents 312.96 and 373.53 MW, respectively, being the same across all cases. Vinasse, generated in the conventional process as well as in the biochemical processes, was not considered as a product, and destined to ferti-irrigation. Its energy content was not considered in calculations.

Table 4.9 Amounts of products involved in each study case

Products	Base	Zero	E2G	B2G	FT	BGT
Ethanol (liters/h)	10,192.2	0	30,576.6		0	0
Butanol (liters/h)	3049.9	0	0	11,714.6	0	0
Acetone (liters/h)	1072.8	0	0	3215.4	0	0
Ethanol (liters/h)	273.1	0	0	819.2	0	0
Gasoline (liters/h)	1686.1	0	0	0	8093.2	0
Diesel (liters/h)	1596.4	0	0	0	7662.6	0
Waxes (liters/h)	857.5	0	0	0	4116.0	0
Fats (liters/h)	335.8	0	0	0	1612.0	0
Surplus electricity (MWh)	196.14	276.52	200.87	199.57	117.44	289.57

17 Global efficiency and net productivity per hectare

Fig. 4.16 presents the results of the global efficiency and net productivity per hectare for the different study cases considered herein.

In terms of global efficiency, which measures how much of the sugarcane energy is converted into products, the results varied between 60.56% for the ZRO case and 52.01% for the B2G case. No improvement was observed in the energy conversion of the plant when biochemical conversion routes were added (E2G, B2G). However, improvements were obtained when thermochemical routes were added (BGT and FT) and when both routes were combined (BSE). In the latter, the results presented an intermediate efficiency value in comparison with the biochemical and thermochemical routes separately.

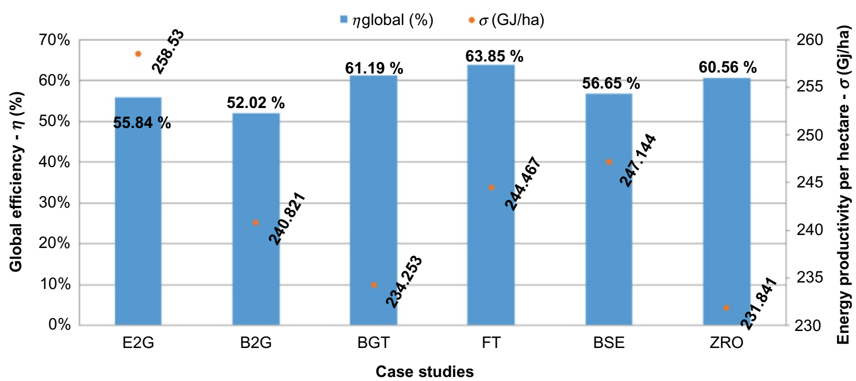

Fig. 4.16 Global efficiency and net productivity per hectare.

The efficiencies obtained for cases E2G, B2G, and BSE are lower than for the ZRO case. The utilization of thermal energy and electricity in the processes strongly influences the final result obtained. Therefore the global efficiencies of energy conversion in the cases that present biochemical routes are lower than the ZRO case due to the high energy demands (ZRO represents the conventional system currently employed by the Brazilian industry). This might seem contradictory, as there are no benefits associated with the diversification of production. However, analysis of the FT case, which presents the higher diversification within the analyzed study cases, demonstrates its highest global efficiency value.

The low energy performance indicated the necessity of increasing the output energy flows. In other words, improvements are still necessary in the processes, aiming at better conversion of bagasse.

However, when energy productivity is calculated per hectare, the worst result corresponds to the ZRO case. This affects land use as it is possible to increase the production of fuels, energy, and food without the occupation of more area. In this sense, the E2G case, with cellulosic ethanol, results in the highest net productivity value (258,532 GJ/ha). It must be highlighted that in the calculations of efficiency and energy productivity per hectare, the energy contents of conventional ethanol and sugar were taken into account in calculations.

In environmental terms, the avoided emissions demonstrate that the best study case was FT (highest avoided emissions), while the worst case was ZRO. Fig. 4.17 depicts the results for avoided emissions.

It is observed that the ZRO case avoided the least emissions (218.56 thousand tonnes per crop, just below the BGT case, with 255.2 thousand

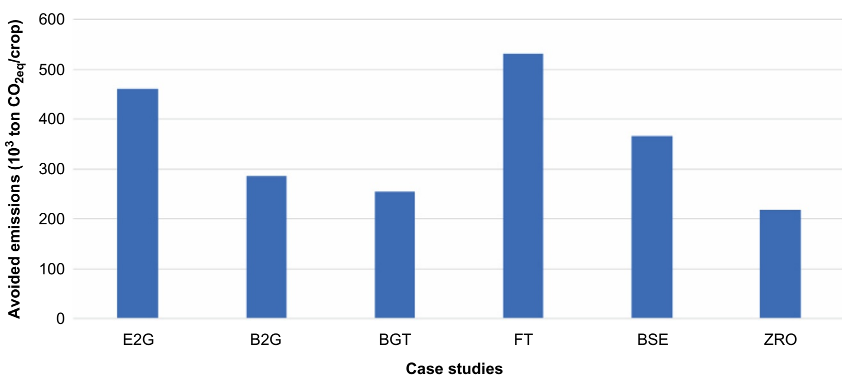

Fig. 4.17 Avoided emissions.

tonnes per crop). In the cases where there are other products besides electricity, there are higher avoided emissions. This demonstrates that the environmental benefits associated with diversification of production in the sugar and ethanol industry reflect on the environmental benefits resultant of the application of the obtained products. The production of biofuels and other chemical products is more attractive, from the viewpoint of avoided emissions, than the production of electricity. This occurs because the Brazilian energy matrix is mainly based on renewable resources.

Comparison of the B2G case with the E2G, FT, and BSE cases highlights much lower avoided emissions than the other processes, because the B2G case presents low productivity. Due to this reason, less products substitute a less fossil fuel-based products, avoiding less emissions.

The FT case, which contemplates the thermochemical route, presented slightly higher performance in comparison with the remaining cases, as more products are obtained yielding a higher substitution of fossil fuels.

18 Comparative analysis

A radar diagram can help compare the results of the study cases, where the indicators are normalized in relation with the highest value. Table 4.9 presents data on the indicators, along with the normalized products. Normalization presented in Table 4.10 consists of calculating the ratio of the difference between the indicator value and the lowest value of this indicator, and the difference between the highest and lowest indicator values. The radar diagram is shown in Fig. 4.18.

Table 4.10 Absolute and normalized indicator values

	Indicator values			Normalized		
Case	**η_{global} (%)**	**σ (GJ/ha)**	**Avoided emissions 10^6 kg de $CO_{2\ eq.}$**	**η_{global}**	**σ**	**Avoided emissions**
E2G	55.84%	258.53	460.54	0.32	1.00	0.78
B2G	52.01%	240.82	286.30	0.00	0.34	0.22
BGT	61.19%	234.25	255.20	0.77	0.09	0.12
FT	63.85%	244.47	530.73	1.00	0.47	1.00
BSE	56.65%	247.14	367.30	0.39	0.57	0.48
ZRO	60.56%	231.84	218.56	0.72	0.00	0.00

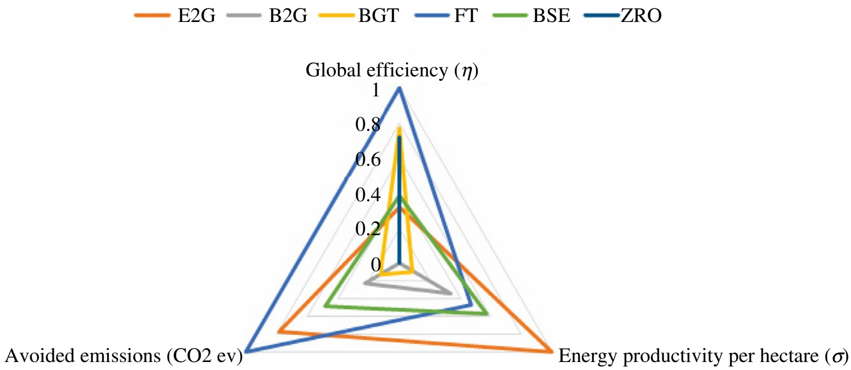

Fig. 4.18 Radar diagram for the study cases.

The cases with the lowest indicators will present normalized value equivalent to zero, which indicates worst performance of the indicator in relation to the remaining cases. The ZRO case, although it presents global efficiency above B2G, E2G, and BSE, presents the worst performance in the other indicators, which contributes for this being the least sustainable case.

The area of each figure can be interpreted as a measurement of sustainability. It is observed that the FT case presents the larger area, which indicates that this study case contemplates the most sustainable process and presents the best combination of indicators. Case ZRO presents the smallest area, indicating that there are other more sustainable alternatives for biomass.

19 Economic indicators

Besides sustainability indicators, it is important to take into account the economic aspect. For electricity generation systems, a widely used indicator is LCOE (Levelized Cost of Electricity), which measures costs over lifetime of the system in relation to the total energy generated, and is given in $ kWh^{-1}. This indicator can be calculated according to Eq. (4.4).

$$\mathrm{LCOE} = \frac{\text{Lifetime costs}}{\text{Total energy generated}} = \sum_{t=0}^{t=T} \frac{\frac{C_t}{(1+r)^t}}{\frac{E_t}{(1+r)^t}} \tag{4.4}$$

where C_t and E_t are, respectively, the costs and the electricity generated in within the period of time t, r is the discount rate, and T is the lifetime of the system. The costs can be expressed in terms of their conventional components as presented in Eq. (4.5):

$$C_t = I_t + O_t + M_t + G_t \tag{4.5}$$

where I is the initial capital cost, O is the operation cost, M is the maintenance cost, and G represents others marginal costs, respectively, in a given period of time t.

However, for a system producing several products such as food, chemical inputs, fuels, and so on, this LCOE is not recommended because each product can have different qualities. Another indicator widely used in the assessment of the economic viability of biorefineries is the net present value (NPV) of the investment, which indicates the financial sustainability of the project, calculated by Eq. (4.6).

$$\mathrm{NPV} = \sum_{n=1}^{t} \frac{\text{Cash flow}}{(1+i)^n} - \text{Total investiment} \tag{4.6}$$

NPV is usually employed because it enables an evaluation in monetary and absolute terms. The use of NPV is recommended when addressing projects of different sizes, where different scenarios are analyzed, with several configurations. Inconsistencies are therefore avoided in the profitability analysis of each project, which could be present if a relative parameter was used to evaluate economic performance. The next section shows an application of the NPV concept to the FT case.

20 Determination of NPV for the Fischer-Tropsch case (FT case)

Calculation of the cash flow for the enterprise should include different financial aspects such as fixed costs (labor costs, maintenance, etc.) and variable costs (raw material costs and other inputs). The investment required for constructing a production unit (plant) can be based on its capacity, and Eq. (4.7) presents an estimate for this value, employing capacity and costs of a known plant with similar components. The initial investment (capital cost or process plant construction costs) can be adjusted from one period to another by the CEPCI index (Chemical Engineering, 2017).

$$\mathrm{Cost}_2 = \mathrm{Cost}_1 \left(\frac{\mathrm{Capacity}_2}{\mathrm{Capacity}_1}\right)^{\alpha} \left(\frac{\mathrm{CEPCI}_2}{\mathrm{CEPCI}_1}\right) \tag{4.7}$$

Table 4.11 Economic parameters used in the analysis

Parameter	Adopted value
Income tax	35%
Other taxes	18
Discount rate[a]	7%
Lifetime	25 years
Construction time	2 years
Depreciation (linear)	10 years
Funding payment period	10 ears
Bagasse price[b]	6.67 US$/$t_c$
Straw price[b]	US$ 19/t_c
Contingencies and working capital[c]	5% of the initial investment

[a]BNDES (2017).
[b]EPE (2016).
[c]Larson et al. (2009).

For the construction of processing plants the value of α corresponds to 0.6 (Dias et al., 2014). The economic parameters used in the analysis are presented in Table 4.11.

21 Cost of gasification and Fischer-Tropsch synthesis plants

There are only a few biomass gasification plants operating on a commercial scale, and some pilot plant projects were discontinued. In fact, there are several technical challenges to be overcome, before any technology can start to operate on biomass. China has been progressively investing in coal gasification plants, importing technology from other countries. Important examples of biomass gasification pilot plants include the Choren project, in Germany, which was designed to gasify 65,000 tonnes of wood per year, the Chermec Project, in Sweden, which produces second-generation bio-kerosene and biogas, and also a biofuel production project in France that converts lignocellulosic raw materials into biodiesel and bio-kerosene (Ett et al., 2014). These high investments represent an obstacle to wider implementation of biomass-fuelled technologies. As there are only a few gasification plants operating at commercial scale, uncertainty can drive up current investment costs to levels that are far from acceptable limits.

Table 4.12 presents the costs of equipment used in gasification and FT synthesis. In the case considered herein, the enterprise revenues are obtained

Table 4.12 Equipment costs of gasification and Fischer-Tropsch synthesis

Stage	Equipment	Cost and capacity	References
Pretreatment	Dryer and Crusher	Cost in 2011: US$ 22.7 million, with reception capacity of 389 MW_{th} of input	Swanson et al. (2010)
	Torrefier (Moving bed)	Cost in 2010: US$ 10.73 million for equipment capable of processing 126 thousand tonnes of biomass per year	Basu (2013)
Syngas production	Gasifier-shell-entrained flow	Cost in 2005: €81 million, with 400 MW_{th}	Vliet et al. (2009)
	ASU unit[a]	Cost in 2003: Total cost of US$ 31.71 million, for an inlet air flow of 18 kg/s	Larson et al. (2009)
	RECTISOL unit[b]	Cost in 2003: Total cost of US$ 122.38 million for a syngas flow of 34.2 kg/s	Larson et al. (2009)
FT synthesis	Steam reformer, Shift reaction reactor, distillation and refining system[c]	Cost in 2003: Total cost of US$ 123.8 million for a syngas flow of 34.2 kg/s	Larson et al. (2009)

[a]Considers the ASU unit, O_2 compressor, and N_2 expander that cost US$ 25.5 million, 4.68 million, 1.23 million, respectively.
[b]Considers gas cooler (recovery boiler), tar filtering and cracking system, RECTISOL Unit and O_2 compressor, which cost US$ 51.6, 26.8, 43.7, 0.28, respectively.
[c]Considers FT reactors costing US$ 38.77 million and reformers (shift and distillation) and refining system costing US$ 85.03 million.

from the sale of produced goods, considering the sale tariffs presented in Table 4.13.

The production costs considered include salaries, maintenance, and operation, and the cost of labor force for an FT processing plant corresponds to 2000 Euros per MW of thermal input (Vliet et al., 2009). Maintenance and operating costs were considered to be 4% of total investment (EPE, 2016; Larson et al., 2009). In BIG-GTCC and cogeneration cycles, maintenance and operation costs were considered to be US$ 144.00 per installed kW and US$ 84.00 per kW, respectively (Dantas, 2013).

Table 4.13 Product tariffs

Product	Price	References
Gasoline	US$ 1.10/L	MME (2016)
Diesel	US$ 1.00/L	MME (2016)
Waxes	US$ 0.22/L	Im-orb et al. (2016)
Greases	US$ 0.22/L	Im-orb et al. (2016)
Electricity	US$ 76.00/MWh	Petersen et al. (2015)

Exchange rate of US$1.00 = R$3.00 and 1 US$ = 0.95 Euros.

The capital cost of plants, with all pretreatment and cleaning systems, ASU, gasifiers, BIG-GTCC system, and so on, corresponds to US$ 632.43 million. After consideration of taxes, fixed and variable costs, a gross profit of US$ 20.34 million per period was obtained. Therefore for the scenario established herein, NPV is negative, corresponding to US$ 583.44 million, demonstrating economic unfeasibility.

Conversion technologies based on the Fisher-Tropsch synthesis are currently in the development phase, and it is still necessary to increase the yields associated with this technology. Table 4.14 presents the influence of productivity increase in NPV, considering 10%, 20%, and 30% increases in productivity and assuming that the investments and costs of production are the same.

The increase in synthesis productivity leads to an increase of US$ 204.33 million in NPV; however, the result is still negative. This indicates that biomass conversion technologies still need to evolve, but it is also necessary to reduce the costs associated with this technology. The high investments required for gasification, FT synthesis, and BIG-GTCC cycle are the major limitation for these technologies to achieve financial attractiveness. This may explain the fact that Brazil does not have thermochemical biomass conversion plants, as the conversion technologies adopted herein are still under development.

In the formulation of the enterprise cash flow, most data were considered constant over time, aiming to present the dynamics of the system's behavior in the long term. In practical terms, all data adopted in the determination of the cash flow vary over time, as happens with the dynamic behavior of the market.

Table 4.14 Influence of productivity increase on the NPV of Fisher-Tropsch synthesis

Productivity increase	Reference case	10% increase	20% increase	30% increase
NPV (mi US$)	−571.91	−503.81	−435.7	−367.59

A sensitivity analysis can help identify which variables affect the economic viability of a biorefinery project, including variations in the prices of products and raw materials, production costs, taxes, investment, and so on. Assuming that the sale tariffs of each biorefinery product varied between −100% and + 100% in relation to the base value, and that the remaining variables were constant, it was found that gasoline presented the highest influence on NPV, as depicted in Fig. 4.19. Although the price of gasoline has a great influence on NPV, even if its price is duplicated, NPV is still not positive.

Diesel is the second product in terms of influence on NPV, and it is interesting to remark that the Fischer-Tropsch synthesis conditions discussed herein aim to maximize Diesel production. Electricity, on the other hand, has a lower contribution to NPV in relation to biofuels, being less profitable in comparison with other products. As the production of greases and waxes is relatively small, these products have little influence on NPV. Regarding production costs, Fig. 4.20 presents, in an analogous way, the influence of each production cost on NPV.

The initial investment (capital cost) has the highest influence on NPV, and if the investment is reduced by 77%, NPV becomes zero. Operation and maintenance costs (O&M), which were assumed to be 4% of the investment, represent the second most influential variable on NPV. Regarding the effect of interest rates, NPV would still be negative at US$ 245 million.

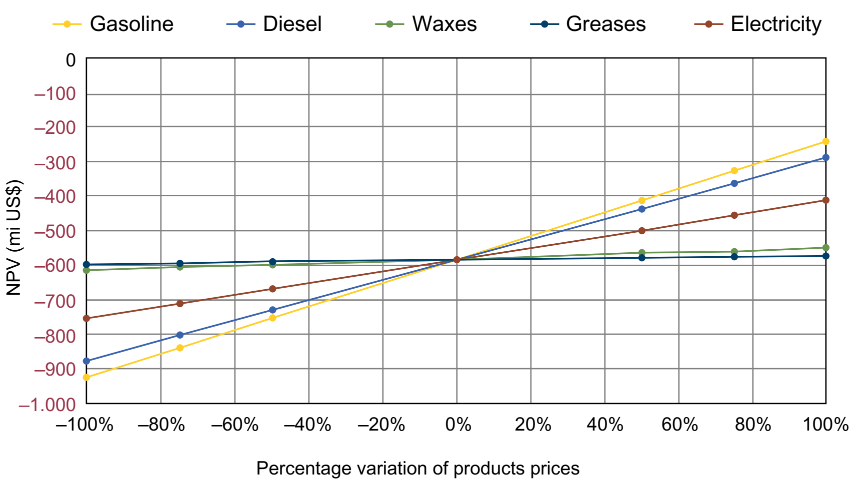

Fig. 4.19 Sensitivity of the product tariffs for the FT case.

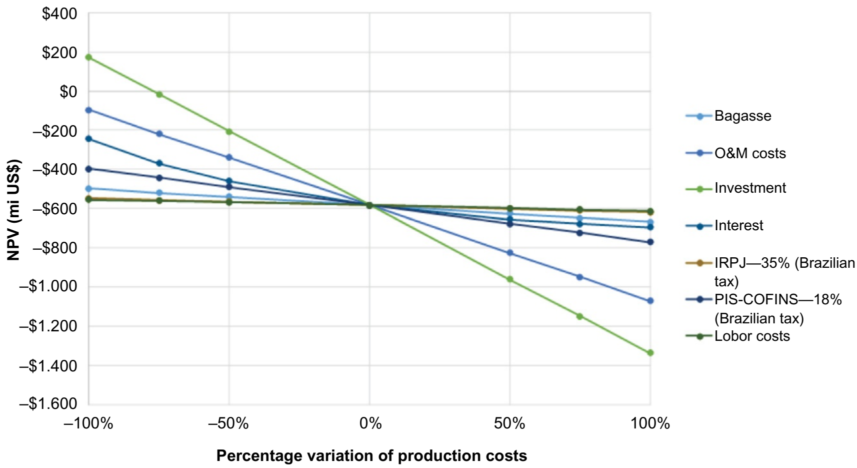

Fig. 4.20 Sensitivity of production and capital costs in the FT case.

The sensitivity analysis, shown in Figs. 19 and 20, indicates the behavior of NPV for variations in economic parameters (e.g., product sale tariffs, capital cost, interest rate, production costs). The higher the slope of the line corresponding to the parameter, the greater its influence on NPV. Therefore the combination of economic analysis and sensitivity analysis enables the evaluation of the economic viability of the enterprise, also determining which products can contribute more significantly to financial viability and what costs influence the final economic result the most.

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CHAPTER 5

Life cycle sustainability assessment in the energy sector

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1 Introduction

The overarching aim of sustainable development is to ensure that all people, in current and future generations, benefit from continuing prosperity (WCED, 1987). The energy sector is critical to this endeavor as it provides the foundations for most other sectors of the economy and society: without a sustainable energy mix, there cannot be sustainable development. It is partly for this reason that the energy sector has seen more vigorous attempts at environmental impact reduction than most other sectors. In particular, attempts to decouple energy generation from greenhouse gas (GHG) emissions have gathered momentum and are drawing enormous industrial

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activity: in 2017 the total global investment in low-carbon electricity generation was USD 315 billion, while energy efficiency initiatives saw USD 236 billion in spending (IEA, 2018).

While the reduction of GHG emissions is critical in the transition to sustainable development, it is important to retain the broader goal of continuing prosperity for all people. This requires that research and development, strategy, and policy-making include a much wider variety of sustainability issues in addition to GHGs. One could argue that this is particularly relevant in the bioenergy sector due to its breadth: the upstream and downstream activities in the sector can span everything from basic waste processing and agriculture to highly advanced conversion and combustion technologies, in the process affecting stakeholders in the entire demographic space. To appreciate this fully, we must first consider the many facets of sustainable development in more detail.

1.1 The UN sustainable development goals as a common frame of reference

Perhaps the most widely used approach to describing sustainability is that which categorizes a wide range of interacting aspects under three overarching "pillars" (United Nations, 2005), otherwise referred to as the "triple bottom line" (Elkington, 1997): environment, economy, and society. In recent years the UN's 17 Sustainable Development Goals (SDGs) (United Nations, 2015), which came into force for member states at the start of 2016, have been widely acknowledged as a common set of themes around which we might further specify the aims of sustainability. The goals are outlined in the following Box.

UN sustainable development goals

1. End poverty in all its forms everywhere
2. End hunger, achieve food security and improved nutrition, and promote sustainable agriculture
3. Ensure healthy lives and promote well-being for all at all ages
4. Ensure inclusive and equitable quality education and promote lifelong learning opportunities for all
5. Achieve gender equality and empower all women and girls
6. Ensure availability and sustainable management of water and sanitation for all
7. Ensure access to affordable, reliable, sustainable, and modern energy for all

8. Promote sustained, inclusive, and sustainable economic growth, full and productive employment and decent work for all
9. Build resilient infrastructure, promote inclusive and sustainable industrialization, and foster innovation
10. Reduce inequality within and among countries
11. Make cities and human settlements inclusive, safe, resilient, and sustainable
12. Ensure sustainable consumption and production patterns
13. Take urgent action to combat climate change and its impacts
14. Conserve and sustainably use the oceans, seas, and marine resources for sustainable development
15. Protect, restore, and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, and halt and reverse land degradation and halt biodiversity loss
16. Promote peaceful and inclusive societies for sustainable development, provide access to justice for all and build effective, accountable, and inclusive institutions at all levels
17. Strengthen the means of implementation and revitalize the Global Partnership for Sustainable Development.

Based on the SDGs we can begin to appreciate the breadth of issues that are relevant to sustainability and the ways in which the energy sector might interact with them. For instance, Goal 7 clearly states that affordable and sustainable energy provisions are, in themselves, desirable goals. However, it is also clear that energy is a crucial enabler of several other goals: it is the single greatest contributor to climate change (Goal 13), accounting for over a third of global GHG emissions (IPCC, 2014), it is essential in the provision of clean water and sanitation (Goal 6), it often involves mining and other causes of land use change (Goal 15), and it is a major global provider of skilled employment (Goal 8) with over 10 million people working directly in the renewables subsector alone (IRENA, 2018).

Accounting for the sustainability impacts of energy systems requires rigorous analysis of the entire life cycle, as will be explored further in the following sections.

1.2 Energy, biofuels, and their relevance to sustainable development

Despite increasing focus on energy efficiency and the deployment of non-fossil energy sources, the world's overall energy production has continued to climb, only recently showing any tendency toward a reduction in the rate of

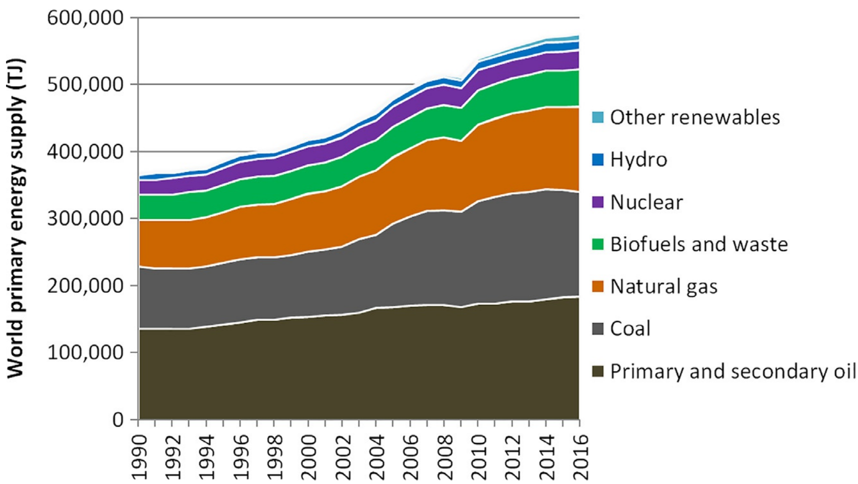

Fig. 5.1 World total primary energy supply (IEA, 2018).

increase. As shown in Fig. 5.1, the only notable reduction occurred around 2008, corresponding to the global recession causing decreased economic and industrial activity. As of 2016, global energy supply was approximately 580 PJ per year, of which 81% was provided by fossil fuels; this percentage was exactly the same in the year 1990.

As a result of the dominance of energy supply by fossil fuels, the sector is characterized by combustion technologies. Therefore it is perhaps not surprising that the most frequently addressed sustainability impacts of energy are climate change and direct emissions of local pollutants. For instance, on the former, the 2015 Paris Agreement has resulted in 141 countries submitted Nationally Determined Contributions with specific emission reduction targets for their energy sectors, at a total estimated cost of approximately USD 470 billion (The World Bank, 2016). On the latter issue of specific pollutants, various governments have enacted ever-tightening legislation to reduce emissions of key pollutants. The EU Industrial Emissions Directive (2012), for instance, contains limits on the emissions of particulate matter (PM), SO_2, NO_x, and CO from industrial combustion activities at various scales. A similarly restricted set of pollutants is also regulated in the transport sector in Europe, the United States, and many other regions via vehicle emissions standards.

The adoption of other, broader policies is also gaining traction in the energy sector including the EU's sectoral targets for 2020 and 2030 which cover emissions, energy efficiency, and the share of renewables in the energy mix (European Commission, 2014). Meanwhile corporate social responsibility (CSR) reporting has gained robustness and widespread adoption over the past decade, with examples including the Global Reporting Initiative which provides the most widely adopted framework for companies to track their progress on a variety of sustainability metrics aligned with the UN SDGs. As of mid-2018, it has received sustainability reporting data from 12,761 organizations, of which 1118 are in the energy sector (GRI, 2018).

Based on initiatives such as those before, there is clear incorporation of some sustainability principles into strategy, policy, and operations within the energy sector. However, it is also clear that governments and industry often base their actions on a limited range of issues, such as climate change and local pollutants, and that these issues are typically tackled at the level of individual power plants or vehicles rather than complete energy life cycles. The tackling of broader sustainability issues may occur more often via the adoption of general principles and attempts to increase transparency, rather than by concrete policies and actions. This is perhaps not surprising when considering the challenge: just as life cycle thinking is recognized as a prerequisite for environmental sustainability (Azapagic, 2004), the same holistic view is needed to ensure broader sustainability (Stamford and Azapagic, 2011). Combined with the need to address a wide range of issues, this means that robust decision-making for sustainable development requires that many criteria are accounted for simultaneously.

Fig. 5.2 provides a hypothetical example of a biogas-fired power plant. As illustrated, when the life cycle of power generation is considered holistic, a true sustainability assessment must consider a range of issues spanning climate change, air and water emissions, employment, safety, and others, all of which vary geographically and temporally throughout the life cycle.

These sustainability issues are particularly broad in the bio-sector due to its great variety of feedstocks and processing routes: agriculture, for instance, includes challenges associated with sustainable incomes for farmers, income distribution, gender equality, exposure to pesticides and other chemicals, and the results of those chemicals entering the environment, to name only a few. Therefore in recent years, attempts have been made to consolidate the key issues for the sector.

1.3 Sustainability issues and indicators

With specific reference to bioenergy, these attempts have resulted in the ISO 13065 standard on Sustainability Criteria for Bioenergy (ISO, 2013a) which provides a framework for sustainability assessment and reporting via a set of principles, criteria, and indicators.

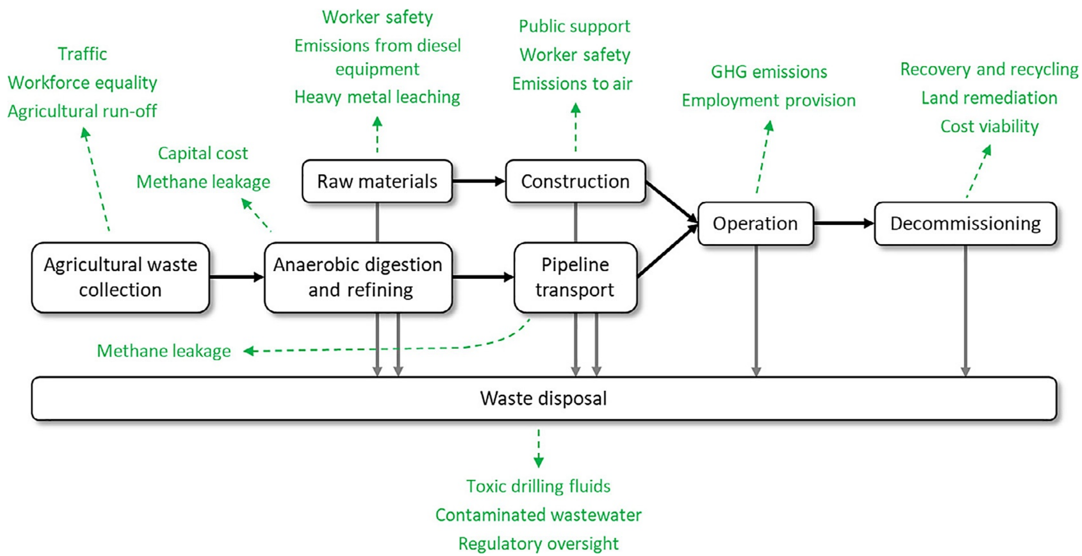

Fig. 5.2 Potential sustainability issues in the life cycle of a biogas-fired power plant.

Table 5.1 Sustainability goals identified in the "principles" of ISO 13065 (ISO, 2013a)

Environment	→ Reduce anthropogenic GHG emissions → Conserve and protect water resources → Protect soil quality and productivity → Promote good air quality → Promote positive and reduce negative impacts on biodiversity → Promote efficient use of energy resources → Promote responsible management of waste
Economy	→ Produce and trade bioenergy in an economically and financially viable way
Society	→ Respect human rights → Respect labor rights → Respect land use rights → Respect water use rights

The main issues identified in ISO 13065 are expressed as sustainability "principles" for the bioenergy sector, as outlined in Table 5.1. Within each of these principles, the standard describes metrics that should be assessed to ensure sustainability of the system under study. For instance, under the "reduce anthropogenic GHG emissions" principle, an operator who conforms to the standard must, at the very least, provide data on the GHG emissions and removals of one life cycle stage. Similarly, under "conserve and protect water resources" the operator must describe the procedures they have taken to identify potential impacts on water quantity.

However, while the standard provides a good basis for the consideration of sustainability in the bioenergy sector, it is targeted at businesses with the intention of improving their reporting procedures and transparency. Consequently, it is not necessarily suitable for the sustainability assessment of products and processes. For instance, like many of the standard's indicators, the water indicator mentioned before requires the identification of policies, procedures, and practices within a single business or project. It does not specify actual measurements associated with a technology, process, or product. Moreover, indirect impacts are generally omitted from the standard; in other words, it deals solely with impacts under the direct control of the operator. Consequently, it does not follow a life cycle approach (except, optionally, in the case of GHG emissions) and therefore cannot truly form a holistic representation of the sustainability impacts, accounting for all stakeholders up- and downstream of the production process.

Finally, it is often beneficial for sustainability assessment to involve as much quantification as possible in order to aid evaluation of the outcomes. Typical sustainability reporting approaches often rely heavily on identification and description rather than quantification.

To this end, the existing literature in the field of sustainability assessment has generated a large number of suggested metrics, otherwise referred to as *sustainability indicators*. The purpose of such indicators is to *simplify*, *quantify,* and *communicate* information across a range of relevant environmental, economic, and social issues. A variety of journal articles have reviewed the use of such indicators and assessment frameworks, such as Singh et al. (2012), and interested readers are directed to these works for a more detailed overview, as well as to the case studies discussed later. Specifically in the energy sector, examples of the use of sustainability indicators are found in Switzerland (Roth et al., 2009; Volkart et al., 2017), the United Kingdom (Stamford and Azapagic, 2012; Cooper et al., 2018b), Turkey (Atilgan and Azapagic, 2016), Mexico (Santoyo-Castelazo et al., 2014), China (Ren et al., 2015), and many other countries.

In many cases, and in line with the holistic principles of sustainability, these suggested sustainability assessment frameworks have taken a life cycle approach to their development and application. Consequently, they have often used, and encouraged the advancement of, related tools and techniques such as environmental life cycle assessment, economic life cycle costing, and social life cycle assessment. Together, tools such as these form the basis of life cycle sustainability assessment, an approach which is continuing to evolve and find application across varied disciplines.

2 Life cycle sustainability assessment

Just as sustainability has often been described as an integration of environmental, economic, and social issues, sustainability assessment typically adopts environmental, economic, and social techniques. However, a key element of life cycle sustainability assessment (LCSA) is its adoption of a life cycle approach, and consequently LCSA often shares many characteristics with the more established practice of LCA, which is described later. Therefore following the outline of LCA, this section describes methodologically consistent approaches to the other two pillars: namely, life cycle costing (LCC) and social life cycle assessment (SLCA).

2.1 Life cycle assessment

Life cycle assessment (LCA) is an environmental sustainability tool that applies life cycle thinking in order to assess the consequences of human activities. Broadly speaking, LCA involves:

1. Quantification of environmental burdens of a product, process, or activity via assessment of the energy and materials used and wastes released to the environment.
2. Quantification of environmental impacts (i.e., translating the earlier burdens into potential impacts).
3. Identification of opportunities for environmental improvements along the life cycle via the identification of "hot spots".

LCA is standardized via ISO 14040 and 14044 (ISO, 2006a,b), in which four key phases are identified, as outlined in Fig. 5.3.

LCA is a well-established technique with a wealth of existing literature demonstrating its use. As the focus of this chapter is on sustainability assessment, rather than LCA, interested readers are directed to other resources for

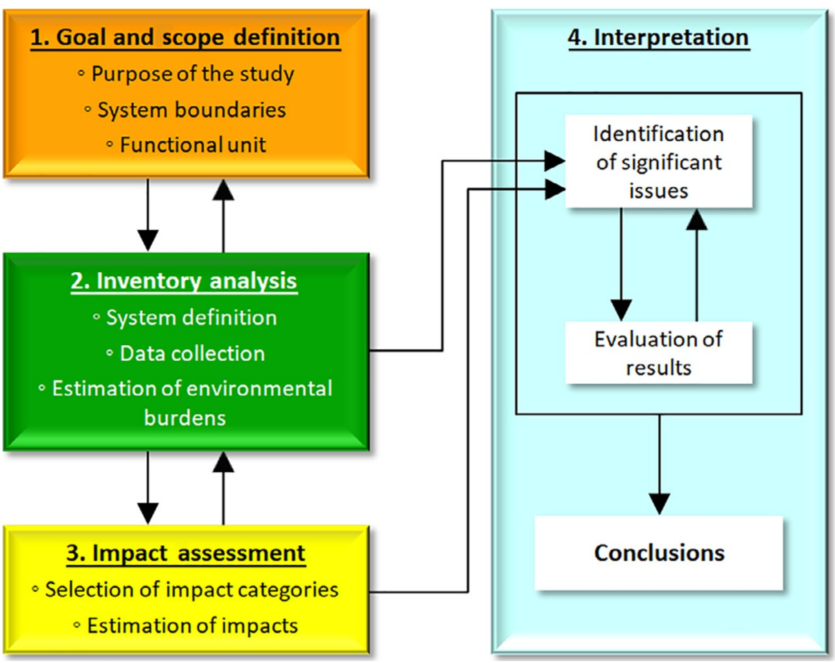

Fig. 5.3 The four phases of life cycle assessment as defined in ISO 14044.

more detail, such as Azapagic (2011). However, in this chapter we will consider to the phases "Goal and Scope Definition" and some elements of "Impact Assessment" as they provide a useful basis for the initial stages of a sustainability assessment and for the environmental indicators, respectively.

The Goal and Scope Definition phase of LCA has three major functions, all of which are equally applicable to LCSA:

1. Define the *purpose of the study*
   - While quite self-explanatory, it is nevertheless important to identify exactly what the purpose of the assessment is for. Is it to compare one product to a competitor? To benchmark the impacts of a new production process? Or to identify improvement opportunities in existing products or processes? Each of these may lead to different methodological choices. For instance, if comparison to other systems is required, the practitioner should be careful to define those other systems such that they provide a genuinely equivalent output. This is closely related to the concept of "functional unit", which is discussed as follows.

2. Define the *system boundaries*
   - The system boundary describes the physical scope of the assessment, that is, the parts of the life cycle which are or are not accounted for. This can take several forms depending on what is most appropriate for the system under assessment and the purpose of the study. The most common system boundaries for LCA are "cradle-to-gate" and "cradle-to-grave", as depicted in Fig. 5.4.

It should be noted that other system boundaries are possible and may be relevant depending on the system under study. For instance, processes seeking to close the material circularity loop by encompassing recycling and remanufacturing might take a "cradle-to-cradle" or "gate-to-gate" approach. Whatever the system boundary chosen, it is critical that it is transparent to readers in order to avoid misinterpretation.

3. Define the *functional unit*
   - The functional unit defines the function of the system and enables comparison of different systems on an equivalent basis. For instance, the function of beverage packaging is to store a certain amount of beverage, but this same function might be provided by 600 g of glass or 35 g of PET. Consequently, an appropriate functional unit might be "beverage packaging for 1 L of water", as opposed to "1 kg of bottles", in order to ensure that the comparator systems are functionally equivalent.

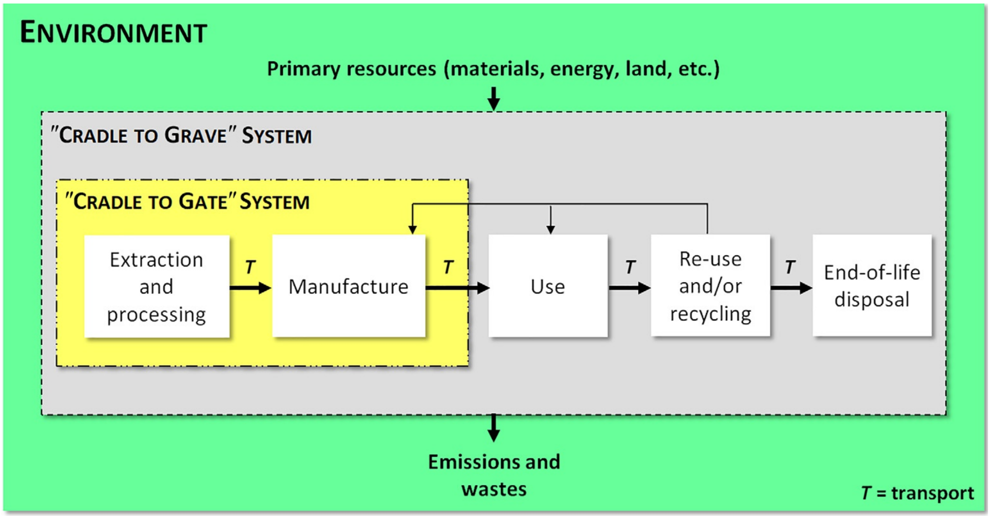

Fig. 5.4 Cradle-to-grave and cradle-to-gate system boundaries. *(Adapted from Azapagic, A., 2011. Chapter 3: assessing environmental sustainability: life cycle thinking and life cycle assessment. In: Azapagic, A., Perdan, S. (Eds.), Sustainable Development in Practice: Case Studies for Engineers and Scientists, second ed. Wiley-Blackwell, Chichester.)*

The Inventory Analysis phase of LCA is concerned with the collection of technical data, such as the mass and energy flows throughout the system's life cycle, and the estimation of flows to, and from, the environment. Typically this is achieved with some reliance on existing databases or literature to provide data for background systems (e.g., data on the environmental burdens associated with material inputs). In Europe, for instance, the Swiss nonprofit "Ecoinvent" database (Ecoinvent, 2018) is the most widely used and accepted life cycle inventory database. Many other similar databases exist, such as the US LCI Database managed by NREL (National Renewable Energy Laboratory, 2012).

The third phase of LCA, Impact Assessment, uses environmental impact coefficients, often referred to as characterization factors, to estimate the potential environmental impacts caused by the burdens identified during the Inventory Analysis phase. These impacts are calculated as follows:

$$E_k = \sum e_{k,j} \times B_j$$

where

E_k = Total environmental impact k

$e_{k,j}$ = Environmental impact coefficient describing the contribution of substance j to impact k

B_j = Environmental burden of substance j contributing to the impact k

The impact categories included in an LCA act as useful environmental indicators for potential adoption in LCSA. However, the exact list of indicators generated by an LCA depends on the impact assessment method adopted and a variety of options exists. Much of the existing literature has used the CML method (Guinée et al., 2002) but there is increasing consensus that this is now outdated. Alternatives include IMPACT2002+ (Jolliet et al., 2003), TRACI (Bare, 2002; Bare, 2011), ILCD (Wolf et al., 2012), and ReCiPe (Goedkoop et al., 2012; Huijbregts et al., 2016). Of these options, ReCiPe is often seen as the state of the art and there is some evidence to suggest that it is the most widely used method, although a plurality is evident in the community (Prox, 2018). Table 5.2 lists the LCA impact categories in the CML, ILCD, and ReCiPe methods.

In all cases, the characterization factors used to account for global warming/climate change are derived from the reports of the Intergovernmental Panel on Climate Change (e.g., IPCC, 2014). It should be noted that this indicator presents specific challenges when applied to systems with biological components, such as the food or bioenergy sectors, due to the need to

Table 5.2 Impact categories in three of the most widely adopted LCA impact assessment methods, arranged according to their equivalence

ReCiPe	ILCD	CML
Global warming	Climate change	Global warming
Terrestrial acidification	Acidification	Acidification
Freshwater eutrophication	Eutrophication	Eutrophication
Stratospheric ozone depletion	Ozone depletion	Ozone layer depletion
Tropospheric ozone formation (humans)	Photochemical ozone formation	Photochemical oxidant creation
Tropospheric ozone formation (ecosystems)		
Human toxicity (cancer)	Human toxicity	Human toxicity
Human toxicity (noncancer)		
Freshwater ecotoxicity	Ecotoxicity	Freshwater aquatic ecotoxicity
Marine ecotoxicity		Marine aquatic ecotoxicity
Terrestrial ecotoxicity		Terrestrial ecotoxicity
Mineral resources	Resource depletion	Depletion of abiotic resources, elements
Fossil resources		Depletion of abiotic resources, fossil fuels
Particulate matter	Respiratory inorganics/ particulate matter	
Ionizing radiation	Ionizing radiation	
Land use/transformation	Land use	
Water use		

account for the two particular problems of biogenic carbon and land use change. These are outlined in the following box.

Biogenic carbon

Biogenic carbon refers to carbon that is sequestered from the atmosphere during biomass growth and may be released back to the atmosphere later due to combustion of the biomass or decomposition (e.g., of food waste). Typically in LCA, and therefore LCSA, it is assumed that these two flows from, and into, the atmosphere are equal and cancel each other out. This is normally performed either by simply ignoring all biogenic carbon flows or by accounting for the negative flow during biomass cultivation followed by an equal, positive flow later in the life cycle. The latter, more explicit approach is supported by standards such as PAS 2050 (BSI, 2011) and ISO 14067 (ISO, 2013b)

However, this is a topic of ongoing debate, particularly for long-lived feedstocks such as wood from forestry, where rapid deforestation can indeed create a net contribution to the environment depending on the extent and speed of reforestation. The issue is also complicated by the potential production of methane, which is a more potent GHG than CO_2, under anaerobic conditions

Biogenic carbon accounting approaches are discussed in more depth in the literature, including carbon payback time, carbon discounting, and time-integrated accounting techniques (Cherubini et al., 2011; ICCT, 2014), but these are beyond the scope of this chapter

Land use change

Soil and vegetation contains large amounts of carbon that may be disturbed as a result of land use change (LUC). Part of this stored carbon is then oxidized and released to the atmosphere as CO_2. In the energy sector, LUC can occur as a direct result of conversion of grasslands or forests to biomass cultivation (direct LUC), or via the displacement of other crop cultivation activities to previously uncultivated land (indirect LUC). This is a major driver of climate change: from 1750 to 2011 it is estimated that LUC accounted for 32% (16%–55%) of cumulative anthropogenic CO_2 emissions (IPCC, 2014)

Direct LUC is often incorporated into LCA modeling, with the simplest method being that proposed by the European Commission (European Commission, 2010) which involves estimating the carbon stored in two carbon pools: carbon stock (i.e., living and dead organic biomass) and soil carbon stock. Indirect LUC is more difficult to account for due to the challenges of consequential analysis, that is, producing an accurate economic model to determine how much crop cultivation activity would be displaced by the additional demand for energy/feedstock crops. This topic is a subject of ongoing research and debate at the time of writing, the depth of which is beyond the scope of this chapter. Readers should note that the majority of many LCA impact assessment methods already include some form of land use accounting in the calculation of climate change impacts

2.2 Life cycle costing and associated techniques

Economic indicators in LCSA are normally based on conventional economic metrics, aligned with the life cycle perspective of LCSA where possible. A popular approach is that of life cycle costing (LCC). Like LCA, LCC follows the life cycle of a product or system within specified system boundaries. However, instead of tracking environmental flows, it includes only monetary inputs and outputs throughout the system.

Therefore LCC is quite well aligned with the ISO 14040/44 methodology for LCA discussed before. In fact, the methodology for LCC proposed by Swarr et al. (2011) is based on several years of work by the Society of Environmental Toxicology and Chemistry (SETAC), which has played a leading role in the development and standardization of LCA; thus LCC is directly intended to align with LCA.

Under this approach, the life cycle cost of a product or process is the sum of all economic costs incurred directly by actors in the life cycle. This can be expressed according to the key stages of the life cycle which, for an energy generating asset or piece of technology, might be represented simply as follows:

$$LCC = C_C + C_{FO} + C_{VO} + C_W + C_E + C_T$$

where

LCC = Total life cycle cost
C_C = Capital cost
C_{FO} = Fixed operating costs
C_{VO} = Variable operating costs
C_W = Cost of waste management, including recycling
C_E = Cost of end-of-life disposal (e.g., decommissioning)
C_T = Cost of transport between stages

In many cases it is possible to estimate LCC values directly from LCA modeling inputs: the latter will include flows of materials and energy throughout the life cycle, therefore attaching cost data to those flows results in a methodologically harmonized LCC model.

From LCC it is possible to derive other common economic indicators, such as value added (VA). VA is defined as the sale price minus the total costs of bought-in materials and/or services; the latter represented by LCC. Thus the VA accrued over the life cycle can be estimated as follows:

$$VA = SR - LCC$$

where

VA = Value added

SR = Sales revenue

LCC = Life cycle cost

Included in the guidelines for LCC is the possibility of discounted cash flow analysis, which can help to bring the LCC outputs in line with typical industrial expectations. In such cases, this can be achieved by multiplying each year's costs by a discount factor which can be expressed as follows:

$$P(T) = \frac{1}{(1+r)^T}$$

where

$P(T)$ = Discount factor

r = Discount rate (%)

T = Time units (e.g., years)

The advantage of discounted cash flow is that it accounts for (a) the *opportunity cost*[1] of investment and (b) the *risk*[2] to the investor. Thus discounted economic indicators are often more realistic in a market setting than undiscounted values.

In addition to LCC, LCSA can include other commonly used discounted economic metrics such as net present value. However, perhaps the most commonly used discounted cost indicator in the energy sector is levelized cost, which is used by many governments, including that of the United Kingdom (BEIS, 2016), and international bodies, including the International Energy Agency (IEA and NEA, 2015), to appraise energy projects. A typical discount rate used by such analyses would be 5%–10%. However, it is sometimes argues that such high discount rates are not commensurate with sustainability principles as they result in future costs being highly discounted and, in effect, ignored. In such cases one can argue that costs are being passed to future generations on the assumption that their effects will be minimal. This is particularly relevant for long-lived energy systems that include considerable end-of-life cost components, such as nuclear power or

[1] The opportunity cost represents the lost opportunity incurred by investing in a particular project. In other words it is the rate of return that an investor could expect if, instead of investing in the project in question, they invested elsewhere (i.e. in another project, the stock market, bonds or any other form of investment).

[2] Risk is critical to the viability of high capital cost, long-lived projects. An investor in a plant, for instance, must consider what would happen if the plant experienced serious unexpected problems, or if new legislation restricts its profitability.

carbon capture and storage. Therefore it is sometimes argued that discounting should be limited to <5% in a sustainability or social equity setting (see, for instance, HM Treasury, 2003). The counterpoint to this is that free market actors will choose to maximize return on investment and, by necessity, must consider the aforementioned opportunity cost and risk issues. Therefore it may be the case that low discount rates do not reflect viability in a market setting. Thus the practitioner must attempt to balance economic pragmatism against ideological principle.

Levelized cost is essentially calculated as the discounted LCC divided by the discounted energy output over the project lifespan, as follows:

$$LCOE = \frac{\sum_{t=1}^{T} (C_C + C_{FO} + C_{VO} + C_W + C_E + C_T)_t \times P_t}{\sum_{t=1}^{T} E_t \times P_t}$$

where

$LCOE$ = Levelized cost of energy
T = Lifetime of the power plant
C_C = Capital cost
C_{FO} = Fixed operating costs
C_{VO} = Variable operating costs
C_W = Cost of waste management, including recycling
C_E = Cost of end-of-life disposal (e.g., decommissioning)
C_T = Cost of transport between stages
P_t = Discount factor in year t

2.3 Social life cycle assessment

Of the three pillars of sustainability, the social pillar is the least developed in terms of LCSA tools and techniques. UNEP has published guidelines on social LCA (UNEP, 2009) which are specifically designed to align with ISO 14040/14044 in order to maintain consistency with LCA and LCC. However, operationalizing social LCA is somewhat challenging. This is partly due to the fact that environmental and economic assessment methods have their roots in technical, engineering-oriented disciplines that tend to pursue quantitative metrics, but it is also because many key social issues are simply difficult or impossible to measure. Sustainable Development Goal 3, for instance, outlines the aim of health and well-being, but in order to mathematically assess such a goal we would also need to define the parameters and thresholds of a "healthy" life and attempt to identify and enumerate a universally applicable causality of happiness.

Epistemologically, approaches such as LCA and LCC could be described as positivist in that they seek to identify purely objective knowledge. Conversely, many aspects of the social sciences lean toward interpretivism and, as such, are focused on ideas such as personal experience, perceived knowledge, and social construction: phenomena that are much harder to quantify. Consequently, much prior work in the social sciences is difficult to incorporate into a more numerical LCSA framework (Iofrida et al., 2018).

In addition to this overarching difficulty, the rapidly developing field of social LCA often lacks data, lacks standardized/accepted indicators (Kühnen and Hahn, 2017), and struggles to include "positive" impacts (as opposed to the traditionally "negative" impacts associated with environmental or economic indicators) (Ekener et al., 2018).

There is considerable ongoing research in this area with the aim of integrating social indicators and assessment techniques into LCSA. In general, there is some agreement that genuine success will require greater engagement with the social sciences and the use of qualitative, interpretivist approaches. However, as the topic is broad and evolving, interested readers are directed to dedicated works such as Kühnen and Hahn (2017) and Rafiaani et al. (2018).

Despite the challenges and caveats outlined before, it is possible to identify some commonly used indicators of social sustainability from the existing applied literature. In the gas sector, for instance, several studies have measured employment creation, health and safety issues, public perception, and public nuisance, the latter measured via particular metrics such as noise creation or traffic levels (Cooper et al., 2018a). Table 5.3 provides a selection of the issues and indicators that have been used in literature; note that these are illustrative but by no means exhaustive, and that the specific methodology associated with each indicator is discussed in length in the cited publications.

Readers may notice that some of the indicators in Table 5.3 are taken from LCA: human toxicity potential, depletion of abiotic resources (elements), and depletion of abiotic resources (fossil fuels) are impact categories from the CML methodology (see Table 5.2). This is because the environmental issues addressed by LCA have direct social consequences, meaning that some LCA impacts may be better classified as social issues. In fact, all three pillars of sustainability have strong overlaps meaning that the choice of classification is somewhat subjective.

Table 5.3 Illustrative social sustainability indicators for the energy sector

Issue	Indicator	Unit
Provision of employment	Direct employment	Person-years/GWh
	Life cycle employment	Person-years/GWh
	Local employment	%
	Gender equality	Ordinal scale
Human health impacts	Worker injuries	Injuries/TWh
	Human toxicity potential (excluding radiation)	kg 1,4 DCB[a] eq./kWh
	Human health impacts from radiation (workers and population)	DALY[b]/GWh
Large accident risk	Fatalities due to large accidents	No. of fatalities/GWh
Local community impacts	Spending on local suppliers relative to total annual spending	%
	Direct investment in local community as proportion of total annual profits	%
	Noise	dB
	Traffic increase	%
Public perception	Public support	%, measured by survey
	Media impact	Relative presence on social media
Human rights and corruption	Involvement of countries in the life cycle with known corruption problems (based on Transparency International Corruption Perceptions Index)	Ordinal scale
Energy security	Diversity of fuel supply mix	Ordinal scale
	Fuel storage capabilities (energy density)	GJ/m^3
Intergenerational equity	Depletion of abiotic resources (elements)	kg Sb eq./kWh
	Depletion of abiotic resources (fossil fuels)	MJ/kWh
	Volume of waste requiring long-term storage	m^3/kWh

[a] 1,4-Dichlorobenzene.
[b] Disability adjusted life year.
(Based on Stamford, L., Azapagic, A., 2012. Life cycle sustainability assessment of electricity options for the UK. Int. J. Energy Res. 36(14), 1263–1290; Cooper, J., Stamford, L., Azapagic, A., 2018a. Social sustainability assessment of shale gas in the UK. Sustain. Prod. Consum. 14, 1–20.)

It is also of note that some of the social indicators in Table 5.3 do not span the entire life cycle of the system under assessment either because they are only applicable to one stage or because data are lacking. In some cases it is possible to estimate whole life cycle impacts using an "input-output" approach. This involves using national statistics or corporate data on, for instance, employment levels and injury rates in various sectors and dividing those figures by the annual output of those sectors to estimate the number of employees or injuries per unit of product. The same can be done for each sector or subsector in the life cycle to arrive at a final estimate per functional unit. The difficulty of such an approach is that it is time consuming and requires sufficient resolution in the sectoral statistics to avoid producing inaccurate estimates. Examples of such approaches are found in Stamford and Azapagic (2012) and Atilgan and Azapagic (2016).

The generation of extensive databases for use in social LCA is gaining momentum. Often such efforts are based on input-output approaches as outlined before, including recent resources such as the Social Hotspots Database (SHDB, 2018).

2.4 Allocation of impacts

Industrial processes—and particularly those that involve biological systems—are often multioutput systems with various coproducts. These include, for instance, crops which yield edible and nonedible components, or animal systems such as cattle that yield meat, hide, milk, and fertilizer (manure). Therefore in the bioenergy sector there is a particular need to allocate impacts between the fuel/energy output and the system's coproducts.

Allocation has been explored widely in LCA and often proves controversial. According to ISO 14040/14044, allocation should be avoided where possible by either subdividing the system under study or by *system expansion*. The latter is often seen to be the preferable option and, in its simplest form, involves crediting the system with the avoided burdens incurred by the coproducts. This is sometimes referred to as the "substitution" or "avoided burden" approach.

For instance, an anaerobic digester produces biomethane as its primary coproduct and a nitrogen-rich digestate as a secondary coproduct which can be used as a fertilizer. Under system expansion, we might determine the environmental impacts of producing an equivalent amount of chemical fertilizer and subtract those impacts from the overall system. Such an approach can be applied in LCSA by subtracting the economic and social impacts of the fertilizer as well as the environmental impacts.

If system expansion cannot be performed—for instance, if the coproduct has no equivalent—then *physical* or *economic allocation* can be used. In such cases the impacts of the system are allocated to each coproduct based on mass, energy content, or economic value.

System expansion and economic allocation have disadvantages, primarily in their lack of consequentiality: in other words, the presence of a new process or industry with large volumes of a particular coproduct will have knock-on effects on the market, and these effects in turn may render the chosen substitution or economic allocation obsolete. In contrast, the physical allocation approaches, such as mass or energy content allocation, do not face this problem. They may, however, fail to value the utility of the different coproducts.

The selection of an allocation approach has been shown to dramatically influence the overall results of LCAs in the bioenergy sector (e.g., Stephenson et al., 2010). As a result, it is advisable to explore the importance of allocation choices by applying different types of allocation in a sensitivity analysis.

2.5 Benefits, limitations, and weaknesses of LCSA

LCSA yields a large quantity of data due to the need to perform a holistic assessment across the three pillars of sustainability. It is not uncommon for sustainability frameworks to include 40 or more indicators (see, e.g., Roth et al., 2009; Stamford and Azapagic, 2011). Consequently, in cases where an LCSA is being used to choose between alternatives, interpreting the results is far from trivial. Often the best option is not clear: perhaps trade-offs need to be made between indicators, or value judgment is required, or uncertainty is high.

Uncertainty within LCSA can be explored using *scenarios* or *sensitivity analysis*. In the former, the analysis is rerun using different underlying assumptions: an approach explored further in Section 3. The challenge in that case is ensuring that the selected scenarios are truly reflective of the uncertainty space. In other words, scenario analysis should contain a sufficient breadth of scenarios to cover the entire likely range of possibilities. In the case of sensitivity analysis, one key parameter is varied at a time to explore the effect on the overall results of the LCSA. The chosen parameters should either:

1. Parameters that play a leading role in determining the impacts of the system (e.g., the yield, capacity factor, choice of allocation approach, use case of the product, etc.); or
2. Parameters that are highly uncertain due to poor input data or reliance on assumption (e.g., lifespan of the system, costs or environmental burdens of a major input material, etc.).

However, even accounting for the problem of uncertainty, sustainability is often characterized by the existence of so-called wicked problems (Rittel and Webber, 1973; Peterson, 2009; Azapagic and Perdan, 2014): that is, broadly speaking, that they have ill-defined goals or end points, cannot be reduced to "true-or-false" or "right-or-wrong" status, are defined according to the values of different stakeholders, have a very large number of potential solutions, and are fraught with contradictory or incomplete information. These challenges might leave those wishing to make informed decisions at risk of so-called analysis paralysis: the failure to complete their assessment and make a final decision due to overwhelming complexity. Without some form of robust decision analysis, policy decisions that could be critically important might be avoided, stalled, or replaced with suboptimal solutions that disregard the available information.

One solution to this problem is *multicriteria decision analysis* (MCDA), also referred to as multicriteria decision-making. As a general concept, MCDA aims to support complex decision-making situations with multiple, potentially conflicting objectives which are ascribed differing value by different stakeholders. The field consists of many different schools of thought and techniques, with intense debate between those different schools. However, regardless of the specific methodology chosen, the overarching aims of MCDA are to

- provide a structured, numerical and transparent way of aiding decision-making
- increase understanding of the decision-maker's values and those of others, as well as providing insight into how those values affect the decision
- provide insight into the most influential parameters of the decision, potentially leading to targets or critical trigger points.

MCDA is a very broad field with a variety of methodological approaches, the extent of which is too great to be encompassed here. For such purposes, readers are directed to dedicated publications (such as Azapagic and Perdan, 2005a,b; Department for Communities and Local Government, 2009; Cinelli et al., 2014).

3 Application of life cycle sustainability assessment: Illustrative case studies

The following sections demonstrate the application of LCSA via two illustrative case studies. In both cases, multiple sustainability indicators are used to provide information for decision makers in the energy sector.

3.1 Large-scale biomass combustion

This case study is based on Stamford and Azapagic (2012, 2014). It is set within the context of national energy policy in the United Kingdom, although it is largely applicable to other countries including those of the EU. It considers 12 techno-economic, 10 environmental, and 14 social indicators to assess the first attempts at large-scale biomass-fired electricity generation in UK power plants.

3.1.1 Context

In recent years, increasingly stringent legislation in Europe has placed tightening emissions limits on large combustion plants. These measures have particularly affected coal-fired power generation due to its high emission values for CO_2 and local pollutants. Examples include the Large Combustion Plant Directive (European Commission, 2001) and its successor, the Industrial Emissions Directive (2012), both of which limit the permissible emission of particulate matter (PM), SO_2, NO_x, and CO. When combined with other measures such as carbon emissions trading schemes and taxes, this has led to the early closure of many coal plants and, in some cases, the conversion of those plants to partial or 100% biomass combustion.

Examples in the United Kingdom include RWE npower's Tilbury plant (750 MW), which converted to 100% wood pellets in 2011 (and subsequently closed in 2013), followed by E.On's Ironbridge (600 MW) in 2013. At the time of writing, Drax (4 GW) has also completed the conversion of two-thirds of its capacity to biomass and has proposed full conversion in the future (Selby Times, 2012), as has Eggborough (1.96 GW) (Webb, 2012).

These major, large-scale biomass projects are almost invariably reliant on wood pellets imported from North America. For example, during its operation, Tilbury power station imported around 60% of its pellets from British Columbia (Canada) and 30% from Georgia (United States), the latter using RWE's own dedicated wood pellet production facility (Staves, 2011). Drax also owns major wood palletization assets in North America. In 2014 Drax alone consumed 60% of all wood pellet exports from the United States and is the single largest consumer of wood pellets on the planet (US Energy Information Administration, 2015).

Projects like these, with large-scale local pollutant emission and long-distance fuel transport requirements, prompt questions about their sustainability. Consequently, this section explores these questions based on the assumption of wood pellet imports from Canada and the United States to the United Kingdom. It also considers the use of miscanthus, a popular energy crop which can also be pelletized, is able to grow quickly, and can thrive on land that is suboptimal for food production.

3.1.2 Goal and scope definition

The purpose of this case study is to evaluate the sustainability of large-scale biomass-fired electricity generation in a UK setting. The assessment takes a cradle-to-gate approach, as outlined by the system boundary in Fig. 5.5. The functional unit is 1 kWh of electricity generated at the power plant.

3.1.3 Inventory analysis

This section outlines the main assumptions and data sources for the case study.

Cultivation and processing of wood pellets

Life cycle inventory data on wood cultivation and processing into pellets are taken from Ecoinvent v2.2 (Ecoinvent Centre, 2010), adapted to use the appropriate national electricity mix. As in Ecoinvent, 28% of the wood is assumed to be beech, 72% spruce (based on current consumption of each species). Pellets are produced from the residual wood that is a by-product of wood planing, with the main product being sawn timber. Therefore allocation is necessary for the impacts of wood cultivation, felling, and planning; this has been carried using economic allocation because there are no equivalent coproducts, meaning system expansion and substitution is not possible.

However, as global wood pellet demand is increasing greatly (Cocchi et al., 2011; Pöyry, 2011), a move toward dedicated production of wood for pellets is anticipated. Thus the study also considers the production of pellets from wood felled specifically for that purpose. In this case, debarking, chipping, drying, and pelletization take place in the same facility [as is the

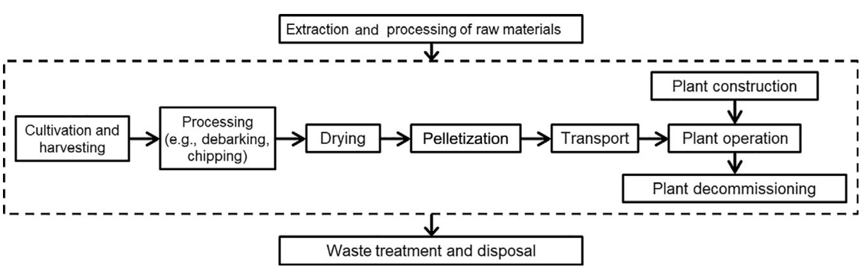

Fig. 5.5 System boundary for the LCSA of electricity generation from biomass.

case at RWE's Georgia pelletization plant, one of the largest in the world (Georgia Biomass, 2011)].

In all cases, the resulting pellets have a moisture content (MC) of 10%, a dried mass of 650 kg/m^3, and a net energy density of 12,164 MJ/m^3. These figures are in good agreement with those given by other sources, such as the UK Forestry Commission's Biomass Energy Centre (Biomass Energy Centre, 2012).

Cultivation and processing of miscanthus

The use of miscanthus as an energy crop is growing in popularity worldwide. As is the case for wood, pellets have been considered due to their homogeneity and high energy density making them favorable from the perspective of large power plant owners.

As a relatively new energy crop, long-term fertilizer and herbicide requirements are uncertain (and ultimately depend on site-specific conditions). Some trial sites have not required fertilizer, whereas others have, therefore this study assumes a 50:50 split between cultivation with and without fertilizers. Specific requirements are taken from Gilbert et al. (2011). In both cases glyphosate is used as a weed killer. Cultivation occurs over a period of 23 years, 21 of which are harvested with an average yield of 14 oven-dry tonnes/ha, which is thought to be typical based on UK experience (Gilbert et al., 2011).

Late harvesting is assumed (March/April as opposed to December) in order to minimize moisture content. Typical MC at late harvest is below 20% (Hopwood, 2010), and the MC of the pellets is assumed to be 10%. Any drying of miscanthus is not accounted for: it is assumed that air drying will be sufficient. The miscanthus is baled, chipped, and fed into a pelletizer to form a final product with a net calorific value of 15.65 MJ/kg (gross = 17.2 MJ/kg).

Fuel transportation

After production in the United States/Canada, wood pellets are assumed to be transported 100 km by rail to the nearest port, then shipped to the United Kingdom on a Supramax-class transoceanic freight ship [capacity 51,500 t, as specified in Ecoinvent (Ecoinvent Centre, 2010)]. Distances have been estimated as straight-line shipping routes using a mapping tool (Free Map Tools, 2012) and are therefore likely to be slightly underestimated. The routes are described in Table 5.4. An average of the biomass trade routes above is assumed, that is, one-third of the fuel comes from British Columbia, one-third Nova Scotia, and one-third Georgia. Following arrival in the

Table 5.4 Origins and shipping routes of North American wood pellets analyzed in this study

Origin	Destination	Route	Approximate distance (km)
Vancouver, British Columbia, Canada	London	South along USA West Coast → through Panama Canal → direct across Atlantic Ocean	16,500
Nova Scotia, Canada	London	Direct across Atlantic Ocean	5000
Savannah, Georgia, United States	London	Direct across Atlantic Ocean	7000

United Kingdom, the pellets are transported 10 km to the power plant by a Euro 5-compliant lorry.

While wood pellets are predominantly imported to the United Kingdom, miscanthus is more often grown domestically; thus it is assumed that 75% of miscanthus pellet supply is UK sourced. Domestic miscanthus pellets are transported from the pelletization facility 100 km to the power plant by a Euro 5-compliant lorry.

The remaining 25% of supply is sourced evenly from the United States and Canada using the same transport arrangements as that of wood pellets.

Power plant operation

Combustion takes place in a large 500 MW power plant, the construction of which is based on coal plants in Ecoinvent (Ecoinvent Centre, 2010). The efficiency of the plant is 35%, based on the figure of 35.3% achieved by RWE npower's Tilbury power plant after conversion to biomass (Staves, 2011).

The recent average capacity factor of plant-biomass power stations in the United Kingdom is 47.1% [5 year average, 2007–11 (DECC, 2013)]. Therefore a capacity factor of 50% is assumed. In the base case, the plant is assumed to have electrostatic precipitators capturing 99.95% of particulate matter, low-NO_x burners reducing NO_x by 25%, and flue gas desulphurization removing 90% of SO_2. These pollution controls are based on the technologies implemented at Drax at the time of its first conversion to biomass combustion.

In the case of wood pellets, direct emissions were estimated using the GEMIS v4.71 database (Öko-Institut, 2012). These were integrated with the other background processes from the Ecoinvent v2.2 database (Ecoinvent Centre, 2010) using GaBi 4.4 life cycle assessment software (PE International, 2008). Emissions of heavy metals were calculated using data on the metal content of virgin wood and the proportions of each metal emitted to air, as reported by Krook et al. (2004).

In the case of miscanthus pellets, background data are also from Ecoinvent v2.2 (Ecoinvent Centre, 2010), while miscanthus composition (including heavy metals) was taken from Obernberger et al. (2006) allowing direct combustion emissions to be calculated in GEMIS. Aerial emissions of heavy metals were calculated based on Krook et al. (2004) data on the proportion of metals emitted to air. All data were combined in GaBi 4.4 (PE International, 2008). It should be noted that the elemental composition of miscanthus is greatly influenced by soil composition, time of harvest, and the amount of leaf matter harvested. Direct emissions are therefore likely to be less accurate than those from wood pellets.

Waste disposal

As in Ecoinvent, 50% of the wood ash from combustion is assumed to be disposed of in sanitary landfill, 25% in municipal incineration, and 25% via spreading on agricultural land.

3.1.4 Indicators and impact assessment

The sustainability indicators considered in this assessment are presented in Table 5.5. The quantification of these metrics follows the approaches discussed in Section 2, with extra detail available in Stamford and Azapagic (2014).

3.1.5 Interpretation

The results of the case study are shown in… including data on comparator technologies which are taken from Stamford and Azapagic (2012). Due to the volume of results, each pillar of sustainability is assessed in turn.

The techno-economic indicators (Fig. 5.6) show biomass to compete well against the other technologies, benefitting from a potentially infinite fuel reserve (assuming appropriate management of the biomass resource), the best dispatchability of all options due to its relatively low capital cost, and levelized costs that are slightly higher than gas power but lower than all other options. However, readers should note that this assessment relies

Table 5.5 Sustainability indicators for the assessment of large-scale biomass power and its alternatives

	Sustainability Issue	Indicator	Unit
Techno-economic	Operability	Capacity factor (power output as a percentage of the maximum possible output)	Percentage (%)
		Availability factor (percentage of time a plant is available to produce electricity)	Percentage (%)
		Technical nondispatchability (ramp-up rate, ramp-down rate, minimum up time, minimum down time)	Summed rank
		Economic nondispatchability (ratio of capital cost to total levelized generation cost)	Percentage (%)
		Lifetime of global fuel reserves at current extraction rates	Years
	Technological lock-in resistance	Ratio of plant flexibility (ability to provide trigeneration, negative GWP and/or thermal/thermochemical H_2 production) and operational lifetime	$Years^{-1}$
	Immediacy	Time to plant start-up from start of construction	Months
	Levelized cost of generation	Capital costs	£/MWh
		Operation and maintenance costs	£/MWh
		Fuel costs	£/MWh
		Total levelized cost	£/MWh
	Cost variability	Fuel price sensitivity (ratio of fuel cost to total levelized generation cost)	Percentage (%)

Table 5.5 Sustainability indicators for the assessment of large-scale biomass power and its alternatives—cont'd

	Sustainability Issue	Indicator	Unit
Environmental	Material recyclability	Recyclability of input materials	Percentage (%)
	Water ecotoxicity	Freshwater ecotoxicity potential	kg 1,4 DCB[a] eq./kWh
		Marine ecotoxicity potential	kg 1,4 DCB[a] eq./kWh
	Global warming	Global warming potential (GHG emissions)	kg CO_2 eq./kWh
	Ozone layer depletion	Ozone depletion potential (CFC and halogenated HC emissions)	kg CFC-11 eq./kWh
	Acidification	Acidification potential (SO_2, NO_x, HCl, and NH_3 emissions)	kg SO_2 eq./kWh
	Eutrophication	Eutrophication potential (N, NO_x, NH_4^+, PO_4^{3-}, etc.)	kg PO_4^{3-} eq./kWh
	Photochemical smog	Photochemical smog creation potential (VOCs and NO_x)	kg C_2H_4 eq./kWh
	Land use and quality	Land occupation (area occupied over time)	m^2yr/kWh
		Terrestrial ecotoxicity potential	kg 1,4 DCB[a] eq./kWh
Social	Provision of employment	Direct employment	Person-years/TWh
		Total employment (direct + indirect)	Person-years/TWh
	Human health impacts	Worker injuries	No. of injuries/TWh
		Human toxicity potential (excluding radiation)	kg 1,4 DCB[a] eq./kWh
		Total human health impacts from radiation (workers and population)	DALY[b]/kWh

Continued

Table 5.5 Sustainability indicators for the assessment of large-scale biomass power and its alternatives—cont'd

	Sustainability Issue	Indicator	Unit
	Large accident risk	Fatalities due to large accidents	No. of fatalities/ PWh
	Energy security	Amount of imported fossil fuel potentially avoided	toe/kWh
		Diversity of fuel supply mix	Score (0–1)
		Fuel storage capabilities (energy density)	GJ/m^3
	Nuclear proliferation	Use of nonenriched uranium in a reactor capable of online refueling; use of reprocessing; requirement for enriched uranium	Score (0–3)
	Intergenerational equity	Use of abiotic resources (elements)	kg Sb eq./ kWh
		Use of abiotic resources (fossil fuels)	MJ/kWh
		Volume of radioactive waste to be stored	m^3/TWh
		Volume of liquid CO_2 to be stored	m^3/TWh

[a] 1,4-Dichlorobenzene.
[b] Disability-adjusted life years.
(Based on Stamford, L., Azapagic, A., 2014. Life cycle sustainability assessment of UK electricity scenarios to 2070. Energy Sustain. Dev. 23, 194–211.)

on cost data from approximately 2013, since which the industry has seen extreme cost reduction for wind and solar PV. For instance, the last two "contract-for-difference" auctions in the UK energy market have suggested total costs as low as £50/MWh for solar PV and £57.50/MWh for offshore wind (DECC, 2015; BEIS, 2017). Against these costs, the estimate presented here for biomass (£77/MWh) is less competitive, although biomass combustion does retain the ability to quickly ramp output up and down, which is not possible with wind or PV.

The issue before highlights a common problem in LCSA: rapidly developing technologies require regularly updated LCA, LCC, and SLCA

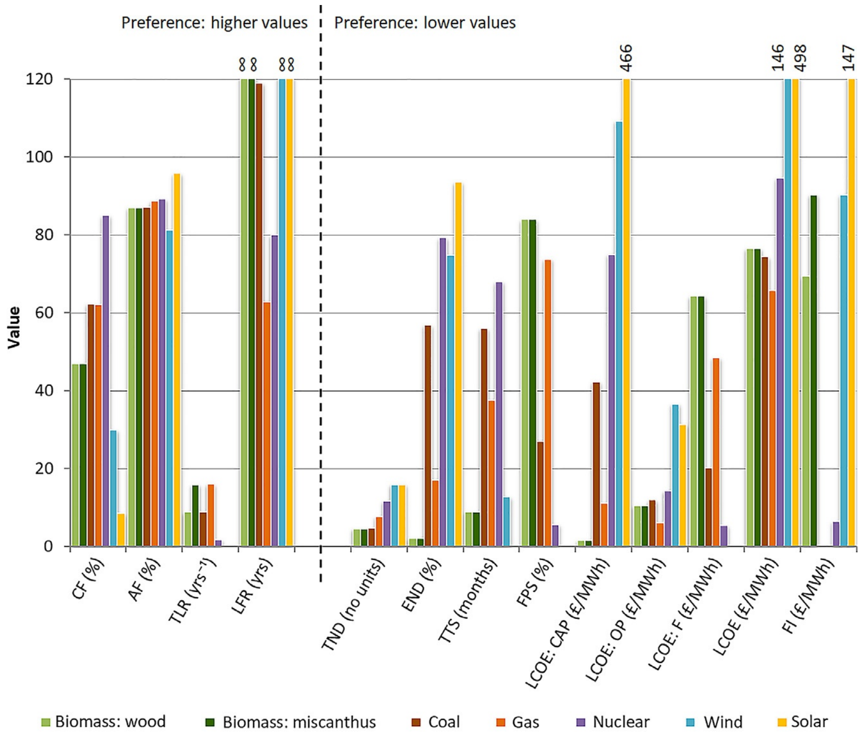

Fig. 5.6 Techno-economic sustainability indicator results [*CF*, capacity factor; *AF*, availability factor; *TLR*, technological lock-in resistance; *LFR*, lifetime of fuel reserves; *TND*, technical nondispatchability; *END*, economic nondispatchability; *TTS*, time to plant start-up; *FPS*, fuel price sensitivity; *LCOE*, levelized cost of electricity (capital, operation and maintenance, fuel, and total); *FI*, financial incentives].

models. The ideal solution to this would be a "live" model that is periodically updated as technology improves, but in practice the labor and resource requirements of this arrangement may be excessive. Therefore we must interpret the results of LCSA with some caution, and preferably conduct sensitivity analyses and scenario analyses where possible. The latter two options are explored further later.

As shown in Fig. 5.7, biomass is estimated to be a mid-ranking option for a large number of indicators. Its global warming potential is estimated at 112–123 g CO_2 eq./kWh, demonstrating that, despite the need for cultivation, pelletization, and long distance fuel transport, it still retains a carbon footprint considerably lower than either gas (380 g) or coal (1070 g). However, biomass is typically not competitive with wind or nuclear power: both options are superior to biomass for nine out of 10 indicators. In the case of

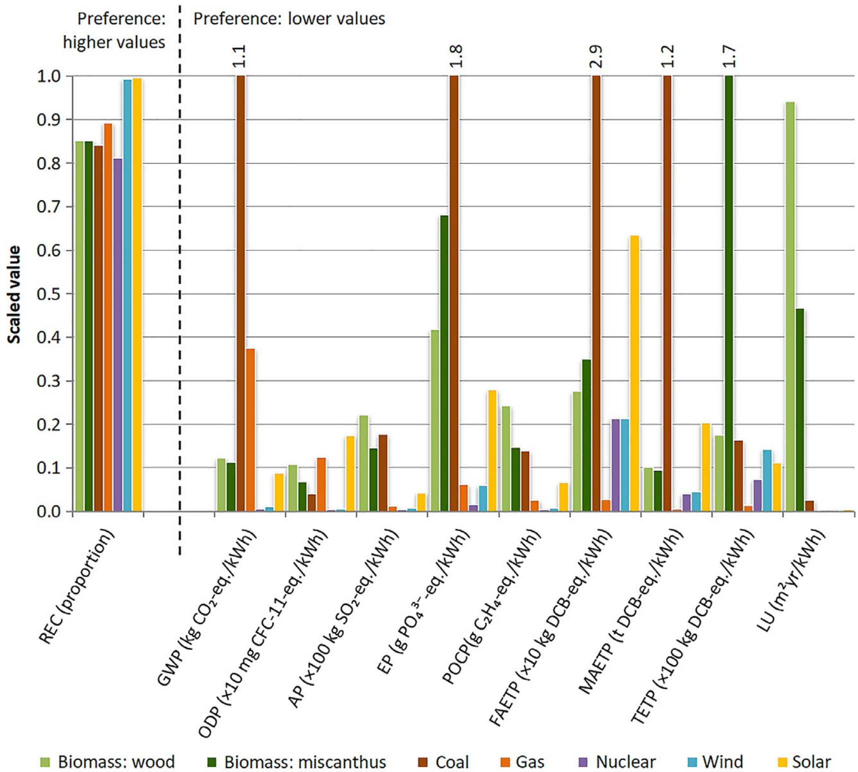

Fig. 5.7 Environmental sustainability indicator results (*REC*, recyclability; *GWP*, global warming potential; *ODP*, ozone layer depletion potential; *AP*, acidification potential; *EP*, eutrophication potential; *POCP*, photochemical oxidant creation potential; *FAETP*, freshwater ecotoxicity potential; *MAETP*, marine ecotoxicity potential; *TETP*, terrestrial ecotoxicity potential; *LU*, land use).

land use, both biomass options are the worst available due to their need for forestry or agricultural land. In fact, the land usage is so great that providing approximately 20% of current UK demand via miscanthus would require about 2.5 million hectares for crop cultivation. For context, the entire land area of England is 13 million hectares.

For terrestrial ecotoxicity, miscanthus is the worst option of all by an order of magnitude. This is because the most common disposal option for biomass ash is spreading on farmland as a low-grade fertilizer and liming agent, which has the unintended side effect of increasing heavy metal contamination in agricultural soils. In this study, 25% of ash is assumed to be disposed of by agricultural spreading. However, note that the metal content of miscanthus, and therefore the resulting TETP of miscanthus power, is highly variable depending on the cultivation site.

In terms of social indicators (Fig. 5.8), the life cycle of biomass power generates more employment than all other technologies with the exception of solar PV and potentially wind power. For the majority of the indicators it is a mid-ranking option but seems particularly beneficial in terms of intergenerational equity: as shown in the figure, it is typically the case that renewable energy options incur high abiotic depletion of elements due to their large requirements for metal per unit energy generated. In contrast, fossil fuel options cause orders of magnitude less depletion of elements, but obviously far higher depletion of fossil fuels. In contrast, biomass shows depletion of elements that is approximately 90%–99% lower than either wind or solar PV, but fossil fuel depletion that is approximately 85% lower than coal power over the whole life cycle.

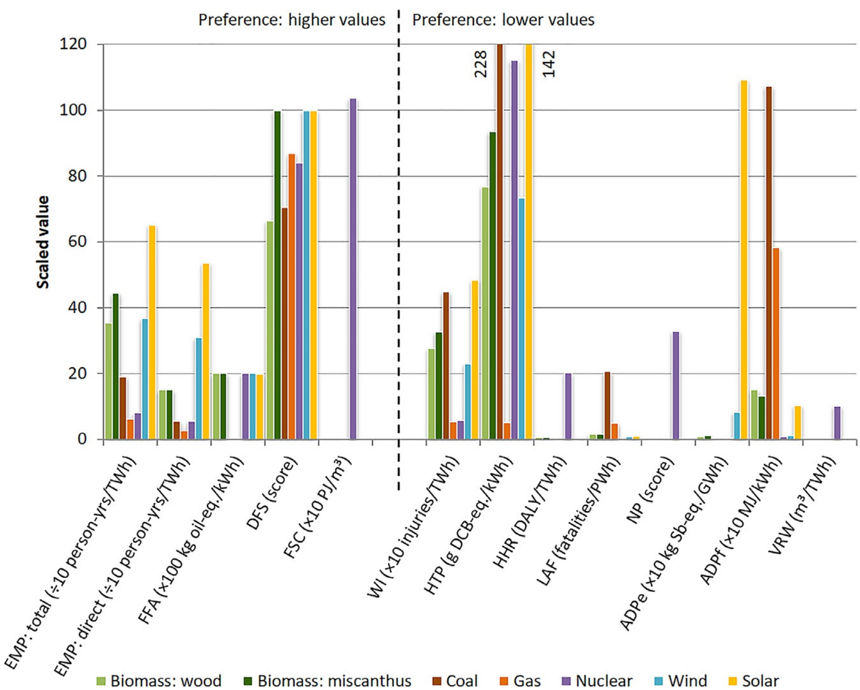

Fig. 5.8 Social sustainability indicator results (*EMP*, employment; *FFA*, fossil fuel avoided; *DFS*, diversity of fuel supply; *FSC*, fuel storage capabilities; *WI*, worker injuries; *HTP*, human toxicity potential; *HHR*, human health impacts from radiation; *LAF*, large accident fatalities; *NP*, nuclear proliferation; *ADPe*, abiotic depletion of elements; *ADPf*, abiotic depletion of fossil fuels; *VRW*, volume of radioactive waste).

Given the fact that a large fraction of the environmental and human health impacts of biomass are traceable to the direct emissions during operation, it is useful to explore the effects of differing pollution control measures via *sensitivity analyses*. Fig. 5.9 shows how the key affected indicators respond to

– The absence of flue gas desulphurization (FGD).
– The addition of selective catalytic reduction (SCR), reducing NOx emissions by 85%.
– An improvement in overall plant efficiency from 35% to 40%.

The figure shows all impacts relative to the base case (FGD, ESP, low-NOx burners, and an efficiency of 35%). As shown in the figure, when FGD is not implemented, the overall life cycle acidification and photochemical smog impacts of biomass power increase by 12%–134%. This difference is particularly notable in the case of miscanthus due to its potentially higher sulfur content per unit energy content. As already observed from Fig. 5.7, biomass already performs quite poorly for these impact categories, therefore this suggests that legislation should be enacted to ensure FGD is fitted.

The use of SCR reduces the impacts shown in Fig. 5.9 by an average of 6% with the most dramatic effect seen in the eutrophication potential of wood-fired power plants (a reduction of 18% over the life cycle).

Thus it could be concluded that the most important considerations are the use of FGD and the overall efficiency of the plant. However, since the best case only reduces impacts by an average of 18%, the rest of the life cycle requires attention for biomass to convincingly compete against wind or nuclear power. Eutrophication and ozone layer depletion, for instance, are mostly attributable to the importation of pellets by sea, therefore the transport stage is the key area for improvement.

These insights, based on life cycle thinking, can help to scrutinize policy. For instance, the Industrial Emissions Directive includes NO_x emission limits of $200\,mg/m^3$ which necessitates selective catalytic reduction (SCR) (Directive 2010/75/EU, 2012). However, while the earlier analysis shows that SCR is beneficial, it suggests that policy effort might best be deployed elsewhere.

3.2 Future electricity scenarios for the United Kingdom

As demonstrated before, LCSA can provide useful insights for present-day or near-term decision-making in the energy arena. However, energy strategy often requires longer term thinking. Future scenario analysis is a useful tool

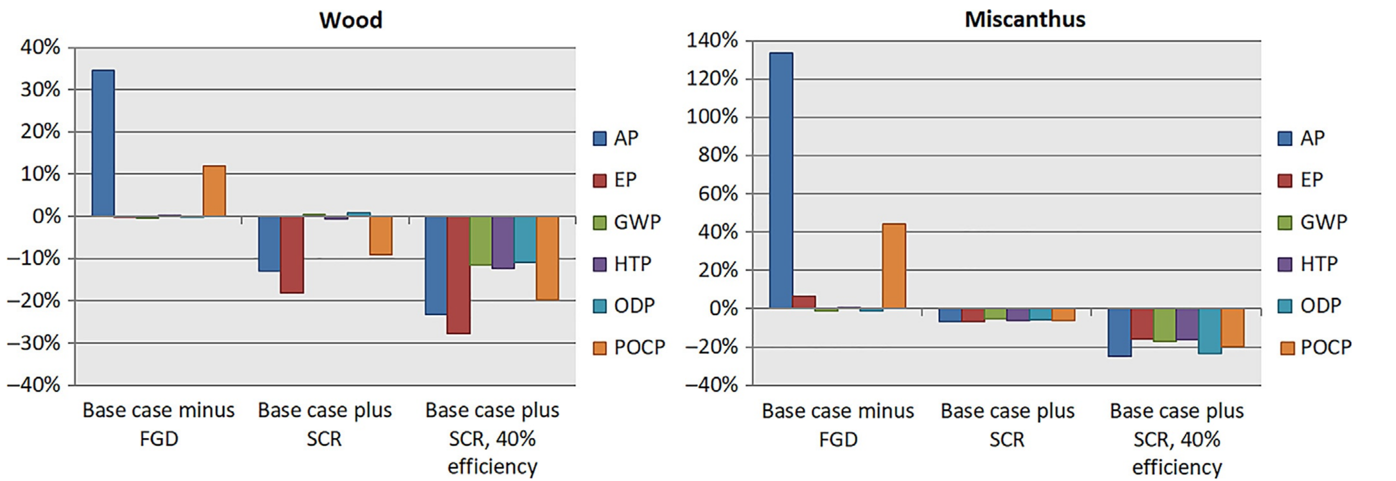

Fig. 5.9 Sensitivity analysis of pollution control measures for wood and miscanthus biomass-fires electricity generation (*FGD*, flue gas desulphurization; *SCR*, selective catalytic reduction; for other nomenclature see Figs. 5.7 and 5.8).

that is regularly used for strategy development. While conventional scenario analysis has focused on economic criteria, it is possible to apply LCSA in the same manner.

The following case study is based on Stamford and Azapagic (2014). It uses the same technologies and data sources as Section 3.1 but develops the analysis further to explore the sustainability impacts of the entire UK electricity mix up to the year 2070. It also serves as an example of the implementation of learning curves, parameter variation, and simple decision analysis.

3.2.1 Goal and scope definition

Achieving the UK's legally binding target of reducing GHG emissions by 80% by 2050 (compared to 1990 levels) will require a complete decarbonization of the UK electricity mix in that time period (UKERC, 2009). Given the long lives of several electricity generation assets, such as nuclear plants with a 60-year design life, it is also true that decisions made today will be with us beyond 2050.

Consequently, the goal of this case study is to explore the sustainability of potential future electricity mixes considering a range of generation technologies. The functional unit of the analysis is 1 kWh of electricity in the year of interest, and the evaluation includes the same 36 sustainability indicators covered in the previous section. A cradle-to-gate system boundary is adopted, in line with the previous case study, beginning with raw material extraction and ending with the generation of electricity at the power plant. Consequently, the transmission and distribution infrastructure are not included.

3.2.2 Scenario development

Three main scenarios are considered, each with either one or two subscenarios depicting possible futures for electricity in the United Kingdom to 2070; their characteristics are summarized in Table 5.6. All the scenarios are driven by the need to reduce GHG emissions, as this is one of the main energy policy drivers in the United Kingdom (DECC, 2011a,b). The three main scenarios explore three different GHG reduction levels for the electricity mix—65%, 80%, and 100%—by 2050 relative to 1990. The most ambitious of the three is based on the fact that, to achieve the national target of 80% overall reduction of GHG emissions, a 100% reduction is required in the electricity mix due to the greater difficulty of decarbonizing the heat and transport sectors. The 65% and 80% scenarios are chosen to examine the

Table 5.6 Summary of scenarios in this case study (all reductions refer to a 2009 baseline year)

Scenarios			Subscenarios	
65%	• Limited action is taken to prevent climate change • Total (direct) UK GHG emissions reduce by 24% (including international aviation and shipping) by 2070 • Electricity is significantly decarbonized, with emissions reduced by 65% by 2050 and 80% by 2070 • Electricity demand increases slowly, increasing by 50% by 2070	→ →	65%-1 65%-2	Subscenario with coal CCS but no new nuclear build. The mix in 2070: 68% fossil and 32% renewables Subscenario with both new nuclear build and coal CCS. The mix in 2070: 37% fossil, 30% nuclear, and 33% renewables
80%	• Decarbonization of electricity is intermediate between scenarios "65%" and "100%", reaching 80% reduction by 2050 (in line with Government targets for the whole economy) and eventually 98% by 2070 • Follows the same electricity demand profile as the 100% scenario	→	80%	Only one subscenario considered. Includes new nuclear build and some coal CCS. The mix in 2070: 10% fossil, 29% nuclear, and 61% renewables
100%	• Similar cumulative whole-economy GHG emissions to UKERC's "Carbon Ambition" scenario (UKERC, 2009) in line with the UK GHG budgets • Total UK GHG emissions reduce by 80%	→ →	100%-1 100%-2	Subscenario with no new nuclear build, dominated by solar PV and offshore wind. The mix in 2070: 100% renewables Subscenario with new nuclear build and

Continued

Table 5.6 Summary of scenarios in this case study (all reductions refer to a 2009 baseline year)—cont'd

Scenarios			**Subscenarios**	
	(including international aviation and shipping) by 2070 • GHG emissions from electricity are effectively zero by 2050 • Total energy demand reduces by 30% by 2070, but electricity demand increases by 60% as transport and other services switch to electricity (demand peaks in 2050 at 78% higher than 1990, then declines to 60% with efficiency improvements)			renewables. The mix in 2070: 50% nuclear and 50% renewables

implications of falling short of this target, with the 80% scenario matching the national target and 65% being less ambitious still.

It should be noted that the UK's emission reduction target refers only to direct emissions of GHGs rather than life cycle emissions. Therefore the reduction targets considered in the scenarios also refer to the direct emissions; however, the implications of reaching these targets are estimated on a life cycle basis.

The narratives for the scenarios are based on work by the Tyndall Centre (Azapagic et al., 2011) but have been developed further to focus solely on electricity.

The electricity mixes for each subscenario are shown in Fig. 5.10. As illustrated, the amount of coal and natural gas diminishes with time to be replaced by nuclear, wind, solar PV, biomass, and/or coal with carbon capture and storage (CCS). In the most aggressive scenarios, all fossil fuel (with and without CCS) is eliminated by 2050.

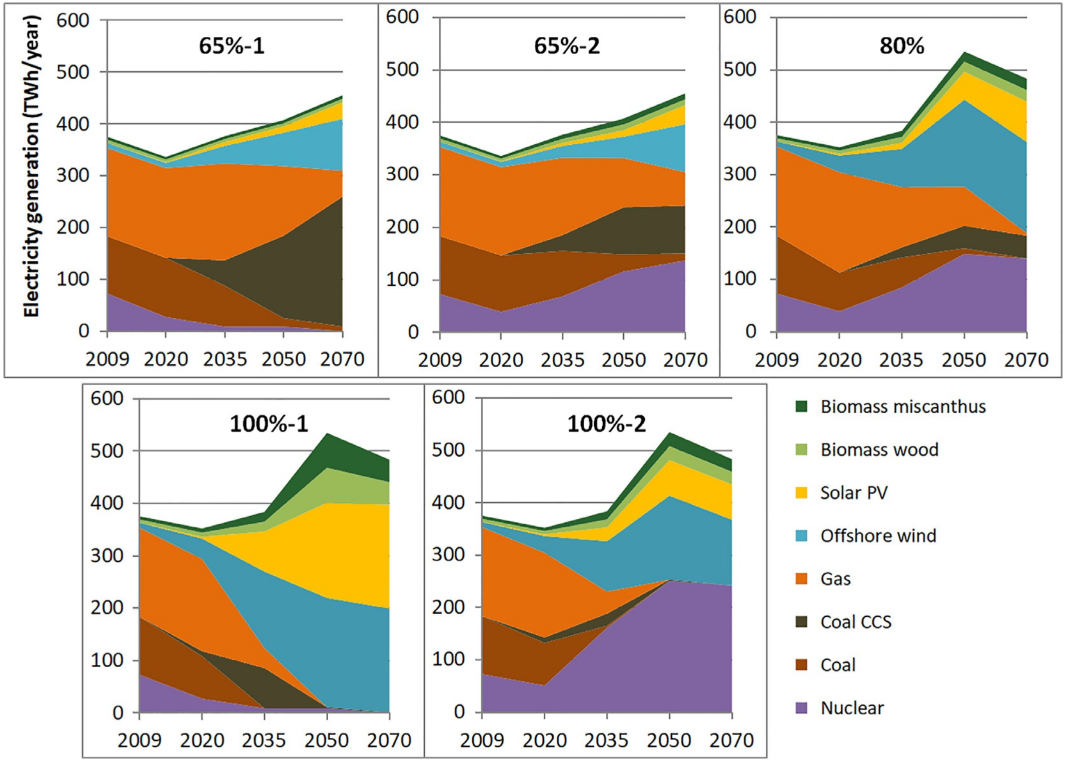

Fig. 5.10 Electricity mixes through time under the five subscenarios.

3.2.3 Inventory analysis

The environmental, economic, and social data for each technology are the same as those discussed in the previous case study (see Section 3.1) with two major amendments: firstly, coal with carbon capture and storage (CCS) is added due to its potential for future deployment; secondly, the impacts of coal CCS, wind, and solar PV are altered through time using learning rates and parameter amendment due to their relative immaturity and rapid development. The assumptions and data are outlined later, with more information available in Stamford and Azapagic (2014).

The use of *learning curves* to estimate future cost reduction for energy technologies is well established as a useful tool for scenario analysis (McDonald and Schrattenholzer, 2001; Rubin et al., 2007; Ferioli et al., 2009; van den Broek et al., 2009; IEA, 2013). It is defined as the reduction in costs that are achieved for every doubling of installed capacity, and this principle can be in LCSA as well as pure costing exercises (Stamford and Azapagic, 2018).

Prior work has suggested a learning rate for PV systems of approximately 18% (IEA, 2010) and 12% for wind (IEA, 2013). For coal CCS, the rate is less certain due to the lack of commercial deployment, but rates of 3.5%–4.9% have been estimated (Rubin et al., 2007).

As described in Ferioli et al. (2009), the relationship between cost reduction and cumulative output can be expressed as:

$$C(x_t) = C(x_0)\left(\frac{x_t}{x_0}\right)^{-b} \text{(monetary unit)}$$

where

x_t = Cumulative installed capacity at a point in time t (GW)
x_0 = Arbitrary starting point in time in cumulative capacity (GW)
C = Installed system cost at either x_0 or x_t (monetary unit)
b = A learning parameter (dimensionless)

The learning parameter b represents the slope of the power-curve fitting the cumulative installed capacity x_t against the installed costs C over the time period of interest. The learning rate LR can then be determined using b as follows:

$$LR = 1 - 2^{-b}\, x\, 100\ (\%)$$

In this case, learning curves are used to estimate future reductions in costs and employment levels (which are closely related to costs). Future change to other indicators derived from LCA and SLCA are based on predictions of technological development such as improvements in solar cell efficiency and wind farm capacity factor.

3.2.4 Interpretation

Due to the number of indicators and technology options as well as the time-span of the assessment, it is not possible to show all the outputs here. Interested readers are directed to Stamford and Azapagic (2014) for further detail. However, this section highlights some of the key results to illustrate how such an assessment can assist decision-making and policy.

As shown in Fig. 5.11, whichever scenario is pursued the dispatchability of the electricity mix deteriorates; in other words it becomes more difficult to match supply to demand from minute to minute. This is because, with the exception of biomass, the output of the low carbon technologies is either dependent on weather (wind and solar) or quite invariable for technical and economic reasons (nuclear). It is notable that dispatchability worsens even in scenarios that fail to meet the national GHG emission targets. This highlights the need for measures such as energy storage and demand-side management.

Capital expenditure rises greatly in future scenarios, primarily because low-carbon technologies tend to be capital intensive. This results in expenditure equivalent to £30–40 billion per year by 2070, highlighting the need for secure mechanisms of borrowing. In contrast, while the overall cost of electricity increases in all scenarios, the increase is less dramatic. Of the scenarios in which GHG emission targets are met, 100%-2 is the cheapest option by 2070, being only 14% more expensive than the current electricity mix per unit of electricity generated (8.7 c.f. 7.6 pence/kWh).

The GWP of annual electricity production reduces markedly in all scenarios (Fig. 5.12), falling from 184 Mt. CO_2 eq. in 2009 to a range of 10.5 Mt. (100%-2) to 51.4 Mt. (65%-1) by 2070. However, the importance

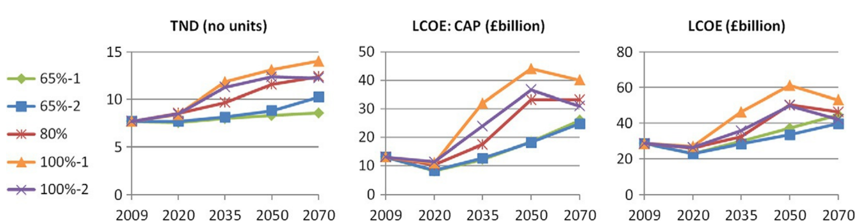

Fig. 5.11 Selected techno-economic indicators for all subscenarios to 2070, impacts expressed per year (for nomenclature refer to Fig. 5.6).

of considering the whole life cycle is clear when considering scenarios 100%-1 and 100%-2: despite both being zero-carbon at the point of generation by 2070, 100%-1 has an overall carbon footprint twice as high as 100%-2. This is due to the latter's greater reliance on wind and nuclear power.

Scenario 100%-2 also has the lowest acidification potential from the 2030s onwards, where most other scenarios achieve only modest reductions due to their use of coal CCS and biomass. Fig. 5.12 also shows that terrestrial ecotoxicity is likely to worsen in future. This is mostly due to the increased metal extraction and processing per unit of electricity generated, which results from greater reliance on renewables. It is also due to the spreading of ash from miscanthus combustion, as discussed in the previous section. Consequently, policy and regulatory oversight of both these industries will become increasingly important.

Future employment in the electricity sector looks set to increase regardless of the scenario chosen, as shown in Fig. 5.13. Across the whole life cycle, 100%-1 is estimated to employ 109,300 people by 2070, compared to 46,100 in 2009. This is mostly due to the labor intensive life cycles of wind and solar power. In contrast, human toxicity shows only modest changes throughout the time period.

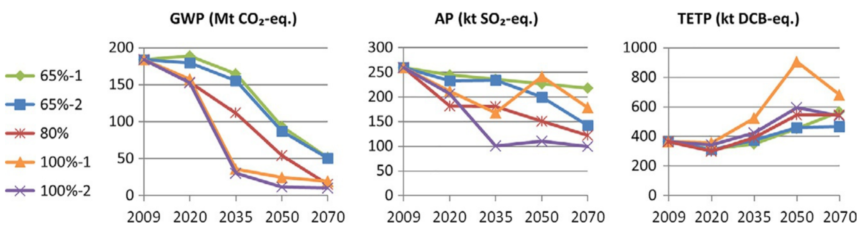

Fig. 5.12 Selected environmental indicators for all subscenarios to 2070, impacts expressed per year (for nomenclature refer to Fig. 5.7).

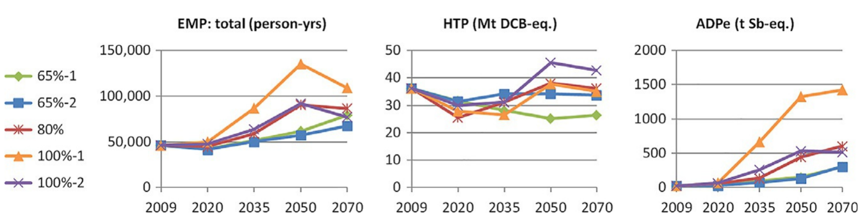

Fig. 5.13 Selected social indicators for all subscenarios to 2070, impacts expressed per year (for nomenclature refer to Fig. 5.8).

As mentioned before, renewables tend to require more metal resources per unit electricity generated when compared to fossil fuels. Consequently, depletion of elements increases greatly from 22 t Sb eq./year in 2009 to 298–1430 t by 2070 (see Fig. 5.13). This highlights the critical importance of reuse, recycling and implementation of circular economy principles in general over the coming years.

3.2.5 Case study conclusions

Robustly evaluating a complex LCSA can be very challenging. In such cases, multicriteria decision analysis can be an extremely useful tool, as discussed in Section 2.5. However, a simpler approach is a *summed rank analysis*. It should be stressed that this is a simplistic analysis that ignores both the distribution of results for individual indicators and the importance of the issues addressed by each indicator: each indicator is given equal weight within its group.

It is relatively straightforward to rank each subscenario against each sustainability indicator in the year 2070 and sum their ranks to obtain a single score; the lower the score, the better the option. To avoid bias resulting from the different number of indicators in the techno-economic, environmental, and social dimensions, the analysis should be hierarchical: summed ranks should first be created for each dimension and then the overall ranking estimated based on the summed ranks for the three dimensions.

When such an analysis is performed, the baseline 2009 electricity mix is preferable from the techno-economic perspective with a score of 37, followed by 100%-2 which has an equal share of nuclear and renewables, with 39. The renewable-intensive 100%-1 and CCS-intensive 65%-1 have the joint worst score of 47 for techno-economic performance. In terms of environmental impacts, 100%-2 has the best score of 21, followed by 80% with 25. The 2009 mix and 100%-1 are the worst ranked options. However, 100%-1 appears to be the best option from the social perspective, followed by 100%-2, with the 2009 mix being the worst.

Overall, the ranking suggests that all 2070 electricity mixes are superior to the 2009 mix with the exception of 65%-1 which scores the same. The best option, within the limitations of this simplified ranking approach, is 100%-2 (score of 5), followed by 80% (9). Thus it appears that aggressive decarbonization using a mix of nuclear power and renewables is likely to be the preferred route.

4 Conclusions

Technological progress in the field of energy is driven primarily by cost and carbon, but it is important not to lose sight of the broader goals of sustainable development. This means evaluating a broad range of issues simultaneously, spanning the three "pillars" of environment, economy, and society; these might range from carbon footprint to ecotoxicity, capital and levelized costs, human health impacts, employment provision and public support, among many others. In all cases it is important to take a life cycle approach in order to ensure that information being used to make decisions is holistic and does not ignore important impacts or simply shift problems from one part of the life cycle to another.

To achieve this, a variety of approaches exist in various stages of development. The key components of life cycle sustainability assessment (LCSA) are environmental LCA, life cycle costing and social life cycle assessment, the latter of which in particular is evolving rapidly. As LCSA improves to meet the demands of policymakers, industry and society in general, there is an opportunity for true interdisciplinary work, drawing on expertise from economics, engineering, toxicology, climate policy, and all of the social sciences.

Sustainable development is an aspirational goal for all of society and energy is one of its critical enablers. Since sustainable decision-making relies on robust, broad understanding of the systems we develop and operate, the role of LCSA is more important than ever.

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CHAPTER 6

Hot-spots and lessons learned from life cycle sustainability assessment of inedible vegetable-oil based biodiesel in Northern Viet Nam

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1 Introduction

Fossil fuel energy supply has steadily increased twofold from >5300 Mtoe in 1973 to around 11,110 Mtoe in 2014, providing >80% of total primary energy supply for four decades, despite increasing nonfossil energy (IEA, 2016a). This domination of fossil fuel is projected to continue until 2035 (BP p.l.c., 2016). Since fossil fuels are depletable, this will lead to a massive future burden on natural resources. Furthermore, fossil fuel combustion is the key driver of the surge in global carbon dioxide (CO_2) emissions, which reached 32 $GtCO_2$ in 2014 (IEA, 2016b). As carbon dioxide emissions are the major contributor to climate change, several substitutions of fossil fuel are of great interest to international communities regarding future energy guarantee and environmental and human well-being protection. Vegetable oil-derived biodiesel is considered as an ideal alternative to fossil diesel (petrodiesel) in the transport sector. This type of fuel is renewable and environmentally friendly, with the potential to mitigate climate change and cause less harm to human health (Achten, 2010). However, several disadvantages of biodiesel have also been indicated; for example, higher impacts on the ecosystem due to fertilizer and other agricultural chemical use (Achten, 2010), land-use changes (Fargione et al., 2008), and higher net production costs (Rajagopal and Zilberman, 2007). Due to both the pros and cons of biodiesel production and utilization, scholars have argued about net benefits and sustainable potential of biodiesel for years. To settle this controversy, biodiesel systems need to be evaluated with an appropriate sustainability assessment tool that can consider the trade-off between various positive and negative impacts of the system.

Viet Nam is an S-shaped country located in Southeast Asia with a long coastal line of about 3440 km starting from the Gulf of Tonkin to the South China Sea and the Gulf of Thailand. Viet Nam has a tropical monsoonal climate with high temperature and humidity. The nation has two main climate regions. In the north, the climate is highly humid tropical monsoon with four seasons including spring, summer, autumn, and winter. The southern and central regions have a moderate tropical climate with dry and rainy seasons.

Crude oil, natural gas, and coal are the three essential natural resources in Viet Nam. With the blooming in the national economy and the increasing population, the exploitation of crude oil has been boosted for decades for both domestic uses and exportation. Under the current technology, the potential crude oil reserves of the country have remained at 4.4 billion tonnes since 2011 (US EIA, 2017a). However, as crude oil is a limited resource, current national crude oil production has decreased by about 20% from 403,000 barrels per day in 2004 to 320,000 barrels per day in 2016 (US EIA, 2017a). Moreover, although Viet Nam is a net exporter of crude oil, this nation is also a net importer of oil products in which ~67% of total petrodiesel consumption is from foreign sources (Le et al., 2016; Viet Nam Customs, 2015). Meanwhile, the world oil price is unstable and fluctuates year by year. It is crucial for Viet Nam to diversify its fuel sources.

On the other hand, in Viet Nam, transportation plays an importation role in the development of the nation as a proper tool to strengthen economic activities and to support social welfare. However, transportation is also the most contributor to the increasing air pollution in the urban area, especially in Ho Chi Minh City (in the South) and Hanoi (in the North), two most major cities of the nation. Recently, the considerable amount of particulate matter in the urban ambient air has been a serious problem caused by the petroleum fuel combustion. On the other hand, Ha Long Bay—Quang Ninh Province, located in the Northeast of Viet Nam, possesses a stunning landscape with >1600 of limestone islands and islets. Ha Long has been inscribed in the Natural World Heritage Sites since 1994 and is one of the most popular tourist attractions in Viet Nam. However, ~550 cruise boats operating in Ha Long Bay consume about 22,000 kiloliters of fuel per year leading to several environmental problems in the Bay. Uncollected solid wastes, wastewater discharge, and fuel oil leakage from those boats are primary drivers of water quality degradation in this area.

Therefore biofuel in general and biodiesel, in particular, is recognized as an essential solution for the energy insecurity and current environmental issue caused by the transport sector in Viet Nam. In 2007 Viet Nam introduced a new Energy Development Scheme, in which by 2015, about five million tonnes of ethanol 5% (E5) and biodiesel 5% (B5) were expected to come into national use, with total biofuel 5% consumption predicted to reach 36 million tonnes by 2025. Henceforth, several efforts and activities from both the government and private sector have been conducted. Since then, this country has to struggle plenty of issues to enact this biofuel scheme and only just after various endeavors of the government E5 has been available on the market since January 1st, 2018. However, B5 is still under feedstock identification stage. Previous studies on biodiesel feedstocks in Viet Nam mostly focused on *Jatropha curcas* (MARD, 2010) and catfish fat (Rang, 2007), which only have high potential to apply in Central and Southern Viet Nam. This issue called for another attempt to reveal appropriate feedstocks for biodiesel production in the North of Viet Nam. Accordingly, a project namely "Multibeneficial Measures for Mitigation of Climate Change in Viet Nam and Indochina Countries by Development of Biomass Energy," funded by the Japan Science and Technology Agency (JST) and the Japan International Cooperation Agency (JICA), as one of projects of the Science and Technology Research Partnership for Sustainable Development was implemented from 2011 to 2016 (hereinafter called SATREPS Project). This project strived for identifying potential feedstocks for biodiesel production in each region of Viet Nam, including the northern part, and proposing a closed loop system of biodiesel production and utilization, starting from oil plant cultivation to biodiesel end-use. This system came with high expectations of reducing environmental problems and enhancing application of biodiesel, in order to support the green economic development in this country.

The intercropping of *Hibiscus sabdariffa* L. (Hibiscus) and *Vernicia montana* L. (Vernicia), and the cultivation of *Pongamia pinnata* (Pongamia) were highly recommended, due to their ability to grow well in low fertile soil and provide short- and long-term economic profits. Furthermore, as the extracted oils from Hibiscus-Vernicia and Pongamia seeds are inedible, their use has no conflict with food production in Viet Nam. Consequently, those plants can become feedstocks for the manufacture of biodiesel utilized in daily buses, coal mining dump trucks, and cruise boats in Northern Viet Nam, which require nearly 60,000 kiloliters fuel annually. Note that in this paper, inedible oil means not only oil that cannot be consumed, due to its low quality or toxicity, but also oil used for neither cooking nor any other form of food supply. As mentioned before, the implementation of biodiesel systems does not always mean win-win outcomes. The sustainability of the entire life cycle of new biodiesel systems needs to be evaluated.

This study aimed to access the sustainable potential of inedible vegetable oil-based biodiesel systems in Northern Viet Nam. Simultaneously, the felicity of different Hibiscus, Vernicia, and Pongamia biodiesel blends was examined. Options and recommendations for the sustainable development of inedible vegetable oil-based biodiesel in Northern Viet Nam were introduced subsequently.

2 Potential cultivation areas and feedstocks for biodiesel production in Northern Viet Nam

2.1 *Pongamia pinnata*

Pongamia pinnata (Pongamia), also known as Karanja, Pongam, or Indian beech, is a medium-sized evergreen or briefly deciduous tree up to 25 m tall and belongs to Fabaceae family. This plant is native to tropical and temperate Asia including Bangladesh, India, Myanmar, Nepal, and Thailand. Pongamia is well grown in deep well-drained moist sandy loam soil, up to 1200 m of elevation with the temperature range of 0–50°C and annual rainfall from 500 mm to 2500 mm. Moreover, it can tolerate various adverse climatic and soil conditions, including drought, saline and alkaline soils, and poor sandy and rocky soils (Orwa et al., 2009). Pongamia was introduced to Viet Nam primarily for mangrove forest plantation to prevent salinization and soil erosion in coastal areas or urban landscape planning. This plant has recently been recognized as a potential feedstock for biodiesel production that has average oil content in seed of ∼29.2% (Table 6.1). However, there is limited empirical evidence on the seed production of Pongamia, especially in Viet Nam. Therefore several assumptions were applied in this study.

2.2 *Hibiscus sabdariffa* L.

Hibiscus sabdariffa L. (Hibiscus) is an annual, erect shrub with an average height of about two meters which belongs to the Malvaceae family and is mostly distributed in tropical areas (McClintock and El Tahir, 2004). Hibiscus also has several other names, such as sorrel and jelly okra. Almost all parts of this plant are edible that can be used for many purposes, including as a vegetable or for calyx and fiber production, as well as for medicinal supplies. Viet Nam started to plant Hibiscus in 1957, and this plant has since gained its popularity for calyx production over the last decade. Favorable climatic conditions for the growth of Hibiscus are humid weather with temperatures ranging from 16°C to 38°C, and annual precipitation of 1500 mm (VAFS, 2009). In the North of Viet Nam, propagation begins from May

Table 6.1 Main chemical compositions and potential yield and cultivation areas of Pongamia, Vernicia, and Hibiscus seeds

	Composition (%)						
Variety	**Oil content kernel/seed**[b]	**Sugar**[b]	**Vitamin E**[b]	**Phytosterols**[b]	**Seed yield (kg tree^{-1})**	**Promising land-use types for cultivation**	**Available area for cultivation (thousand ha)**[a]
Pongamia	31.2/29.2	8.90	0.072	0.097	9–90[c]	Unused mountainous land[d] Rocky mountain without forest[e] Open-pit mines and mining dump sites	45.6 in Quang Ninh Province
Vernicia	58.0/32.6	1.42	0.148	0.069	4–11[f]	Unused mountainous land[d] Rocky mountain without forest[e]	997.2 in upland provinces near Viet Nam-China border
Hibiscus	NA[g]/20.0	3.85	0.001	0.023	0.15–0.22[h]		

[a]Data from Ministry of Environment and Natural Resources, 2013.
[b]Unpublised data from a research group in Osaka Prefecture University.
[c]Data summarized by Halder et al. (2014) and Murphy et al. (2012).
[d]Mountainous areas of which land use has not been identified yet.
[e]Barren areas in rocky mountains.
[f]According to Tran (1996).
[g]Not available.
[h]Data reported by several farmers in the north of Viet Nam and Pham (2016).

to June and fruits can be harvested after 6 months. Hibiscus calyces are famous for food uses in Viet Nam, mostly as fresh food or as an ingredient for making juice, wine, and jam. Hibiscus seeds are provided only for sowing. Average oil content of the seed is ~20% (Table 6.1).

2.3 *Vernicia montana* L.

Vernicia montana L. (Vernicia) is a wood tree up to 15 m tall belonging to the Euphorbiaceae family. Other names for Vernicia include wood-oil-tree, mu-tree, and abrasion-oil tree. This plant is native to Southeast Asia and southern China. Oil derived from Vernicia seed is a quick-drying oil, namely "Abrasin oil" which is commonly used for manufacturing paint or Chinese black ink (Oyen, 2007). Vernicia can be grown in areas with annual rainfall of 1600–2500 mm and average temperatures of 20–25°C (VAFS, 2009). In Viet Nam, Vernicia is a native plant, which is mainly distributed in the mountainous areas in the Northern and Central parts. Vernicia grows quickly, and its fruits start to bear after three years of sowing (Tran, 1996). In Northern Viet Nam, Vernicia seeds are directly sold to China after being harvested and sun-dried. Average oil content in the Vernicia seed is ~32.6% (Table 6.1).

2.4 Proposed cultivation areas

The development of the feedstock scenarios was from multi data sources. It started with the current Viet Nam policies on forest protection and development, land-use planning, socioeconomic development scheme, and coal mining development plan to figure out what activities could be supported and allowed in each region. Then, land-use status was obtained from the annual land-use report of Ministry of Environment and Natural Resources, Viet Nam. This study analyzed data about land-use of Viet Nam in 2013, and open-pit mines and mining dump site area in 2012 and 2014. After that, all those data were integrated with the information about feedstock yield, oil content, and potential feedstock for each region based on SASTREPS Project pilot sites. Current data showed that mountainous areas, especially provinces dwelling near to the national border, proved the highest amount of potential oilseed crop production.

Furthermore, under current policies of Viet Nam, investment in those areas could be supported by several policies including policies on forest protection and development (Law on Forest Protection and Development, 2004; Prime Minister, 2007a), land-use planning, and socioeconomic development schemes include regional development, upland provinces development, and supporting provinces dwelling near the border between Viet Nam and China frontier for economic development and national security (Prime Minister, 2007b). Moreover, coal mining in Quang Ninh Province contributed about 95% of total national hard coal production, both underground and open-pit mining. However, this activity has considerably affected natural resources and the environment of this area, including deforestation, forest degradation, soil erosion, abandoned mine lands, and water pollution. Therefore according to the National Mining Development Plan, all open-pit mines must be closed by 2020 that made ~6699 hectares (ha) of open-pit mine lands and mining dump sites need to be reclaimed in Quang Ninh Province.

To avoid the land-use conflict between oilseed crop cultivation with other economic activities, especially food crop production, this research only considered unused low fertile and degraded land areas (including unused mountainous land, rocky mountains without forest and mining reclamation areas) and unused oilseeds as potential sources for biodiesel feedstock acquisition. Table 6.1 presents data on biodiesel production and yield and potential land-use types.

2.5 Promising feedstocks for biodiesel production

In 2013 SATREPS Project started to implement a pilot plantation of several oil plants, including Pongamia, Vernicia, Jatropha (*J. curcas* L.), and Camellia (*Camellia oleosa*) in Quang Ninh Province. The following year, a total of 6700 7- to 8-month seedlings were transplanted into three hectares of a coal mining dump site (Nui Beo), in which the number of Pongamia, Vernicia, Jatropha, and Camellia seedlings were 500, 2200, 2000, and 2000, respectively. Initial results showed that after 18 months, Pongamia was the most feasible species to grow in this area since it had the highest growing rate of more than two times faster than other plants, and survival rate of 97%, in which that of Jatropha, Vernicia, and Camellia were 46%, 65%, and 86%, respectively (SATREPS Project's expert observation data).

Other practical data from various cultivation fields proved that Vernicia and Hibiscus could grow well in the low fertile soil and precipitation conditions of the Northern area of Viet Nam (Tran, 1996). On the other hand, previous studies denoted that although biodiesel from Hibiscus can meet almost all quality requirements according to biodiesel standards of Viet Nam (TCVN/QCVN) and other countries such as Japan (JIS K2390), the

United States (ASTM D6175), and Europe (EN 14214) (Anwar et al., 2010; Nakpong and Wootthikanokkhan, 2010; Nguyen and Otsuka, 2016), the yield of Hibiscus seed (200–1500 tonnes ha^{-1}) was not as high as Vernicia seed (1800–3000 tonnes ha^{-1}). However, Vernicia biodiesel could not meet several requirements of biodiesel standards (Table 6.2). Therefore an optimal blend of Hibiscus-Vernicia biodiesel was considered and investigated. Previous research proved that the volumetric mixture of 70% Hibiscus biodiesel and 30% Vernicia biodiesel is an appropriate combination (Nguyen and Otsuka, 2016). Main properties of Hibiscus-Vernicia and Pongamia biodiesel fuels compared to common biodiesel specifications are presented in Table 6.2.

Since 2013, market demand for Vernicia oil, as well as its price, has decreased gradually. Moreover, Hibiscus oil is unused material in Viet Nam. Hence, concerning favorable physicogeographical conditions of each plant, the employment of Pongamia in Quang Ninh Province and Hibiscus and Vernicia intercropping in high mountainous areas near Viet Nam and China border to produce oils as feedstocks for biodiesel production in Northern Viet Nam is practicable.

2.6 Goal and scope of the study

The sustainability of the entire biodiesel life cycle system in Northern Viet Nam was evaluated based on impacts of different Pongamia and Hibiscus-Vernicia biodiesel blend systems on five main enviro-economic categories including ecological footprint, biocapacity, ecosystem quality, human health, and costs and benefits. Results of Triple I were used to propose feasible options and implications for biodiesel policies toward sustainable development.

The scope of this study was limited to the North of Viet Nam.

2.7 System boundary and functional unit

The boundaries started with the production Pongamia, Vernicia, and Hibiscus oils (raw material acquisition) and ended with the combustion of several biodiesel fuels and their blends in targeted engines. Fig. 6.1 illustrates the system boundaries for the life cycle assessment (LCA) of biodiesel in Northern Viet Nam. The entire life cycle of biodiesel production was supposed to comprise all stages from cultivation of Pongamia in mining dump sites and other low to middle mountainous area, and intercropping of Hibiscus and Vernicia in high mountainous areas near national border; harvesting,

Table 6.2 Properties of different vegetable oil-based biodiesels in Northern Viet Nam compared to Viet Nam and international biodiesel standards

	Biodiesel				Biodiesel fuel standards[a]			
Property	**Hibiscus**	**Vernicia**	**H70V30[b]**	**Pongamia**	**TCVN/ QCVN**	**JIS K2390**	**ASTM D6751**	**EN 14214**
Ester content (% mass)	>98	>98	>98	>98	≥96.5	≥96.5		≥96.5
Kinematic viscosity at 40°C ($mm^2\ s^{-1}$)	4.39	7.70	5.46	4.85–5.43[c]	1.9–6.0	3.50–5.00	1.9–6.0	3.50–5.00
Density at 15°C ($g\,ml^{-1}$)	0.89	0.91	0.90	0.89[c]	0.86–0.90	0.86–0.90	–	0.86–0.9
Flash point (°C)	156[d]	167[e]	159.3	116–180[c]		≥120	≥130.0	≥101
Water ($mg\,kg^{-1}$)	<500	<500	<500	<500	≤500	≤500	≤500	≤500
Acid value (mg KOH g^{-1})	0.30	0.17	0.25	0.40–0.42[c]	≤0.50	≤0.50	≤0.50	≤0.50
Iodine value (g iodine $100\,g^{-1}$)	99	160	118	89[c]	≤120	≤120	–	≤120
Total glycerol (% mass)	<0.1	<0.1	<0.1	<0.1	≤0.24	≤0.25	≤0.24	≤0.25
Solubility at 25°C (ppm WAF concentration)	1.86	5.28	2.20	–	–	–	–	–
Interfacial tension ($mN\ m^{-2}$)	30.84	32.66	31.38	–	–	–	–	–

[a]TCVN/QCVN, JIS K2390, ASTM D6751 and EN 14214 are biodiesel specifications of Viet Nam, Japan, the United States, and Europe, respectively.
[b]Volumetric mixture of 70% Hibiscus biodiesel and 30% Vernicia biodiesel.
[c]Data from Atabani et al. (2013) and Meher et al. (2004).
[d]Data from Anwar et al. (2010).
[e]Data from Shang et al. (2010).

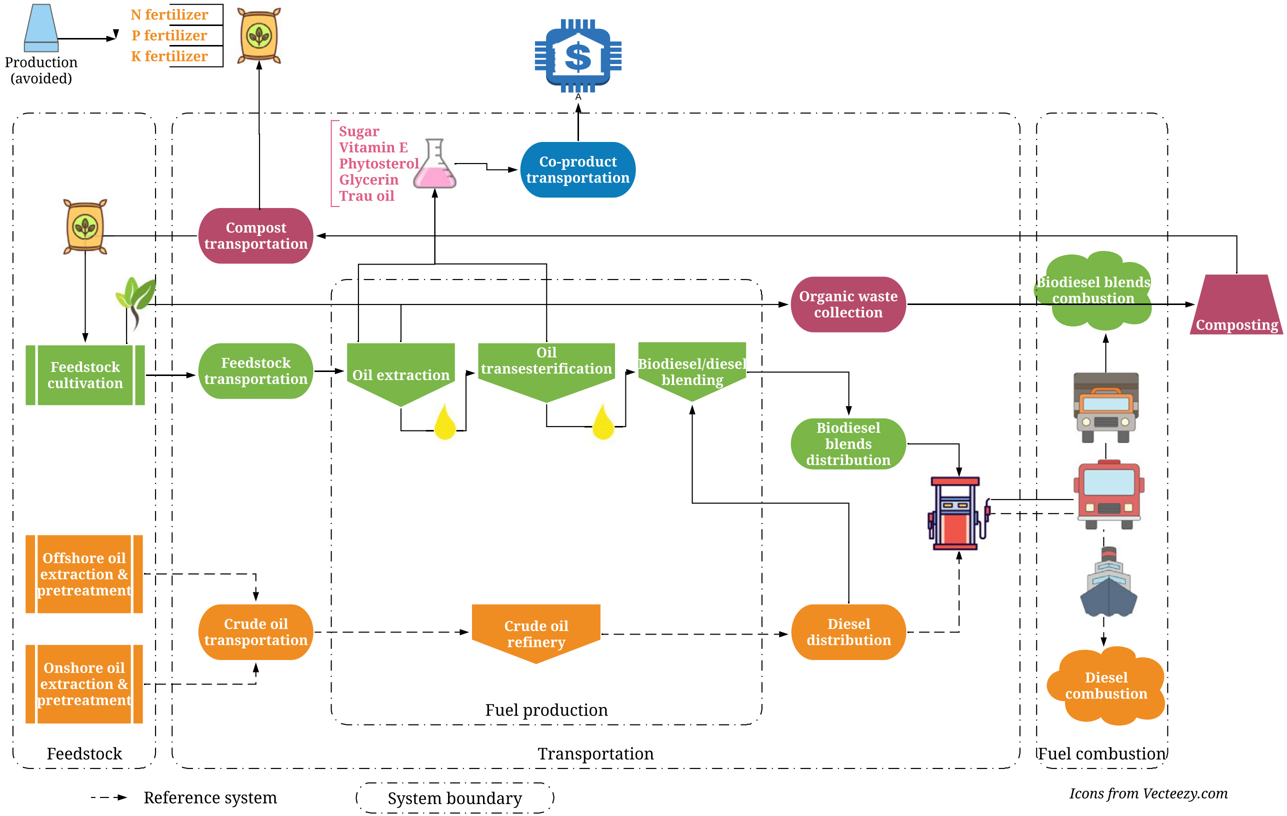

Fig. 6.1 System boundary of biodiesel production and use in Northern Viet Nam.

sun-drying, and transportation of oilseeds; extraction of oil and other medicines and coproducts from those seeds; esterification of Pongamia, Hibiscus-Vernicia crude oils to obtain biodiesel (fatty acid methyl esters); distribution and use of biodiesel in daily buses, including Ha Noi city buses, and long-distance buses from Ha Noi to Quang Ninh and other Central and Southern cities, and coal mining dump trucks and cruise ships Quang Ninh Province; Hibiscus leaves and Pongamia, Hibiscus-Vernicia deoiled cake used as composts back to the cultivation field to offset a certain amount of mineral fertilizer use according to the nutrient component in dry matter.

Vernicia trees have a long lifetime of about 50–70 years (Nipakhonsom et al., 2012) and their maximum production can last for 30–40 years (Bernál et al., 2014; Morton, 1987). Likewise, Pongamia has perennial nature with >80 years lifespan (Chandrashekar et al., 2012). Furthermore, the lifetime of oil mills for oil extraction and chemical plants for the esterification of vegetable oil mostly ranges from 25 to 50 years (Azadi et al., 2014; Jungbluth et al., 2007). Therefore the project lifetime in this study was set to 30 years. The functional unit for the life cycle assessment was 1-year biodiesel combustion in designated vehicles in Northern Viet Nam, which consume ~60,000 kiloliters of fuel annually.

3 Inclusive impact index (Triple I)

Triple I was used as a final indicator for the sustainability assessment (Eq. 6.1).

$$\mathrm{III} = (\mathrm{EF} - \mathrm{BC}) + \alpha[(C - B) + \beta\mathrm{HR} + \gamma\mathrm{ER}] \tag{6.1}$$

where EF is ecological footprint (global hectare (gha)); BC is biocapacity (gha); ER is ecological risk; C is costs (US \$); B is benefits (US \$); HR is human risk; and α, β, and γ are the conversion factors from economic value (US \$) to gha, from HR value to economic value (US \$), and from ER value to economic value (US \$), respectively. The estimation of all parameters in Triple I was conducted following the Triple I framework developed by Nguyen et al. (2017), in which an LCA tool so-called IMPACTS 2002+ was adopted to estimate the HR—human health impacts [Disability Adjusted Life Years per person (DALY pers^{-1})] and ER—ecosystem quality impacts [Potential Disappeared Fraction of species on 1 m^2 of earth's surface during 1 year (PDF m^{-2} year^{-1})]. EF and BC were calculated under the life cycle-based ecological footprint assessment (Huijbregts et al., 2008) with updated equivalent factors according to a new guideline from Global Footprint Network (Lin et al., 2016). Life cycle costing (LCC) was applied to estimate cost and benefit parameters of Triple I. The development and calculation of the entire system were operated by integrating Simapro 8.5 with a spreadsheet.

3.1 Ecological footprint and biocapacity estimation method

Ecological footprints related to the production of raw materials and their transportation and energy used were supported by Simapro 8.5. This study only considered yearly average carbon storage in standing biomass (Vernicia and Pongamia trees) and harvested Vernicia seeds, based on proportion of Vernicia oil sold for uses other than producing biodiesel. Carbon embodied in other agricultural residues (leaves and branches) and oil cake were not considered because the absorbed CO_2 would release back to the environment due to burning or composting. Moreover, due to the carbon cycle, CO_2 content in calyces used for food supplies would release back to the environment right after consumption. Yield factor was calculated with data of 2014 from FAOSTAT (2017). Accordingly, yield factor for cereals is 1.43 ha wha^{-1} (wha is the world average hectares of a given land-use type) and oil crops is 1.05 ha wha^{-1}. Land occupation other than for oil crop cultivation was not included in this calculation.

When using an area for oil crop propagation, that area will turn into arable land. Since the cultivation was practiced in mine dumping sites and other the marginal lands, this would result in the gain of the agricultural productive area, meaning an increase in biocapacity. The required plantation area to acquire a certain expected amount of biodiesel was estimated by integrating biodiesel production efficiency with Pongamia and Hibiscus and Vernicia seed yields and their oil contents. If the total required area was less than available area in Northern Viet Nam, this indicated the (+) biocapacity of the system. Vice versa, if the total required area was larger than the available one, this was indicative of the ecological footprint or (−) biocapacity. Then, the final biocapacity was the sum of (+) and (−) biocapacity.

3.2 Conversion factor calculation

Several scholars noted the close relationship between countries' gross domestic product (GDP) and their ecological footprints (Rainham and McDowell, 2005). Therefore to convert from economic value to global hectare, the ratio of total EF of the country/region, where the target system is implemented, to its gross domestic product (GDP) in the same year was applied (Otsuka, 2011) (Eq. 6.2).

According to the International Monterey Fund (2018) and Global Footprint Network (2018) and, the gross domestic product (GDP) of Viet Nam, as of 2014, were ~\$186 billion and 160 million gha, respectively.

$$\alpha = \frac{EF_{2014}}{GDP_{2014}} = \frac{160.3 \times 10^6}{185.9 \times 10^9} = 8.62 \times 10^{-4} \left(\text{gha}\,\$^{-1}\right) \qquad (6.2)$$

Since a "healthy" individual can contribute to a country's economy during that person's lifetime, the number of years lost due to death and disability means the noneconomic-contributing period of that person. Therefore the monetary value of DALY per person was computed by multiplying the value by GDP per capita (β) in the same year (Dalal and Svanström, 2015). Therefore the conversion factor β was set to \$2343 based on GPD per capita of Viet Nam in 2017 (The World Bank Group, 2018).

With efforts to develop a worldwide database on the value of ecosystem services, the Foundation of Sustainable Development collected and summarized various studies related to monetary valuation of ecosystem services (van der Ploeg et al., 2010). According to the database, monetary values of coral reefs and mangroves in Viet Nam with different services, including recreation, food, raw materials, medical, gene pool, and nursery, varied from $\$0.165\,\text{ha}^{-1}\,\text{year}^{-1}$ to $\$2363.8\,\text{ha}^{-1}\,\text{year}^{-1}$. Thus the conversion factor γ was estimated as average monetary values of ecosystem services in Viet Nam. $\gamma = \$526.417\,\text{ha}^{-1}\,\text{year}^{-1}$.

4 Life cycle inventory (LCI)

4.1 Determination of exhaust gas composition

Generally, the effects of biodiesel and its blends on engine performance and emissions vary due to the diversity of, for instance, the origins of biodiesel including oil seeds and climate conditions where the oilseeds grow, and the types of engines and their working conditions (Atabani et al., 2013; No, 2011). In fact, available information on exhaust gases from vehicles using biodiesel and combustion gases of Pongamia, and Hibiscus and Vernicia biodiesels are limited and not evident enough for a specific estimation. Therefore to determine the difference between exhaust gases of petrodiesel and biodiesel blends, this study combined a regression model for predicting the percent change in exhaust emissions based on the concentration of biodiesel in the blend developed by United States Environmental Protection

Agency (US EPA, 2002), with base case emissions of petrodiesel in road transport vehicles (buses and heavy-duty truck) and maritime navigation from a report about emissions of transport in the Netherlands (Klein et al., 2016). Although these findings may not reflect a quantitatively accurate prediction of the actual differences, they could provide a proper trend for comparison.

4.1.1 Variations in the characteristics of combustion emissions from diesel engines with diesel and biodiesel blends

US EPA (2002) developed regression models for estimating the percent change in exhaust emissions as a function of the concentration of biodiesel in conventional diesel fuel. This study has been widely recognized and applied in various biodiesel assessment reports (Air Quality Expert Group, 2011; Charman et al., 2012; Lapuerta et al., 2008). The original equation was adapted following the scope of this study.

Regression equations were applied to estimate the difference between the use of biodiesel and diesel in Viet Nam. The estimation was made using the following equation:

$$SF_x = e^{(a_x \times \text{vol\%bdf})} \tag{6.3}$$

where SF_x is emission scaling factor of emission x and a_x is coefficient related to emission x which were considered as statistically significant with 95% confident (Table 6.3), and vol%bdf is the volumetric percentage of biodiesel in the blend ranging from 0 to 100.

Table 6.3 Coefficients of basic emission correlations (US EPA, 2002)

Emissions	Coefficients
NO_x	0.0010375
PM	−0.047395
HC	−0.0118443
CO	−0.0058238
CO_2	0.0000177
Acetaldehyde	−0.001606
Ethylbenzene	−0.006970
Formaldehyde	−0.001696
Naphthalene	−0.002847
Xylene	−0.004078

4.1.2 Biodiesel effects on gaseous toxics

Scaling factors of toxic gaseous for different BDF blends relative to diesel were calculated as follows (US EPA, 2002):

$$SF_{TG} = a_{TG} \times \text{vol\%bdf} + 1 \tag{6.4}$$

where a_{TG} is the coefficient related to emission x which was considered as statistically significant with 90% confident (Table 6.3).

4.1.3 Base case emissions from petroleum combustion

The base case emissions of petrodiesel were obtained from spreadsheet data attached with a report on the calculation method of the transport emissions in the Netherlands (Klein et al., 2016). Characteristics of exhaust gases from diesel engines in buses, light/heavy-duty trucks, and maritime navigation were used. All data were from 2014; however, data from 1999 were used for SO_2 due to the petrodiesel standard of 1999 in the Netherlands being the same as current petrodiesel specifications in Viet Nam, with a sulfur content of up to 500 ppm. Since the sulfur content within the fuel positively correlates with the emission of SO_2 in exhausted gas (IPCC, 2006; Kristensen, 2012), the concentration of SO_2 was extracted from the percentage of biodiesel in the fuel. According to biodiesel specifications of Viet Nam (QCVN 1:2015/BKHCN), the sulfur content in biodiesel should be <10 ppm and <500 ppm in terms of petrodiesel.

In that report, Klein et al. (2016) considered several sources of emissions from the engine operation. Total emissions from road transport, for example, include tailpipe emissions, evaporative emissions from road vehicles, and PM emissions from tire and brake wear and road abrasion. In the case of maritime navigation, only exhaust emissions including SO_2, nitrous oxide (N_2O), ammonia (NH_3), heavy metals and volatile organic compounds (VOC)/polycyclic aromatic hydrocarbons (PAH) components were monitored.

Emission factors for calculating transport emissions of several vehicles are presented in Table 6.4. The data were analyzed under the condition of the Dutch transportation system.

According to Klein et al. (2016), combustion emissions mainly include CO, nitrogen oxides (NO_x), particulate matter (PM_{10}), N_2O, NH_3, methane (CH_4), SO_2, CO_2, VOC and PAH components and heavy metals; and evaporative emissions are VOC components, only accounted in the case of petrodiesel used.

Table 6.4 Emission factors of cruise ship, bus, and heavy-duty truck (Klein et al., 2016)

	Emission factor ($g\,kg^{-1}$ fuel)		
Emission	**Cruise ship**	**Bus**	**Heavy-duty truck**
NO_x	50	33	52.2
PM	4	0.862	1.033
SO_2	3.4	0.933	0.933
CO	10	5.203	3.356
CO_2	3173	3173	3173
Acetaldehyde	0.114	0.086	0.149
Ethylbenzene	0.03	0.013	0.022
Formaldehyde	0.348	0.058	0.1
Naphthalene	0.04	0.01	0.016
CH_4	0.24	0.058	0.1

4.1.4 Fuel leakages and use

In consultation with a maritime engine expert in Ha Long Bay, it was found out that 1% and 2% fuel leakage were applied to new (operating from 2010) and old (operating before 2010) engines, respectively. According to the board registration record, there were 81% of ships registered before 2010 and the others accounted for 19%. Thus the fuel leakage rate was set to 1.8%.

Other studies on biodiesel derived from soybean (US EPA, 2002) and *Hibiscus cannabinus* (Jindal and Goyal, 2012; Sorate, 2013) claimed that fuel consumption of pure biodiesel is from 9% to 12% higher than that of petrodiesel, due to the lower calorific value and higher density. However, those statements were not evident enough to determine which part was due to the lower calorific value of biodiesel and which part was the contribution of the fuel density. Moreover, the functional unit of the system was based on volumetric consumption of fuel (kiloliters per year) then allocated to the mass value considering the difference in density of fuels. Therefore to avoid double counting, no adjustment in fuel consumption between biodiesel and diesel was employed.

Annually, ~550 cruise ships operating in Ha Long Bay, coal mining dump trucks, city buses in Ha Noi (data from SATREPS Project), and long-distance buses from Ha Noi to other major cities/provinces (own computation data based on fuel consumption rate and distances) consume ~60,000 kiloliters of petrodiesel. According to fuel densities, annual mass fuel consumptions were changed following the type of biofuel used (Table 6.5).

Table 6.5 Annual fuel consumption by biodiesel and its blends

Types of fuel	Petrodiesel B0 (tonnes)	B5 (tonnes)		B10 (tonnes)		B20 (tonnes)		B100 (tonnes)
		B100	Total	B100	Total	B100	Total	
Pongamia blend CS[a]	17,755	945	17,813	1891	17,871	3782	17,966	18,908
Pongamia blend[b]	9421	502	9451	1003	9482	2008	9544	10,032
Sub-total (Quang Ninh)	*27,176*	*1447*	*27,264*	*2894*	*27,352*	*5789*	*27,530*	*28,941*
Hibiscus-Vernicia blend (Ha Noi)[c]	22,041	1183	22,122	2366	22,203	4732	22,365	23,661
Total	***49,217***	***2630***	***49,386***	***5260***	***49,555***	***10,522***	***49,895***	***52,601***

[a]Fuel used in cruise ships in Quang Ninh with the fuel leakage of 1.806%.
[b]Fuel used in coal mining dump trucks in and buses from Quang Ninh to Ha Noi.
[c]Fuel used in Ha Noi city buses and long-distance buses from Ha Noi to Quang Ninh, Central cities, and Ho Chi Minh (in the South of Viet Nam).

4.1.5 Evaporation weathering in the marine environment

Evaporation rate and components of diesel vapors were estimated through the previous study on diesel components and weathering behaviors. Table 6.6 lists main components of diesel from a study of Wang et al. (2003), which analyzed the composition of diesel fuel oil no.2 in Canada.

Data from our previous study showed that in the case of petrodiesel, 75% of the oil spill was rapidly volatilized within 5 days after the spill (Nguyen and Otsuka, 2016). This is also in accordance with the study from the US National Research Council indicating that the evaporation weathering of diesel and fuel oil no.2 spill would lead to 75% or more of fuel release into the atmosphere (US National Research Council, 1975).

Aliphatic and aromatic compounds contribute about 98.4% of diesel mass. On the other hand, oil weathering processes also affect the concentrations of components existing in a fuel and its evaporation process (US National Research Council, 1975). Although aromatics have higher water solubility than aliphatics, they also show higher vapor pressure. Since total petroleum hydrocarbons comprise a significant amount of various components and vary between fuels, obtaining a detailed physiochemical analysis of diesel fuel is impossible (Brewer et al., 2013). Consequently, this study downscaled total percent of saturates and aromatics to 75% with the equal allocation. This made the evaporation rates of aliphatics and aromatics become 67% and 8%, respectively. Since the presence of biodiesel does not affect the evaporation behavior of diesel components in the blend (DeMello et al., 2007), the rate of evaporation of oil spills was allocated based on its volumetric contribution.

4.2 Allocation methods

Regarding the allocation methods of biodiesel and coproducts obtained throughout biodiesel life cycle system, several allocation approaches applied were as follows:

– The cut-off approach was used for marketable coproducts of the system and composts from Hibiscus leaves and Pongamia and Hibiscus-Vernicia oil cake. Accordingly, sugar, medicinal compounds, Hibiscus calyces, and

Table 6.6 Components of diesel by hydrocarbon groups (Wang et al., 2003)

Component	Concentration (weight %)
Saturates	88.2
Aromatics	10.2
Resins	1.7
Waxes	1.7

glycerin were immediately sold to the market without any further processing. Regarding reapplied compost, the amount of compost applied back to the field was determined based on nutrient components of its origin and requirements from the cultivation.

– The consequential approach was applied for surplus composts that would be used in other fields. Being based on nutrient components of Hibiscus leaves and Pongamia and Hibiscus-Vernicia oil cake, this paper assumed that surplus composts could be used to avoid the relative amount of mineral fertilizer including urea, phosphorus, and potassium fertilizers.
– The closed-loop scenario was developed in terms of reapplied composts into the field.

4.3 Net present value (NPV) and discount rate calculation

The computation of NPV was as follows (Huppes et al., 2004):

$$\mathrm{NPV} = \sum_{t=0}^{n} \frac{C_t}{(1+r)^t}\,(\mathrm{US\,\$}) \tag{6.5}$$

where n is the period of assessment (year), r is the discount rate, and C_t is the estimated costs in year t. In Triple I, the time-equivalent value of total one-time payment (TP) is considered under the adjustment of NPV, in which n is the project lifetime, and C_t is the average amount of TP over project lifetime period. The discount rate (r) is a key factor in the estimation of NPV mostly influenced by the inflation and interest rate (Eq. 6.6) (Davis et al., 2005).

$$r = \frac{\mathrm{Rate}_{\mathrm{Interest}} - \mathrm{Rate}_{\mathrm{Inflation}}}{1 + \mathrm{Rate}_{\mathrm{Inflation}}} = \frac{6.5\% - 2.67\%}{1 + 2.67\%} = 3.74\% \tag{6.6}$$

The average inflation rate of Viet Nam from January 2016 to December 2016 was 2.67% (calculated based on consumer prices) (Trading Economics, 2017). The interest rate was 6.5%, according to the State Bank of Viet Nam.

4.4 Base case assumption

To date, no practical information on proper fertilizers used in Pongamia cultivation and Hibiscus-Vernicia intercropping field in Viet Nam has been provided. Therefore the base case assumption of annual fertilizer use was developed from the previous literature concerning current conditions in the North of Viet Nam.

In mountainous areas of Viet Nam, the propagation of Vernicia was direct seed sowing with no care and fertilizer use. Moreover, Vernicia cultivation guidance noted that when planting Vernicia concurrently with another annual crop on the same field, the use of separate fertilizer for Vernicia is unnecessary (Vietnamese Academy of Forest Sciences, 2009). Consequently, the amount of fertilizer required in Hibiscus monoculture was used as the annual fertilizer input of the intercropping field. However, since Hibiscus and Vernicia are supposed to be planted in low fertile soil, 100 kg ha^{-1} of urea fertilizer use was set to ensure the growth of tree and seed yield. This assumption was based on the application in Malawi, in which 50 kg of nitrogen (N) ha^{-1} is applied to increase fruit yields (Morton, 1987). The dose of 100 g DAP (diammonium phosphate) equivalent per plant was also applied in Pongamia field to maintain and assure its seed production in low fertile soil (Wani et al., 2006). On the other hand, although former pilot Pongamia plantation in Quang Ninh Province applied the tree density of 3000 trees ha^{-1} could confirm their growth rate, the potential of seed production is still unknown. Moreover, several scholars claimed that seed productivity of Pongamia considerably depends on planting density (Niemiec, 2015; Syamsuwida et al., 2015) and the tree density of up to 500 tree ha^{-1} was considered to be appropriate (Murphy et al., 2012). Therefore the planting density of 500 trees ha^{-1} was adopted in the base case scenario of this study.

Hibiscus leaves and Pongamia and Hibiscus-Vernicia seed cake were used to offset fertilizer use on the field since they have a high nutritional composition (McClintock and El Tahir, 2004) and have been recognized as excellent organic fertilizer (Do and Nguyen, 2003; Wani et al., 2006). The amount of composts applied to the field was calculated from the nutrient components of the leaves and seeds for each year according to the literature. Average nutrient compositions of Hibiscus leaves and Pongamia, Hibiscus, and Vernicia seedcake are described in Table 6.7.

Table 6.7 Nutrient components in Hibiscus leaves, Pongamia oil cake, Hibiscus oil cake, and Vernicia oil cake

Component	Nitrogen (N %)	Phosphorus (P %)	Potassium (K %)
Pongamia oil cake[a]	5.21	0.56	0.91
Hibiscus fresh leaves[b]	2.08	1.17	0.29
Hibiscus oil cake[c]	4.94	0.63	0.03
Vernicia oil cake[d]	3.50	0.97	0.50

[a]Data from Wani et al. (2006).
[b]Data from Al Shooshi (1997) and Nnebue et al. (2014).
[c]Average data from Al Shooshi (1997), Duke (1983), Hainida et al. (2008), McClintock and El Tahir (2004), and Morton (1974).
[d]Data from Do and Nguyen (2003).

The assumption was also made that there would be no change in fuel consumption in the future.

4.5 Base case scenario

4.5.1 Market of petroleum diesel in Viet Nam

- Domestic offshore crude oil was derived from Bach Ho offshore oil field, which contributed to more than half of the country's crude oil production (US EIA, 2012).
- Foreign crude oil was imported from Middle East onshore fields (mostly from Azerbaijan).
- Dung Quat refinery, the first large-scale refinery of Viet Nam, used mixed crude oil, in which 80% domestic oil and 20% foreign crude oil was used to produce petrodiesel (Le et al., 2016).
- The total amount of diesel in the market of Viet Nam consisted of around 33% and 67% from domestic and imported sources, respectively (Vietnam Customs, 2015).

In general, key phases in life cycle of petrodiesel in Viet Nam include extraction of crude oil from offshore (domestic) and onshore (Middle East); transport of crude oil to Dung Quat oil refinery to produce diesel fuel, which contributes about 33% to the diesel market in Viet Nam, with the remaining 67% being imported from other countries, mostly Singapore, Thailand, and China.

4.5.2 Inedible vegetable oil-derived biodiesel life cycle in northern Viet Nam

The inventory data for the biodiesel system includes Pongamia cultivation, Hibiscus-Vernicia intercropping, vegetable oil extraction, biodiesel production, transportation, and end-use stages, are presented in Table 6.8. Life cycle stages of biodiesel production and use in Northern Viet Nam are as follows (Fig. 6.1):

- *Feedstock propagation*: In high mountainous areas (Northwest), Vernicia seeds are planted in the nursery for 8 months for germination and then transplanted to the field. The plantation method of Hibiscus involves direct seed sowing. Since the Vernicia and Hibiscus were intercropped, the appropriate tree density of Vernicia was 400 trees ha^{-1} (Vietnamese Academy of Forest Sciences, 2009), and for Hibiscus was 25,000 trees ha^{-1} and then thinning to around 10,000 trees ha^{-1} (Vietnamese Academy of Forest Sciences, 2009). Similarly, in Quang Ninh Province, 8-month Pongamia seeds are also transplanted to the field with the density of 500 trees ha^{-1} (refer to session 6.4.4 for detail base case assumption).

Table 6.8 Life cycle inventory of 1 tonne of biodiesel production in Northern Viet Nam by unit process

No.	Process	Hibiscus-Vernicia biodiesel (Ha Noi)			Pongamia biodiesel (Quang Ninh)		
		Input	Co-/products	Cost[a] ($)	Input	Co-/products	Cost[a] ($)
1	**Feedstock cultivation**						
1.1	***Seedling and soil preparation***[b]						
	Land (ha)	2.36	–	–	0.39	–	–
	Fertilizer ($)	–	–	1588.92	–	–	326.14
	Manure (tonne)	1.89	–	–	0.39	–	–
	NPK 16-19-16 (kg)	94.43	–	–	19.38	–	–
	NPK 20-10-10 (kg)	21.34	–	–	2.93	–	–
	Phosphate fertilizer (kg)	17.48	–	–	2.40	–	–
	Transport (tkm)	220.89	–	311.97	1.31	–	1.90
	Vernicia seed (kg)	4.16	–	7.49	0.28	–	1.38
	Nursery bags (kg)	29.80	–	110.40	4.09	–	15.16
	Labor cost—seedling	–	–	644.44	–	–	105.92
	Seedling site preparation	–	–	3.83	–	–	0.53
	Waste treatment—seedling	–	–	0.17	–	–	0.02
1.2	***Intercropping/cultivation***						
	Fertilizer ($)	–	–	298.67	–	–	6.69
	Manure (kg)	2700.20	–	–	–	–	–
	Phosphate fertilizer (kg)	284.21	–	–	51.44	–	–
	Potassium fertilizer (kg)	142.85	–	–	–	–	–
	Urea (kg)	10.88	–	–	1.14	–	–

Continued

Table 6.8 Life cycle inventory of 1 tonne of biodiesel production in Northern Viet Nam by unit process—cont'd

No.	Process	Hibiscus-Vernicia biodiesel (Ha Noi)			Pongamia biodiesel (Quang Ninh)		
		Input	Co-/products	Cost ($)	Input	Co-/products	Cost ($)
	CO_2 (tonne)	30.86	–	–	4.18	–	–
	Labor cost—cultivation	–	–	513.69	–	–	65.31
	Transport (tkm)	390.15	–	327.65	0.15	–	0.13
	Pongamia seed (kg)	–	–	–	–	3876.24	–
	Vernicia seed (kg)	–	8966.18	–	–	–	–
	Hibiscus seed (kg)	–	4098.46	–	–	–	–
	Hibiscus dried calyces (kg)	–	1074.68	(5814.02)	–	–	–
	Phosphate fertilizer (kg)	–	109.64	–	–	41.67	–
	Potassium fertilizer (kg)	–	18.75	–	–	28.28	–
	Urea (kg)	–	222.28	–	–	156.53	–
2	**Vegetable oil extraction plant**						
2.1	***Plant construction cost***[b]						
	Machinery	–	–	2145.49	–	–	684.73
	Installation	–	–	172.86	–	–	55.17
	Rental land	–	–	13.82	–	–	4.41
2.2	***Solvent***						
	Methanol (kg)	708.33	–	849.99	306.22	–	367.47
	Hexane (kg)	855.14	–	855.14	253.89	–	253.89
	Water (kg)	13,055.57	–	7.57	3876.24	–	2.25
	Electricity (MJ)	11,213.44	–	191.18	2982.08	–	64.74

	Transport (tkm)	8782.04	–	7376.91	108.82	–	91.41
	Labor cost	–	–	35.32	–	–	12.00
	Land management fee	–	–	0.02	–	–	0.01
	Maintenance	–	–	145.59	–	–	44.16
	Taxes and insurance	–	–	5.02	–	–	1.52
	Vernicia oil (kg)	–	2776.83	–	–	–	–
	Hibiscus oil (kg)	–	768.44	–	–	–	–
	Pongamia oil (kg)	–	–	–	–	1075.27	
	Deoil cake (kg)	–	9510.31	–	–	2800.97	
	Phytosterol (kg)	–	3.12	(467.34)	–	0.08	(11.96)
	Sugar (kg)	–	216.98	(366.55)	–	306.73	(306.73)
	Vitamin E (kg)	–	6.66	(1665.08)	–	2.35	(587.66)
3	**Biodiesel esterification plant**						
3.1	***Installation cost***[b]						
	Plant construction and installation	–	–	591.68	–	–	591.37
	Rental land	–	–	4.45	–	–	4.44
3.2	***Operating and maintenance***						
	Sulfuric acid (H_2SO_4) (kg)	–	–	–	10.75	–	3.01
	Acetone (kg)	21.51	–	81.72	21.51	–	81.72
	Methanol (kg)	129.54	–	155.45	126.47	–	151.77
	Potassium hydroxide (kg)	5.38	–	3.23	5.38	–	3.23
	Water (kg)	689.66	–	0.40	1334.82	–	0.77
	Electricity (MJ)	148.72	–	2.89	156.23	–	3.04

Continued

Table 6.8 Life cycle inventory of 1 tonne of biodiesel production in Northern Viet Nam by unit process—cont'd

		Hibiscus-Vernicia biodiesel (Ha Noi)			Pongamia biodiesel (Quang Ninh)		
No.	**Process**	**Input**	**Co-/products**	**Cost ($)**	**Input**	**Co-/products**	**Cost ($)**
	Transport (tkm)	149.06	–	125.21	3.99	–	3.02
	Wastewater treatment (m^3)	0.79	–	15.83	1.44	–	28.89
	Labor cost	–	–	5.09	–	–	4.46
	Land management fee	–	–	0.01	–	–	0.01
	Maintenance	–	–	35.20	–	–	35.20
	Taxes and insurance	–	–	591.22	–	–	101.22
	Biodiesel (kg)	–	1000.00	(705.00)	–	1000.00	(724.00)
	Glycerin (kg)	–	100.00	(100.00)	–	100.00	(100.00)
	Vernicia oil (kg)	–	2470.00	(4940)	–	–	–

[a]( ) means benefits from products and coproducts.
[b]Initial capital costs that are one-time investment for 30 years.

In both cases, except for the first year, the cultivation was under a rain-fed system with annual urea, phosphorus, and potassium fertilizer input. The amount of mineral fertilizer use was changed due to the application of composts from Hibiscus leaves and Pongamia and Hibiscus-Vernicia oil cake. The management of the cultivation such as tillage, pruning, and harvesting was done manually.

– *Oil extraction*: A three-phase solvent extraction system is used to obtain sugar, medicinal compounds, and oil (Fig. 6.2). This method was recently developed by a research group in Osaka Prefecture University as contracted to the SATREPS Project. Water, methanol (MeOH), and *n*-hexane are applied to extract sugar, vitamin E, and phytosterol and vegetable crude oil, respectively. Several valuable coproducts are derived with high extraction efficiency. Accordingly, it was reported that 90% of vitamin E and phytosterol and 95% of sugar and oil as their contents in the seed were derived. Most of the solvents were recycled (90%), however about 10% of the total used solvents emitted to the air due to their high volatility. The three-phase oil extraction was applied to Pongamia and Vernicia seeds. Nevertheless, as the low component of medical compounds in Hibiscus seed, only sugar and oil extraction were preferred.
– *Biodiesel production*: The two-phase reaction of biodiesel production from vegetable oil is illustrated in Fig. 6.2. Depending on free fatty acid (FFA) value biodiesel can be obtained via one-step/two-step transesterification (Luu et al., 2014; Thanh et al., 2010), which encompasses two main stages including the esterification process and the transesterification process. Firstly, the esterification is performed using the molar ratio of methanol to free fatty acids (FFA) of 6:1, 1 wt% H_2SO_4, 65°C, and cosolvent is 30% (wt/wt) acetonitrile. Secondly, the transesterification process is performed in 30 min with a methanol-to-oil molar ratio of 1:4 and 0.3 wt % potassium hydroxide (KOH), and 10% (wt/wt) acetone as cosolvent. After the reaction, ~90% of acetone and 25% of methanol are recovered. Following the separation of glycerin, the solution is washed and dried. The conversion yield of biodiesel is around 99%, and a total of 93% by mass was obtained from crude oil. Accordingly, the obtained Pongamia, Vernicia, and Hibiscus oils were collected and transferred to a transesterification reactor to produce biodiesel. Since FFA value of Pongamia oil is >5% and that of Vernicia and Hibiscus are <5%, two-step and one-step transesterification were applied to produce Pongamia biodiesel and Hibiscus and Vernicia blended biodiesel, respectively. The current capacity of the biodiesel pilot plant of the SATREPS Project in Viet Nam is 500 tonnes per year and will be upgraded to 1500 tonnes per year.

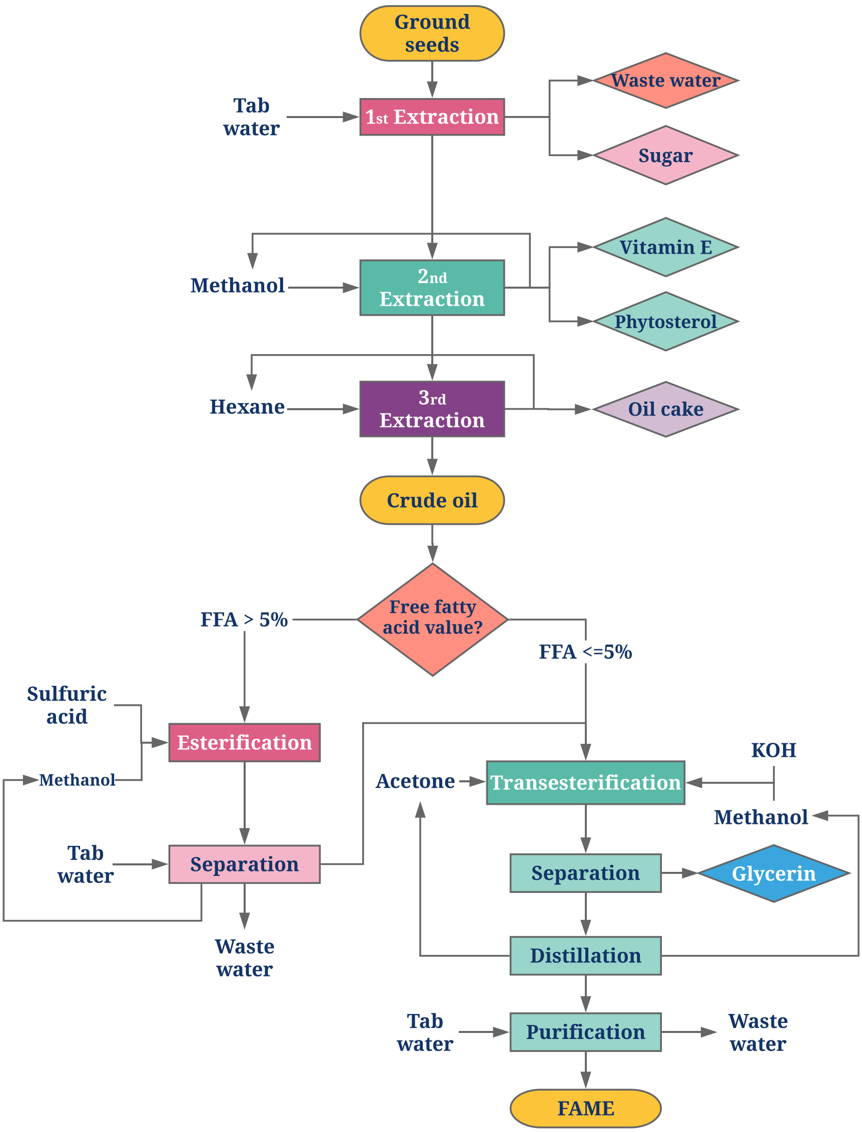

Fig. 6.2 Cascade oil extraction and two-stage reaction of biodiesel production from vegetable oil (SATREPS Project).

– *Blending, distribution, and combustion*: Neat biodiesel (B100) was blended with petrodiesel and distributed in Ha Noi and Quang Ninh Province.
– *Transportation*: The transportation of input materials and output products and coproducts was also analyzed in this study.

4.6 Scenario development

The application of several biodiesel blends was contemplated, including B5, B10, B20, and B100.

4.7 Sensitivity analysis approach

Sensitivity analysis was conducted to determine and estimate which factors influence Triple I and its parameters. Several conditions were considered as follows:

- Fuel price: According to the annual fuel price record and focus of the US EIA (2017a,b) from 2016 January to 2018 June, the lowest price and highest price of fuel were around 20% lower and 15% higher than the current price of fuel, respectively. Therefore two scenarios were made following the 20% decrease and 15% increase in fuel prices.
- Regarding Hibiscus yield: The best case was set based on provided data of some farmers in the north of Viet Nam, the maximum yield of Hibiscus fruits was 10 tonnes per hectare, ~2.2 tonnes of seeds. The worst case was set following the average lowest yield of one tree reported at the SATREPS Project annual meeting (Pham, 2016) in other countries, leading to a 45% decrease in Hibiscus yield.
- Vernicia seed yield: Since the yield of Vernicia strongly depends on the condition of soil and weather and the rate of male and female flowers in the tree (Tran, 1996), two scenarios analyzing a 20% increase and decrease of Vernicia seed yield were made.
- Pongamia seed yield: Two scenarios were made in response to the lowest Pongamia seed yield of 8 kg tree^{-1} in India (Murphy et al., 2012) and the plantation density of 3000 trees ha^{-1} in the pilot area in Quang Ninh Province.
- Many studies claimed that one disadvantage of biodiesel systems was using diesel for transportation of raw materials and distribution of biodiesel. In order to see how it affected the system, Pongamia biodiesel production plant was relocated near its cultivation area that considerably alters the distances of input material transport and biodiesel distribution.
- Since revenues of the newly developed biodiesel system are expected to highly depend on several coproducts, the reduction in price and the existence of those coproducts were also considered in this study.
- The change in conversion factor from economic value to global hectare α was also considered.

5 Results and discussion

5.1 Human health and ecosystem quality impacts

Table 6.9 provides the main findings from LCA and LCC of the four biodiesel systems in Ha Noi and Quang Ninh. Unfortunately, biodiesel systems in Ha Noi showed higher impacts on human health than petrodiesel. The more biodiesel existed in the blend, the more impacts on people's health were observed. In Quang Ninh, nevertheless, the results indicated that the more biodiesel existed in the blend, the fewer impacts on people's health it had. Fig. 6.3 describes the share in total impacts of main processes in the life cycle of the B100 system that can thoroughly explain the different impacts of the two areas. Accordingly, the extraction of seed and the combustion of fuel were the first and the second most significant contributors to human health impacts in Ha Noi and vice versa in Quang Ninh.

Due to the higher concentration of NO_x in the exhaust gas of biodiesel (Table 6.3), the combustion of biodiesel in targeted vehicles enlarged its impacts on human health (Fig. 6.4). Nevertheless, thanks to the better weathering behaviors (Section 4.1), biodiesel blends showed a noticeable reduction in health impacts when used in cruise boats. Therefore only in Quang Ninh, where cruise boats consume more than half of the total annual fuel, biodiesel consumption could promote the mitigation of human health impacts. Moreover, a certain amount of solvent releases in oil extraction phase was another driver of human health impact increase. Since the Hibiscus-Vernicia biodiesel in Ha Noi was supposed to utilize a substantial amount of seed oils, especially Vernicia oil, to produce Hibiscus-Vernicia biodiesel blend and several coproducts, its oil extraction phase presented $\sim$54% of total human health impact in the entire system.

Overall, the total human health impact of the B100 systems in Ha Noi and Quang Ninh were 2.7 times higher and 2.4 times lower than that caused by the petrodiesel system, respectively.

In the context of ecosystem quality, the burden on the ecosystem of biodiesel systems was significantly higher than the petrodiesel system in both areas. Most of the impacts were owing to the application of fertilizers in Hibiscus-Vernicia and Pongamia cultivation (in both areas) and chemicals in oil extraction (in Ha Noi case), which accounted for >80% of tal ecosystem impacts.

Table 6.9 Main results from LCA and LCC

No.	Damage category	B0[a]	B5	B10	B20	B100
1	***Hibiscus-Vernicia biodiesel (Ha Noi)***					
	Human health (DALY $pers^{-1}$)	11,140	12,155	13,124	15,067	30,687
	Ecosystem quality (PDF m^{-2} $year^{-1}$)	8,152,597	145,473,110	268,878,490	516,182,470	2,503,693,000
	Ecological footprint (gha)	20,424	10,853	1169	−18,239	−174,347
	Biocapacity (gha)	N/A	266	533	1067	5362
	Cultivation land occupation (ha)	N/A	2791	5582	11,165	55,825
	Capital cost[b] ($)	N/A	220,539	441,079	882,158	4,410,799
	Operating and maintenance cost ($)	16,986,725	30,937,498	43,722,915	66,804,138	275,087,470
	Benefit ($)	N/A	32,434,525	47,924,751	78,943,108	328,187,160
	Revenue ($)	N/A	1,497,027	4,201,836	12,138,970	53,099,690
	Payback period (year)	N/A	4	3	2	2
2	***Pongamia biodiesel (Quang Ninh)***					
	Human health (DALY $pers^{-1}$)	62,145	60,750	52,058	49,170	26,174
	Ecosystem quality (PDF m^{-2} $year^{-1}$)	15,996,340	26,467,689	32,361,898	50,660,610	197,108,460
	Ecological footprint (gha)	39,023	35,843	20,248	15,268	−24,615
	Biocapacity (gha)	N/A	59	119	237	1186
	Cultivation land occupation (ha)	N/A	561	1122	2244	11,218
	Capital cost[b] ($)	N/A	86,344	172,733	345,388	1,726,942
	Operating and maintenance cost ($)	N/A	22,665,148	22,799,787	24,527,789	38,371,975
	Benefit ($)	N/A	22,357,635	23,853,115	26,760,909	50,076,364
	Revenue ($)	N/A	−307,513	1,053,328	2,233,120	11,704,389
	Payback period (year)	N/A	0	5	5	4

[a]N/A: Data which were not considered in this study.
[b]Average capital cost allocated for 30 years of project lifetime with the discount rate of 3.74%.

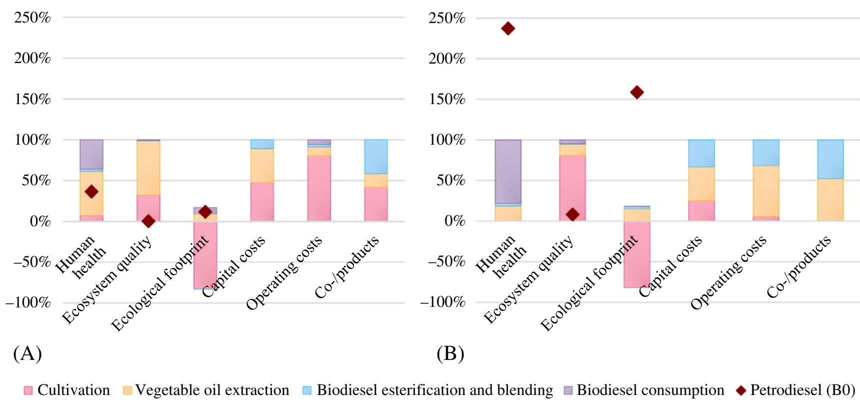

Fig. 6.3 Contribution of unit processes in total life cycle impacts of B100 compared to petrodiesel systems: (A) Hibiscus-Vernicia biodiesel in Ha Noi and (B) Pongamia biodiesel in Quang Ninh.

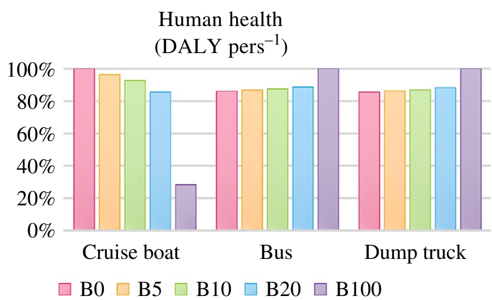

Fig. 6.4 Exhaust gas impacts on human health of different biodiesel blends use in several vehicles.

5.2 Net carbon dioxide emissions

The level of carbon dioxide uptake by standing Pongamia and Vernicia trees was higher than the total carbon dioxide emitted from various activities in the entire life cycle of the biodiesel system (Fig. 6.3). Consequently, minus values of the ecological footprint in B20 (only Hibiscus-Vernicia biodiesel) and B100 systems and considerable reduction in the outcome of ecological footprint in other lower biodiesel systems comparing to the petrodiesel system were obtained (Table 6.9).

5.3 Biocapacity

In terms of required plantation area to provide enough feedstock for annual biodiesel production, the results recorded the gain in biocapacity in all biodiesel systems. Available land in designated areas could cover all the requirement of the cultivation land for feedstock production of all biodiesel blends up to B100 (Tables 6.1 and 6.9).

Principally, use of biodiesel systems of all blends led to considerable reduction in ecological footprint compared with the petrodiesel system.

5.4 Economic evaluation

The estimation of costs and benefits in various biodiesel blend systems was conducted following the LCC method. Results indicated the highest share of feedstock cultivation and vegetable oil extraction plant in total capital cost and annual investment in Ha Noi and Quang Ninh, respectively (Table 6.9 and Fig. 6.3). This situation was mostly due to a considerable amount of oilseeds required to fulfill the need for crude vegetable oil used to produce a certain amount of biodiesel in both areas. Consequently, first-year seed oil preparation, plant construction and installation as capital costs, and chemical substances used for cultivation and oil extraction as other operating costs were the highest contributors to the total costs of biodiesel systems. As an extensive area of feedstock cultivation was required to produce 1 tonne of Hibiscus-Vernicia biodiesel in Ha Noi that ensuring the proper ratio between the two fuels, the investment in Hibiscus-Vernicia intercropping was 48% and 80% of the total capital cost and operating cost of the system, respectively. In Quang Ninh, most of the investment was in oil extraction stage. Biodiesel production and consumption stages came next in annual payment for the biodiesel system (Fig. 6.3). On the other hand, a noticeably high revenue of the biodiesel systems was observed, except for the B5 system in Quang Ninh (Table 6.9). Biodiesel systems of higher blends and Ha Noi biodiesel system compared to Quang Ninh system showed higher revenue and a shorter payback period for all capital costs. This outcome asserted the economic efficiency of all biodiesel systems. However, although biodiesel was the main product of this system, it shared only around 5% and 30% of total system benefits in Ha Noi and Quang Ninh, respectively (Fig. 6.5). Biodiesel systems in Ha Noi highly depended on several coproducts, including Hibiscus calyces in the agricultural operation, and glycerin, sugar, vitamin E, phytosterol, and residue Vernicia oil in the oil extraction and biodiesel production stages (Fig. 6.5).

Closely examining the profit of biodiesel and its direct coproduct (glycerin) in the production stage, their market prices could cover all spending for production only in Quang Ninh system. Biodiesel systems in Ha Noi had to deal with a longer distance of raw material transport and biodiesel

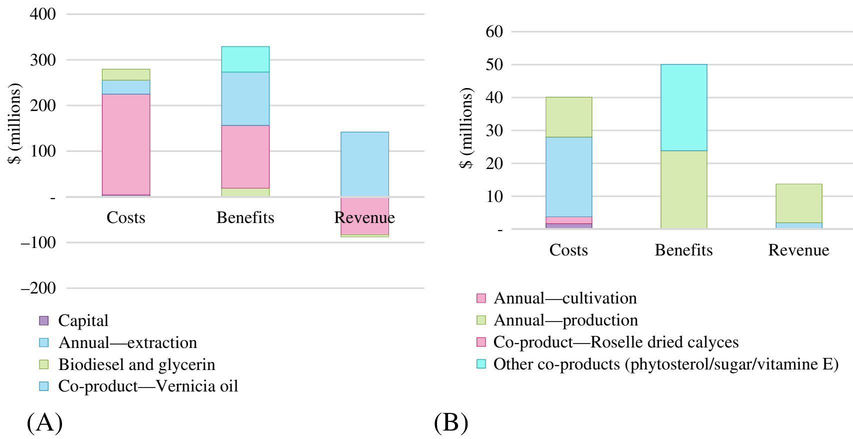

Fig. 6.5 Distribution of different components in average total cost and benefit per year of neat biodiesel system: (A) Hibiscus-Vernicia biodiesel in Ha Noi and (B) Pongamia biodiesel in Quang Ninh.

distribution thus no revenue was reported in biodiesel production stage. Fortunately, together with the oil extraction system, the extended biodiesel production plant, starting from vegetable crude oil extraction to final biodiesel production, could result in substantial revenue of the entire biodiesel system in Ha Noi, even four times higher than in Quang Ninh (Fig. 6.5).

In consideration of the entire biodiesel system, attention should be paid to the difference in revenue between agricultural and biodiesel production stages. Since these two stages both play important roles in current and future development of every biodiesel system, an equal revenue distribution of all biodiesel life cycle chains needs to be taken into account. Either mutual cooperation between agricultural practitioners and biodiesel production practitioners, or the formation of cooperation that can operate the main chains in the biodiesel life cycle from feedstock cultivation to biodiesel production could be possible solutions.

In this study, the results on impacts of biodiesel life cycle systems on human health, ecosystem quality, and net carbon dioxide and their economic incentives are in accordance with previous studies in life cycle assessment of various biodiesel systems. For instance, Achten (2010) noted that due to fertilizer use, jatropha and palm plantation had a high burden on ecosystem quality in terms of increasing terrestrial acidification and eutrophication. Moreover, the decrease in global warming potential given by net CO_2 reduction was also recorded. The study also confirmed the profitmaking potential of marginal land conversion in biodiesel feedstock cultivation.

Another study conducted by CheHafizan and Noor (2013) likewise indicated that under an LCA perspective, biodiesel had more preferable environmental behaviors than petrodiesel as it could lower abiotic depletion potential, global warming potential, and ozone depletion potential due to reduction of crude oil use, the increase in CO_2 absorption in the agricultural stage, and the lowering of emissions from crude oil extraction and refining, respectively. Their paper also denoted the higher acidification and eutrophication potential due to fertilizer utilization, and higher photochemical oxidation potential due to the higher concentration of NO_x in combustion emissions of biodiesel compared to petrodiesel systems.

5.5 Triple I

Fig. 6.6 presents values of Triple I and its parameters after applying conversion factors in four biodiesel systems: B5, B10, B20, and B100 in Ha Noi (a) and Quang Ninh (b). In Ha Noi, although neat biodiesel (B100) had the highest impact on human health and ecosystem quality, it supported a significant decrease in ecological footprint. Moreover, the revenue from the B100 system was also the highest. Impacts of B100 in Quang Ninh were similar to that in Ha Noi except for the human health impact. This system could reduce nearly 60% of human health impact compared to petroleum system.

When the value of Triple I is less than zero, the studied system is identified as a sustainable system. As shown in Fig. 6.6, out of the four blends in both areas, only the B100 system in Ha Noi proved its potential sustainability. Other systems could not reach the sustainable level since their reduction of the ecological footprint was particularly low and less revenue was obtained.

Biodiesel systems have been proposed to combat natural resource depletion and allow reduction of petrodiesel use. The existence of biodiesel was not expected to raise the total fuel consumption or to form a new fuel system in parallel with the petroleum system. Hence, as an alternative source, biodiesel should be considered under the business-as-usual scenario (with neat petrodiesel use) to delineate what human beings and the environment could gain from implementing this system. Therefore different influences between the current petrodiesel system and the execution of the biodiesel system were incorporated into Triple I and its parameters, from now on referred to as avoided scenarios. Fig. 6.6 shows that when biodiesel and its blends were used instead of neat petrodiesel, all biodiesel systems are sustainable.

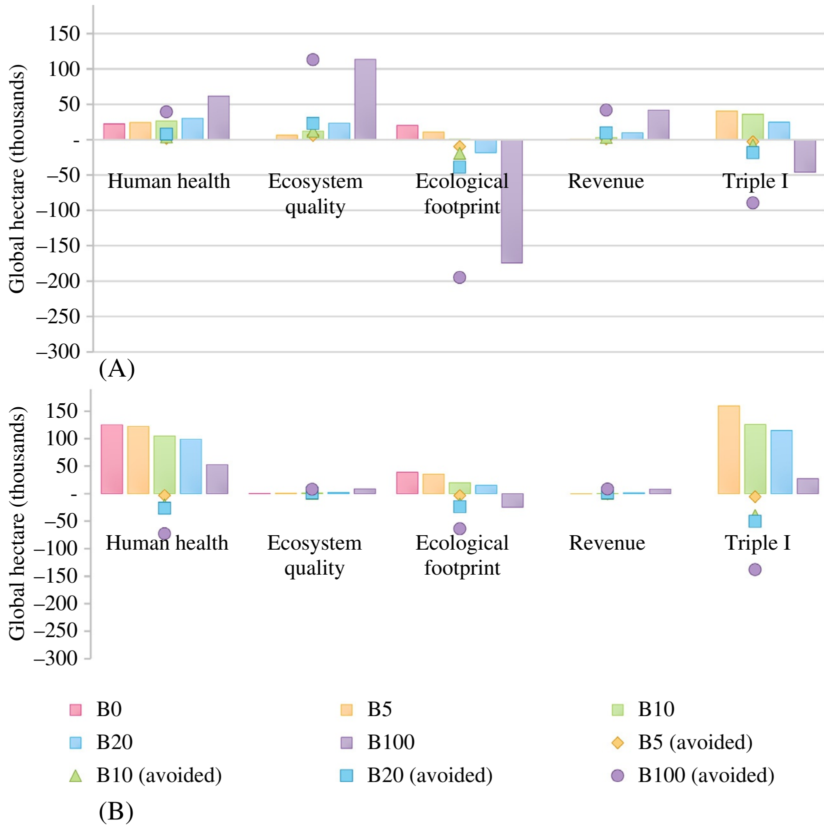

Fig. 6.6 Sustainability evaluation and total life cycle impacts of different biodiesel blends by global hectare: (A) Hibiscus-Vernicia biodiesel in Ha Noi and (B) Pongamia biodiesel in Quang Ninh.

The findings were interesting that although Ha Noi biodiesel system had higher sustainable potential in a stand-alone system, as an alternative to petroleum Quang Ninh biodiesel system became more sustainable than Ha Noi biodiesel system.

5.6 Sensitivity analyses

Triple I and its parameters were subjected to different conditions (Figs. 6.7 and 6.8) to examine which factors could affect the system. Different analyzed cases were as follows:

– Cases 1 and 2: 20% decrease and 15% increase in fuel price, respectively.
– Cases 3 and 4: Hibiscus seed yield decreased by 45% and increased to 2.2 tonnes/ha, respectively.

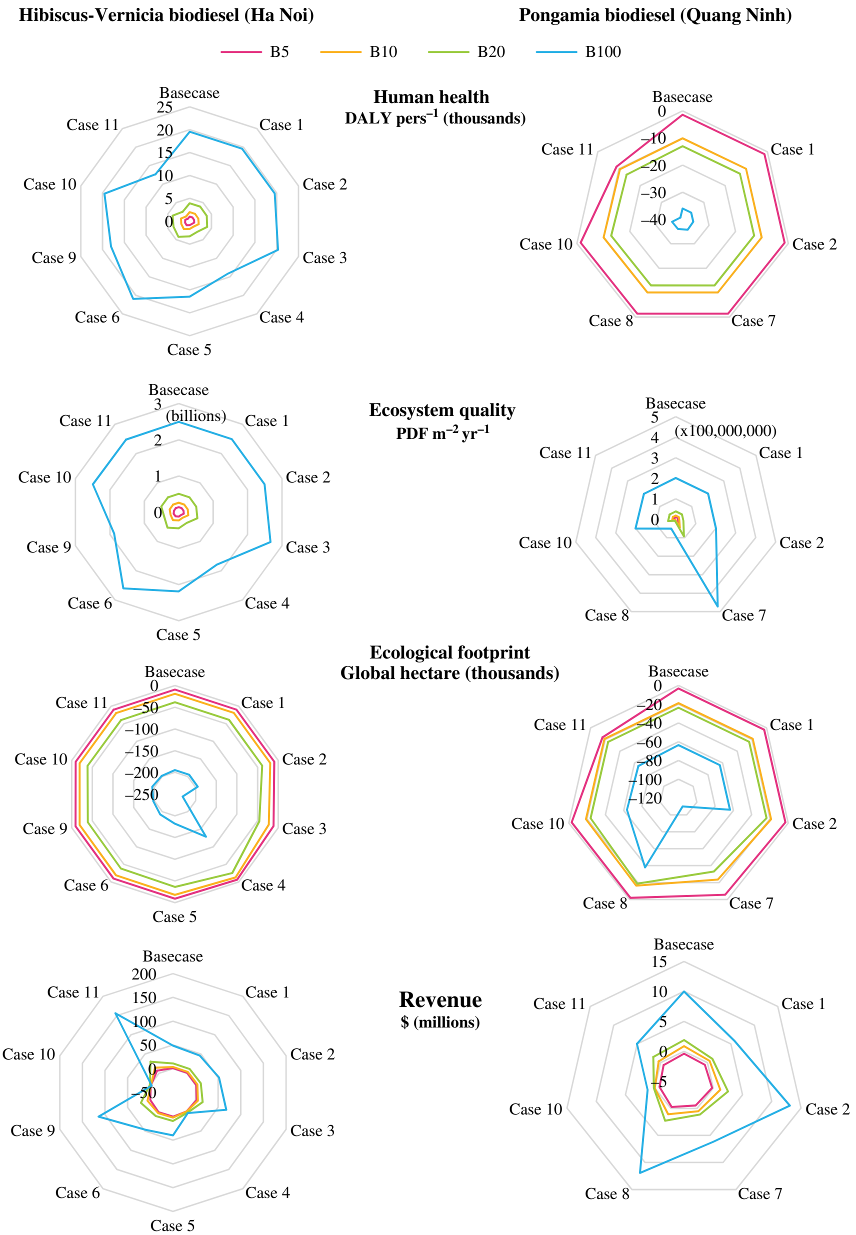

Fig. 6.7 Sensitive analysis of factors affecting Triple I parameters.

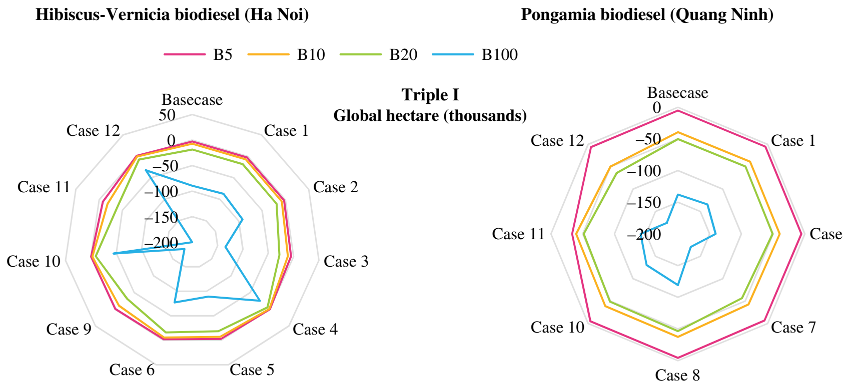

Fig. 6.8 Sensitive analysis of factors affecting the sustainability of biodiesel systems.

– Cases 5 and 6: Vernicia seed yield decreased and increased by 20%, respectively.
– Cases 7 and 8: Pongamia seed yield decrease to 8 kg tree^{-1} and plantation density increase to 3000 trees ha^{-1} respectively.
– Case 9: Pongamia biodiesel plant was placed near to cultivation area (upland provinces).
– Case 10: 30% decrease in price of all coproducts.
– Case 11: Only vegetable oils were directly extracted from seeds without other coproducts (sugar and medicinal compounds).
– Case 12: Ratio between EF and GDF of Viet Nam was changed to the average value of 10 years from 2005 to 2014, which resulted in a 151% increase in the conversion factor α

In the first 11 cases, responses of Triple I parameters were varied. Generally, the B100 system was the most sensitive to different influences. Fig. 6.7 showed that the decrease and increase of fuel price (cases 1 and 2) did not affect much on the revenue of biodiesel systems in Ha Noi since it highly depended on other coproducts. The change in fuel price significantly alters the revenue of biodiesel systems in Quang Ninh.

The most affected and controversial factor to the entire system in Ha Noi biodiesel was the yield of Hibiscus (case 3 and case 4). With the decrease and increase of Hibiscus yield, significant fluctuation of all parameters was observed. The reduction of Hibiscus seed yield resulted in more impacts on human health and ecosystem quality. Meanwhile, it lowered ecological footprint and increased net revenue. That was because Hibiscus biodiesel shares the higher part in the Hibiscus-Vernicia biodiesel mixture when yield decreases, and the cultivation area needs to be extended to fulfill the annual fuel demand. As discussed in Section 5.1, agriculture practice and oil extraction processes were the highest and the second highest contributors to ecosystem quality and human health impacts, respectively. Hence, the more area used for propagation, the more impacts on ecosystem quality. Moreover, if the same intercropping system is applied to an extended area, the increase of Vernicia seeds would raise the amount of solvent used for oil extraction, further impacting human health. Additionally, the expansion of the oil crop cultivation area could mean cropland conversion and food crop conflict. This system would likewise only produce more coproducts without any efficiency of fuel production. Similar to the change in Hibiscus yield, the decrease of Pongamia seed yield also increase biodiesel system impacts on ecosystem quality in Quang Ninh and vice versa.

On the other hand, the revenue of all biodiesel systems was noticeably affected by the change in Hibiscus yield, Vernicia yield, Pongamia yield, and the price of coproducts. Furthermore, when biodiesel production plant was relocated to upland provinces (case 9), near the cultivation areas, a considerable growth of revenue was observed, more than threefold in Ha Noi B100 system, due to the reduction of seed transportation and labor costs. Concerning no coproducts obtained from oil extraction scenario (case 11), hereinafter referred to as oil only biodiesel system, this assumption resulted in a noticeable decrease in human health impacts of both Hanoi and Quang Ninh biodiesel systems. Nevertheless, there was a different trend of system revenues between Ha Noi biodiesel and Quang Ninh biodiesel whereas a threefold increase in Hibiscus-Vernicia biodiesel and a twofold decrease in Pongamia biodiesel were reported. Since the relevant chemical compositions in Pongamia seed is high (Table 6.1), an appropriate volume of coproducts was obtained that could cover all costs of the production. However, those coproducts from Hibiscus and Vernicia seeds were not enough to pay off the investment. Hence, the oil only biodiesel system could make higher revenue in case of Hibiscus-Vernicia biodiesel.

Regarding the results of Triple I (Fig. 6.8), as alternatives to petrodiesel, the use of biodiesel of all blends from B5 resulted in a sustainable value, regardless of various fluctuating conditions. However, the sustainability of B5 systems was right on the edge between sustainable and unsustainable. The B5 system in Ha Noi became almost unsustainable when coproducts' price decreases, and even turned into unsustainable when Hibiscus yield increased. This response revealed that B5 systems were unstable and would be in the danger zone when extreme conditions occurred. Furthermore, the oil only biodiesel system substantially improves the sustainable potential of Ha Noi biodiesel system. Indeed it made the sustainability of Ha Noi biodiesel system became higher than Quang Ninh system in the base case due to the considerable decrease in human health impact and remarkable increase in revenue.

As discussed in Section 3.1, the ratio (α) between ecological footprint and GDP represents how effectively natural resources were employed for economic development in a country. A lower value means the more proper use of those resources and vice versa. Fig. 6.8 (case 12) demonstrates how the rise of this ratio influenced the sustainability of biodiesel system. A remarkable increase in Triple I value of Ha Noi system and decrease in Triple value of Quang Ninh system were discovered. This finding revealed that although the two systems were sustainable, Ha Noi biodiesel system associated with the higher economic development and Quang Ninh biodiesel system came with the better quality of the environment.

5.7 Social issues

The critical social impacts of the biodiesel systems were that the development and application of the nonedible vegetable oil-derived biodiesel system could engender various merits to the society in Northern Viet Nam. This contribution can be placed on, for instance, local community and workers as job creation and favorable working conditions, and the society as economic enhancement, technology development, and conflict prevention. Table 6.10 introduces some main contributions of different biodiesel blend systems to the local development in the North of Viet Nam.

According to the General Statistical Office of Viet Nam (2018), as of 2016, the number of unemployed in Quang Ninh, Ha Noi, and other high mountainous areas were nearly 35,000 persons, 141,915 persons, and 124,810 persons, respectively. When the complete life cycle of biodiesel is installed in Northern Viet Nam, the engagement of local employees will be encouraged. Therefore this system can support this area in reducing unemployment. The number of workers in biodiesel systems increased for B5, B10, B20, and B100 systems (Table 6.10), leading to the total of 0.92%, 1.85%, 3.70%, and 18.47% reductions in the unemployment rate, respectively. Consequently, added labor income together with annual revenue from biodiesel can considerably contribute to the economic development of those areas, especially mountainous provinces which have the highest poverty rate in Viet Nam (GSO VN, 2018). A preliminary

Table 6.10 Estimation of annual contribution of biodiesel system to local community in Northern Viet Nam

No.	Social aspect	B5	B10	B20	B100
1.	***Pongamia biodiesel in Quang Ninh***				
	Annual biodiesel production (tonnes)	1447	2894	5789	28,941
	Employee in (persons)[a]	121	242	485	2425
	Unemployment reduction	0.36%	0.71%	1.42%	7.12%
	Added labor income ($)[b]	271,598	543,195	1,086,624	5,431,950
2.	***Hibiscus-Vernicia biodiesel in Ha Noi***				
	Annual biodiesel production (tonnes)	1183	2366	4732	23,661
	Employee in mountainous provinces (persons)[a]	692	1384	2769	13,843
	Unemployment reduction in mountainous provinces	0.55%	1.11%	2.22%	11.09%
	Added labor income in mountainous provinces ($)[b]	1,370,105	2,740,210	5,480,420	27,402,100
	Employee in Ha Noi (persons)[a]	18.81	37.62	75.25	376.23
	Unemployment reduction in Ha Noi	0.01%	0.03%	0.05%	0.27%
	Added labor income in Ha Noi ($)[b]	47,806	95,611	191,222	956,112

[a]Full-time employment with working time of 8 h per day and 300 days per year.
[b]Calculated based on regional minimum wage rates applied to employees working under employment contracts of Viet Nam, in which the minimum wage rates per month of Quang Ninh, Ha Noi, and other high mountainous areas are ~$150, $170, and $132, respectively.

estimation illustrates the higher economic contributions of higher blend biodiesel systems (Table 6.10). In addition to economic benefits, reducing the unemployment rate can also diminish other related issues, such as crime and physical and mental health problems.

Since this biodiesel system is a legitimate system for applying to Northern Viet Nam, its establishment, management, and operation have to abide by Viet Nam Labor Law and other related regulations. Thus there is to be non-infringement with respect to child labor, unfair practices, and discrimination in salary and gender, and labor rights must be upheld.

6 Conclusions and recommendations

The sustainability of the entire biodiesel life cycle system in Northern Viet Nam was evaluated under a life cycle sustainability assessment index—Triple I. Accordingly, the application of four biodiesel blends—B5, B10, B20, and B100 in Ha Noi and Quang Ninh Province were investigated. Subsequently, those systems were also placed under various changes in, for instance, crop yield, fuel price, distribution distance, coproduct prices, and biodiesel production technology.

Results of Triple I could be used to propose feasible options and implications for biodiesel policies toward sustainable development as follows:

Implications for sustainable biodiesel policy in Northern Viet Nam

Integrating Triple I results in the context of Northern Viet Nam, the sustainability of the biodiesel system will occur if the implementation complies with the following principles:

– The cultivation area should not exceed the total area of open-pit mining and mining dumpsites, in order to prevent the land occupation side effect of oil crop cultivation. This issue is supported by the suggestion from Fargione et al. (2008) that the cultivation of oil crops should be placed on marginal land to reduce carbon footprint and avoid conflicts with food crops and food security. Hence, the employment of Pongamia in Quang Ninh Province and Hibiscus and Vernicia intercropping in high mountainous areas near Viet Nam and China border, to produce oils as feedstocks for biodiesel production in Northern Viet Nam is an ideal option for both environment recovery, biodiesel feedstock acquisition, local economic development, and poverty reduction.

– Moreover, data on the land use of each biodiesel system indicates that within current seed yield, the biodiesel system in Northern Viet Nam can provide up to blend B100.
– Among the main stages in the entire biodiesel life cycle, the equal distribution of economic profits between feedstock cultivation and biodiesel production is fundamental for the practical application and future development of the system. This issue raised the need for either the mutual cooperation between oil crop cultivation practitioners and biodiesel production practitioners or a corporation that can organize the two stages.
– As an alternative to petrodiesel, biodiesel systems of all the studied blends, including B5, B10, B20, and B100, have demonstrated their promising potential of enviro-economic benefited renewable sources. However, since the sustainability determination of the higher blends is higher than the lower ones, the application of higher biodiesel blends is recommended under the particular circumstance.
– The replacement of petrodiesel with biodiesel has contributed to the significant enhancement of human health. Biodiesel systems, nevertheless, have led to the diminution of ecosystem quality. As the proportion of biodiesel in the blend increases, its pros as well as cons also increase. It is noteworthy that although both biodiesel systems in Ha Noi and Quang Ninh are sustainable, Ha Noi biodiesel system associated with the higher economic development and Quang Ninh biodiesel system came with the better quality of the environment.

Overall, an appropriate biodiesel system has to be neutral and balanced among the three pillars of sustainable development: economic impacts, environmental impacts, and social impacts (Elkington, 1998; World Commission on Environment and Development, 1987). Taking into account all the discussed factors and considering the stability of the biodiesel system, the higher blends of biodiesel are preferred. Furthermore, since the application of up to B20 does not require diesel engine modification and has similar engine performance to neat petrodiesel (No, 2011), the installation of B20 in the North of Viet Nam is highly recommended.

Complying with the Energy Development Scheme of 5% biofuel in Viet Nam, the B5 system should be implemented. To be more precise, the B5 development scheme needs to focus on the selection of high-yield feedstocks with the qualified biodiesel, as well as the location of biodiesel production plants in order to minimize fuel distribution distance. Accordingly, findings from this study suggest that to maximize the sustainable potential of biodiesel system in Northern Viet Nam, the utilization of Pongamia biodiesel should follow the full biodiesel production system developed under SATREPS Project, while Hibiscus-Vernicia biodiesel should adhere to the oil only biodiesel production system that omits the coproducts in oil extraction phase.

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CHAPTER 7

A life cycle assessment of tri-generation from biomass waste: The experience of the “agro-combined” of Thibar

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1 Introduction

The transition toward a resource-efficient and sustainable economy is a global challenge aimed at facing environmental problems such as the depletion of natural resources (including fossil fuels), the increase of environmental pressures, and the climate change (Cellura et al., 2013).

The production of energy from renewable sources is a key strategy to ensure the transition to an eco-innovative and low-carbon economy (Beccali et al., 2009). Furthermore, the use of renewable sources for the generation of electricity and useful heating and cooling can bring social and economic benefits, such as creation of new jobs, potential lowering of the price of oil because of lower demand, reduction of energy costs, saving/revenue from the sale of self-generated energy (Bordoni et al., 2010).

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Within the renewable energy sources, bio-energy arising from biomass represents the largest one, accounting for roughly 10% of the world's total primary energy supply (IEA—International Energy Agency, 2012).

It provides an alternative to fossil-based energy (Iakovou et al., 2010) and can contribute to the bio-economy and the circular economy (European Commission, 2015).

The use of biomass is expected to increase in the next decades, also due to the population growth (Cellura et al., 2014). In detail, the technical potential for biomass is estimated to be possibly as high as 1500 EJ/year by 2050, although most biomass supply scenarios that take into account sustainability constraints indicate a potential of between 200 and 500 EJ/year (excluding aquatic biomass). Forestry and agricultural residues and other organic wastes would provide between 50 and 150 EJ/year, while the remainder would come from energy crops, surplus forest growth, and increased agricultural productivity (WEC—World Energy Council, 2013). The quantification and characterization of organic waste (including food waste and residuals from livestock) along the food supply chain have been proved to be crucial for the identification of potential applications for waste valorization, reducing the overall impacts in a life cycle perspective and encouraging the production of useful coproducts (Corrado et al., 2016).

Non-OECD countries are the main users of biomass. In these countries, traditional biomass (including wood, charcoal, agricultural residues, and animal dung) is still the main source of energy used for cooking, water and space heating. However, often the biomass comes from unsustainable sources, leading to deforestation and soil degradation, and it is burned in very low efficient stoves. Thus the use of biomass can potentially create environmental and health problems unless more efficient stoves and fuels (biogas, ethanol) are deployed.

Thus the creation of a sustainable energy supply chain from biomass, and in particular from biomass wastes, is important to manage the potential environmental issues related to its use, especially in those countries where it is one of the main sources of energy.

Tunisia is one of the few developing countries that developed a proactive policy for the promotion of renewable energy and energy efficiency already in 1985 (GIZ—Deutsche Gesellschaft für Internationale Zusammenarbeit, 2012). However, renewable energy plays a minor role in the energy supply: in 2014 the percentage of renewable energy in the Tunisian electricity mix was evaluated at less than 2% of annual national electricity production

(GIZ—Deutsche Gesellschaft für Internationale Zusammenarbeit, 2014; GIZ—Deutsche Gesellschaft für Internationale Zusammenarbeit, 2015).

In order to strengthen its national policy on sustainable energy from biomass, Tunisia implemented different programs and projects (GIZ—Deutsche Gesellschaft für Internationale Zusammenarbeit, 2012; Reegle, 2012), showing the interest of this country in the exploitation of biomass and waste for energy purposes:

– The four-year program for the 2008–11 period: it included investments for the installation of electricity generation facilities with a capacity of 40 MW using olive residues and with a capacity of 40 MW using household waste;
– Projects are in place to disseminate more efficient biomass stoves and to exploit the rural/agro-business production of biomass;
– A pilot project involving gasification through poultry waste has been launched;
– A 10 MW waste-to-electricity project at the Jebel Chakir landfill has been implemented;
– The 2010 Energy Efficiency and Biomass Project, in collaboration with the World Bank, aimed to develop biomass energy sources as an alternative to fossil fuels in the country.

In Tunisia, there is a significant potential to expand biomass use by tapping the large volumes of unused agricultural and livestock wastes. Domestic production of organic waste was estimated at about 6 million tonnes/year in 2009: 2.2 million tonnes of household waste, 2.2 million tonnes from farms and agroindustry, 1 million tonnes from olive oil processing, 400,000 tonnes from poultry droppings, and 200,000 tonnes from wastewater treatment (Reegle, 2012).

The waste biomass can be used, in particular, for the production of biogas, playing a significant role in reducing environmental problems in Tunisia, and leading to several environmental benefits, for example, reduction of waste disposal, saving of fossil fuels, increase of renewable resources exploitation rate, reduction of greenhouse gas emissions.

Considering the potential of the waste biomass use for energy purposes, Tunisia participated in the Cross-Border Cooperation Italy-Tunisia project entitled "Valorisation ènergètique Des Rèsidues—VEDER," started in February 2013 and finished in September 2016. Such a project was developed with the aim to create a pilot plant in the agricultural district of Thibar (Tunisia) for the production of biogas from agricultural and livestock waste and its successive use in a small size tri-generation system, using high efficiency and low impact technologies.

The overall objective of the project VEDER was to make stakeholders and policy makers aware of the effective energy and environmental benefits arising from the use of biomass wastes for electricity and heat generation.

In the above context, this paper presents a Life Cycle Assessment (LCA) to evaluate the life cycle energy and environmental impacts of energy produced by biogas obtained by biomass waste and fueled in a small size tri-generation plant (producing electricity, thermal, and refrigeration energy) with a useful life of 30 years (8000 h per year).

The analysis included the collection of waste due to agriculture and livestock in the agricultural district of Thibar, the use of waste in a bio-digestion plant for biogas production, and the energy tri-generation from biogas combustion.

The results of the analysis allowed at identifying the hot spots of the supply chain that goes from the biomass waste to the energy production to be taken into account for more sustainable energy production processes.

This study deals with a subject often neglected in developing countries. In fact, the attention to the environment in these countries has been slow if compared to that in the industrialized countries, due to the lack of appropriate infrastructure, unsound policies, and ineffective environmental regulations as well as financial and human resources constraints. Furthermore, environmental issues in developing countries do not always represent a high priority due to competing priorities such as poverty alleviation, rapid economic development, and resolution of internal and external conflicts (Massoud et al., 2010).

Each component of the tri-generation plant has a small size (cogeneration plant: 50 kW_e; absorption chiller: 25 kW_t) and this allows for the creation of a small energy district that can be easily replicated in other Tunisian and Mediterranean areas in the context of the distributed generation. Thus the presented study can be considered as a basis for other similar contexts for transforming the threat of waste in an economic, social, and environmental opportunity. Furthermore, it can contribute to the general topic of sustainable production and consumption, giving some information for a cleaner procurement and management of biomass waste for energy production, in order to achieve a bio-economy based on energy from renewable sources. The experience is coherent with the principles of the circular economy: it allows for reducing wastes disposed in landfill, for producing energy from renewable energy sources (biomass wastes), and for recovering useful heat from electricity production.

2 The LCA applied to the "agro-combined" system of Thibar

2.1 Goal and scope definition

The analysis was developed by applying the LCA methodology, according to the international standards of the ISO 14040 series (ISO 14040, 2006; ISO 14044, 2006), for assessing the eco-profile of energy production by biogas in a tri-generation plant.

The research was carried out in an agricultural district, which has been named by the local authorities as the "agro-combined" system of Thibar.

Thibar is a town with 3500 inhabitants located in northern Tunisia, 50 km west of the city of Beja. The Delegation of Thibar has over 11,000 inhabitants, and administratively it is attached to Beja.

The agricultural area of Thibar is owned by the central State and managed by the Office for Public Lands. The above area covers 2704 ha of which 1061 ha are uncultivated (828.5 ha are forests and 161 ha are pastures). There are various agricultural and livestock activities, such as fruit growing, olive growing, viticulture field crops, sheep and cattle breeding. The cattle are made from 336 dairy cows, 190 heifers, 171 calves, 350 sheep. The agricultural practices are both extensive and intensive. Water for irrigation comes from a hill lake located in the heart of the area and from wells and surface water sources. Water is drawn and collected in a basin through proper water pumping stations. The distribution of irrigation water is made via a network of irrigation. The agricultural and livestock products are mainly cereals, olives, fruits, milk, and calves. Furthermore, wine (350,000 bottles per year) and spirits (80,000 bottles per year) are produced in the agricultural area of Thibar.

2.1.1 Functional unit, system boundaries, impact assessment methodologies, and impact categories

The functional unit defines the quantification of the identified function of the system and provides a reference to which the inputs and outputs are related (ISO 14040, 2006).

The function of the examined supply chain is the tri-generation of electricity, thermal energy for heat, and refrigeration energy. According to Masoni and Zamagni (2011), to measure the performance of a system with a single parameter when both electricity and thermal energy are valuable products and to take into account the different quality of these forms of energy, the functional unit can be expressed in MJ of exergy.

For a given system that does not experience mass flow, exergy can be defined as the maximum amount of work that can be extracted reversibly from an energy flow with respect to a dead state (Beccali et al., 2003; Gokcen and Reddy, 1996).

In this study, the selected functional unit is 1 MJ of exergy and the energy and environmental impacts of the examined supply chain are referred to this functional unit considering the exergy production during the whole life cycle of the plant, calculated as described in the next section.

The selection of the system boundaries is based on the "zero burden" assumption: the production of biomass waste (agricultural waste and animal dejections) in input to the plant does not cause any energy or environmental impact. In detail, the system boundaries include

– The production of the plant components, including the raw materials, energy sources, and materials supply.
– The operation of the plant, including the transport of the biomass waste to the plant, the biogas production, the electricity and thermal/refrigeration energy production from the biogas combustion, and the end of life of the digestate.
– The end of life of the plant that includes the recycling of steel, aluminum, and plastics and the landfilling of the other components.

The energy and environmental impact categories selected to calculate the performance of the investigated functional unit are presented in Table 7.1.

Table 7.1 Energy and environmental impact categories

Impact category	Acronym	Unit of measure
Cumulative energy demand	CED	MJ
Acidification potential	AP	Mole of $H+_{eq}$
Ecotoxicity for aquatic fresh water	E_{FW}	CTUe
Freshwater eutrophication	FE	kg P_{eq}
Human toxicity cancer effects	HT_c	CTUh
Human toxicity noncancer effects	HT_{nc}	CTUh
Ionizing radiation	IR	kg $U235_{eq}$
Global warming potential	GWP	kg CO_{2eq}
Marine eutrophication	ME	kg N_{eq}
Terrestrial eutrophication	TE	Mole of N_{eq}
Ozone depletion	OD	kg CFC_{11eq}
Particulate matter	PM	kg $PM2.5_{eq}$
Photochemical ozone formation	POF	kg NMVOC
Resource depletion, fossil, and mineral	RD	kg Sb_{eq}
Total freshwater consumption	TFC	UBP

Energy factors refer to the Cumulative Energy Demand (Frischknecht et al., 2007a; Prè Product Ecology Consultants, 2012) method that enables the estimation of the consumption of renewable (biomass, wind, solar, geothermal, water) and nonrenewable (fossil, nuclear) energy sources. The environmental characterization factors refer to the ILCD 2001 Midpoint (European Commission—Joint Research Centre—Institute for Environment and Sustainability, 2012) impact assessment method, according to the recommendations of the ILCD Handbook of the European Commission (European Commission, DG Joint Research Centre, Institute for Environment and Sustainability, 2011).

2.2 Life cycle inventory

2.2.1 The tri-generation plant

The examined plant (Fig. 7.1) produces biogas from the anaerobic digestion of agricultural waste and manure coming from the "agro-combined" of Thibar. The biogas feeds an internal combustion engine that generates electricity and thermal energy for heating and cooling.

The biomass waste is transported to the plant with trucks for a distance of 5 km (1). It is temporarily stored in a tank (2) and then goes to the anaerobic digester (3), where the organic matter is decomposed by anaerobic microorganisms producing biogas. The biogas feeds a tri-generation plant for energy production. In detail, the plant produces electricity, which is partially used for the auxiliary consumptions and partially to feed the grid, and thermal energy, used for heating (6) and cooling (7) production. The digestate produced during the process is stored (4) before being disposed or further treated for agricultural purposes (5) (distance from the plant to the final treatments: 100 km).

The plant manages 5900 t/year of waste biomass and produces 193,680 Nm^3/year of biogas (340 Nm^3 per ton of biomass) with a low calorific value of 5.9 kWh/Nm^3.

During the operation the tri-generation system (electrical efficiency: 37%) produces 422,800 kWh of electricity[1] and 571,356 kWh of thermal energy, of which half is used for the heating production and half by the absorption chiller (energy efficiency ratio: 0.9), together with 923.33 m^3 of water, to generate 257,119 kWh of cooling.

[1] The 10% of the electricity produced is used for the auxiliary consumptions.

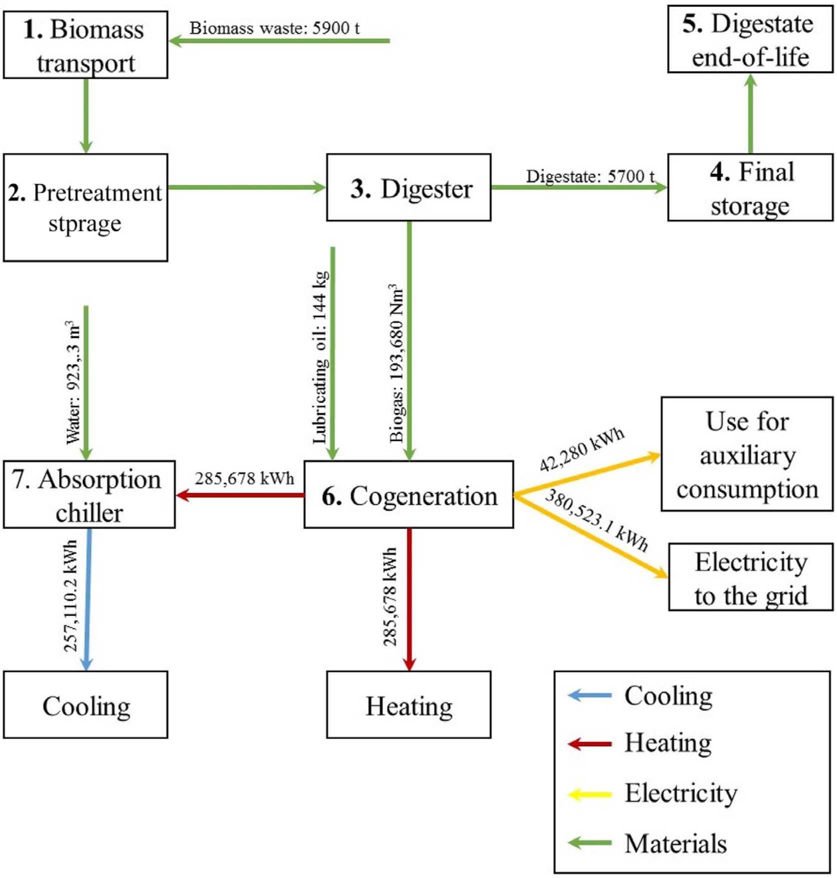

Fig. 7.1 The examined plant.

In order to refer the results of the analysis to the selected functional unit (1 MJ of exergy), the exergy produced during the system life cycle needs to be calculated.

The exergy content of the energy produced by the tri-generation plant can be defined as the sum of electricity plus the useful thermal energy times a Carnot coefficient. This factor is 1 for electricity considering that this form of energy can theoretically be entirely converted into work in a reversible machine. For thermal and refrigeration energy, the corresponding exergy (E) depends on the temperature of the delivered thermal fluid and the dead state temperature and it can be calculated by means of Eq. (7.1):

$$E = Q * \left(1 - \frac{T_0}{T_1}\right) \tag{7.1}$$

where Q is the useful thermal/refrigeration energy, T_0 is the dead state temperature equal to 25 °C, and T_1 is the temperature of the thermal fluid (50°C for heating and 5°C for cooling).

Considering the efficiency of the system components (cogeneration plant and absorption chiller), the yearly exergy production is

– 1369.9 GJ for electricity
– 79.6 GJ for heating
– 73.9 GJ for cooling

Thus the total exergy production during the whole life cycle of the plant (30 years) is equal to 45,701.9 GJ.

2.2.2 Data collection and data quality

The primary data were collected by using the project outlines and the datasheets of the plant components. Table 7.2 presents the main materials used for the construction of each component of the plant.

Table 7.2 Main materials of the examined plant

Component	Material	Quantity (kg)
Pretreatment storage tank (50 m^3)	Concrete	4205
	Polyethylene	50.2
Anaerobic digester (1000 m^3)	Steel	21,609.5
	Concrete	27,660.1
	Polystyrene	1905.6
	Polyethylene	336.3
	Polyester	374
Storage tank (650 m^3) for the final storage	Concrete	22,518
	Polyethylene	81.6
Pumps and pipes[a]	Aluminum	8.6
	Steel	70.4
	Copper	1.5
	PVC	144
Absorption chiller (25 kW$_t$)	Steel	1850
	Aluminum	210
	Copper	240
	Rock wool	20
	Plastics	32.5
	Electronic components	15
	Ammonia	18
Cogeneration plant (50 kW$_e$)	Steel	1360.3
	Iron	330

[a]This item refers only to the pipes for heating the digester. It does not include the pipes of the plant.

The eco-profiles of the raw materials, materials and energy sources, the transport and end-of-life processes are referred to environmental databases (Frischknecht et al., 2007).

2.3 Life cycle impact assessment and interpretation

The energy impact (CED) for the functional unit (1 MJ of exergy) is 4.91E − 02 MJ, of which about 96% is nonrenewable energy (4.71E − 02 MJ).

The production step is responsible of about 76% of CED (3.73E − 02 MJ), while the operation and end of life give a contribution to the total impact of about 20% (9.82E − 03 MJ) and 4% (1.94E − 03 MJ), respectively.

Focusing on the production step, the steel used in the manufacturing of the digester is responsible for about 51% of the energy impact, while about 28% of the impact is attributable to the use of steel in the cogeneration plant.

Table 7.3 presents the environmental impact of the functional unit. The contribution of each life cycle step to the total impacts is illustrated in Fig. 7.2.

The results highlighted that the operation step gives a contribution of about 66% to the impact on HT_c and higher than 90% to almost all the examined impacts categories.

The end-of-life step causes about 87.7% of the impact on E_{FW}, mainly due to the copper treatment, and contributes for less than 5% to the other impacts. The production step is responsible of the main impacts on IR (about 88%), OD (about 50%), and RD (about 96%).

Table 7.3 Environmental impacts of 1 MJ of exergy

Impact category	Unit of measure	Quantity
AP	Mole of $H+_{eq}$	4.20E − 04
E_{FW}	CTUe	3.31E − 01
FE	kg P_{eq}	1.70E − 05
HT_c	CTUh	5.84E − 10
HT_{nc}	CTUh	1.16E − 08
IR	kg $U235_{eq}$	1.45E − 01
GWP	kg CO_{2eq}	9.56E − 02
ME	kg N_{eq}	9.03E − 06
TE	Mole of N_{eq}	1.51E − 03
OD	kg CFC_{11eq}	1.36E − 10
PM	kg $PM2.5_{eq}$	1.16E − 05
POF	kg NMVOC	1.10E − 04
RD	kg Sb_{eq}	7.03E − 08
TFC	UBP	6.03E − 02

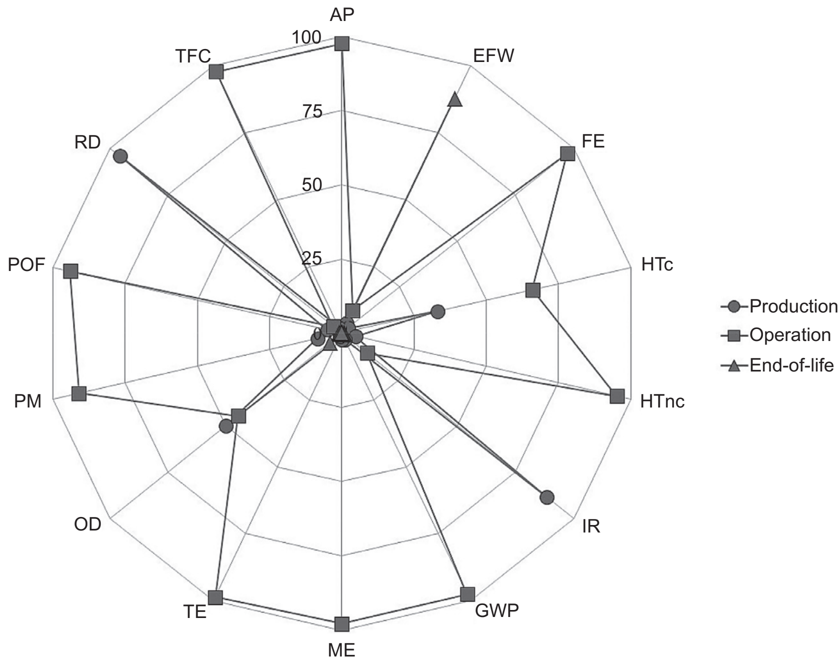

Fig. 7.2 Contribution of each life cycle step to the total environmental impacts.

A dominance analysis of the operation step shows that the biogas production in the digester causes about 95% of the impact on GWP and ME and about 83.5% of the impact on TE; in addition, this step is responsible of about 68% and 58.5% of the impacts on AP and PM, respectively.

The higher impacts on POF, about 64%, are caused during the cogeneration process, while the water use process is the main responsible (about 99.9%) of the impact on TFC. More than 97% of the impact on E_{FW}, FE, and HT are caused during the final treatment of the digestate, and more than 98.5% of the impacts on RD, IR, and OD are caused by the lubricating oil consumption. A contribution to the impacts lower than 1% is due to the transport of the biomass waste to the plant.

Looking at the production step, the main impacts on FE (about 54%), HT_c (about 63%), IR (about 85%), and OD (about 89%) are caused by the cogeneration plant manufacturing, while the contribution of this system to the other impact categories ranges from 13.5% (RD) to 32.5% (E_{FW}). The absorption chiller causes the main impact (about 66.6%) to E_{FW} and it is responsible for about 45% of the impact on FE and for about 31%–33% of the impact on HT and RD. In addition, it influences the other impact categories with a contribution lower than 15%. A negligible impact is caused by the pumps production, except for the impact on RD that is about 51% of the total. The impact of the remaining components (digester, storage tanks, and pipes) varies from 45% (HT_{nc}) to 80% (TFW) and is lower than 4% for E_{FW}, FE, HT_c, IR, OD, and RD. These impacts are mainly caused by the steel use in the manufacturing process of the digester.

3 Conclusion

This paper presented an experience of LCA application to a tri-generation plant in a developing country. This kind of approach represents a novelty in the Tunisian area and can be easily disseminated and applied in other developing countries in the Mediterranean area. The analysis allowed pointing out some important considerations and closing remarks.

First, the critical issues to be tackled for the environmental improvement of the system were identified. In detail, the use of steel in the manufacturing of the digester was identified as a key issue for reducing the primary energy consumption and some environmental impacts of the selected functional unit. Furthermore, the biogas production and combustion are critical processes to be taken into account for improving the environmental sustainability of the examined system. These processes are influenced by different parameters to be carefully assessed and monitored, for example, the biomass waste in input to the plant, its composition, and the biogas production from the input waste. In detail, the yield of biogas and its percentage of methane are dependent on the biomass composition. Using the most appropriate biomass waste composition can help improving the efficiency of the tri-generation plant and to reduce the environmental impacts of the selected functional unit.

The study also highlighted the low contribution of the biomass transport to the total impacts, due to the short distance between the plant and the areas within which biomass wastes are collected. Thus the supply of local resources is one of the main issue to be assessed in order to avoid that the transport of biomass would reduce and, potentially, cancel the environmental benefits of producing energy from a renewable energy source.

The obtained results can help stimulate the debate with local policy makers and organizations about policies that can be applied in order to optimize the biomass waste use for energy generation and to improve its sustainability.

The valorization of waste biomass can generate benefits that not only involve the examined supply chain, but also all the geographically neighboring stakeholders (citizens, farmers, municipalities, etc.), in the perspective of the industrial ecology application.

Some environmental challenges of developing countries, as those related to the waste management, can be turned into opportunities for all the involved stakeholders, both economically (availability of low-cost energy, reduction of waste to be disposed, production of bio-energy to be sold, etc.) and socially (improvement of the hygienic conditions and reduction of health risks, waste reduction, good quality of food produced in good agricultural and livestock areas, good living areas, etc.).

Furthermore, the strategies and procedures developed during this experience can be applicable to other similar experience on bio-energy supply chain in developing countries, even if it is important to outline that the replicability of this experience should look at the specific conditions of the examined bio-energy system, such as biomass typologies, supply systems, and treatment plants.

Finally, the study contributes to the literature of LCA applications with one of the first studies to a tri-generation plant filled with biogas, and its biomass supply chain, in a developing country. This would allow to turn one of the major problems of developing countries, notably the waste management, into a resource for the local territory.

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CHAPTER 8

Life-cycle costing: Analysis of biofuel production systems

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1 Introduction

1.1 Life-cycle costing concepts

Sustainability has emerged as a key concept for the evaluation of human activities. While it is frequently used to describe products and activities, sustainability is a complex concept subject to many interpretations. In 1987 the United Nations released the Brundtland report which defined sustainable development as "development that meets the needs of the present without compromising the ability of future generations to meet their own needs" (Brundtland, 1987). Most people would agree with this definition, but quantifying sustainability, such that claims for the sustainability of a product or project may be validated, is difficult. The United Nations 2030 Agenda for Sustainable Development expressed the concept that sustainability may be broken down into economic, environmental, and social dimensions (United Nations, 2015). This concept provides a means of analyzing

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sustainability in an objective manner. Approaches such as life-cycle sustainability assessment (LCSA) provide a framework for quantifying the sustainability of a man-made system (Kloepffer, 2008). The economic dimension (and perhaps even the other dimensions) can be analyzed through life-cycle costing (LCC).

There are commonly three levels by which LCC may be conducted: (1) fLCC (financial LCC), (2) eLCC (environmental LCC), and (3) sLCC (societal LCC) (Rödger et al., 2017). The concepts behind fLCC, sometimes called the conventional LCC, has its roots in a cost accounting method (Life Cycle Initiative, 2011). It focuses on financial streams of a single actor or entity. These would be items like investment cost, R&D cost, and revenue of a single actor; an industrial or manufacturing firm, for example (Hoogmartens et al., 2014). While the usual costs like raw materials, labor, maintenance, and energy are necessarily included, end-of-life costs and credit for by-products (as negative costs) may also be included if charged to the manufacturer (Hoogmartens et al., 2014). In fLCC, the concept that there is a difference between the present value and the future value of numerically equal amounts of currency can be accounted for by discounting. Inflation or interest rates can be reflected by the computation of a net present value via a discount factor or present worth factor $D=\frac{1}{(l+r)^i}$, where r is the prevailing interest rate and i is the period at which the cost was incurred. For an annual recurring cost, P_i, the net present value (NPV) may be computed as $\mathrm{NPV}=\sum_{i=l}^{n}\frac{P_i}{(l+r)^i}$ where r is the annual interest rate, i is the number of years after the present time when the cost was incurred, and n is the total life of the project in years.

The eLCC is the full life-cycle assessment that integrates economic aspects into environmental impacts assessment (Kloepffer, 2008). The key difference between the fLCC and the eLCC is that all of the impacts on the physical environment as a result of producing the product or service are charged to the life-cycle cost, even if the costs are not borne by the producer. Such costs are referred to as externalities because they are not normally paid for by the principal agent, and the purpose of eLCC is to internalize them into the computations. Thus the eLCC expands to more than one actor. For example, unless there is an imposed carbon tax, any damage from climate change caused by the CO_2 generated from producing and operating the vehicle is not charged to either the producer or the operator. This example also illustrates the challenges faced in the expansion from an fLCC to an eLCC. A financial or monetary cost needs to be assigned to the environmental impact caused by emissions through an appropriate valuation procedure. Pollutants like reactive nitrogen may have a "cascade" of impacts as the nitrogen emission transforms from different types of molecules of varying reactivity and moves through the environment (Galloway et al., 2003). The total cost to the environment thus changes as the nature of the pollutant changes and thus the characterization problem is always changing.

Conceptually, the eLCC is the closest to LCA. Heijungs et al. (2013) proposed a unified framework to integrate both the established computational structure of LCA (Heijungs and Suh, 2002) and that of LCC. By adopting the structure of LCA, the risk of double counting is avoided, since the value added is accounted for in each individual process, and not in subsequent processes. Note that, since the eLCC follows the computational structure of LCA, the implied assumption that the entire system is at steady state is also adopted (Rödger et al., 2017). This systematic approach also makes it easier to scale up the computations for complex, large-scale product systems. This approach makes the basic assumption that the production chain or network adds value to the product, while its end use consumes this accumulated value. Moreau and Weidema (2015) proposed the reinterpretation of eLCC as the "sum of the value added for each activity in the product life-cycle for each and every actor involved, including externalities which are foreseen to be internalized in the decision-relevant future."

Once the different processes included in the LCA have been identified, it is then possible to generate the technical coefficient matrix, $\mathbf{A}_p$, which represents the fixed proportions of material and energy flows for each process in the system. Matrix $\mathbf{A}_p$ consists of rows which represent economic streams, and columns which represent the processes within the system. By convention, a negative entry in the matrix represents an input to a process, while a positive entry indicates an output or a production. The streams may be tangible physical products (e.g., material or energy streams) or intangible outputs with appropriate units of measure (e.g., services). It is then possible to scale the different processes to achieve the desired functional unit as represented in the functional unit vector, $\mathbf{f}_p$, as indicated in Eq. (8.1). For systems with zero degrees of freedom, it is then possible to solve for the scaling vector $\mathbf{s}_p$ by using Eq. (8.2).

$$\mathbf{A}_p \mathbf{s}_p = \mathbf{f}_p \tag{8.1}$$

$$\mathbf{s}_p = \mathbf{A}_p^{-1} \mathbf{f}_p \tag{8.2}$$

This framework can then be extended to calculate for the life-cycle cost of a system by evaluating the scaled technology matrix, $\mathbf{A}_{p,\text{scaled}}$ as shown in Eq. (8.3) where diag($\mathbf{s}$) is the diagonalized scaling vector. The associated costs for the system are then obtained using Eq. (8.4) where diag($\boldsymbol{\alpha}$) is the diagonalized price vector which contains the market value for the relevant streams identified. Finally, the value added vector, $\mathbf{v}$, can be obtained using Eq. (8.5) where $\mathbf{1}$ is a vector consisting of all ones.

$$\mathbf{A}_{p,\text{scaled}} = \mathbf{A}_p \,\text{diag}\left(\mathbf{s}_p\right) \tag{8.3}$$

$$\mathbf{A}_{m,\text{scaled}} = \text{diag}(\boldsymbol{\alpha})\mathbf{A}_{p,\text{scaled}} \tag{8.4}$$

$$\mathbf{v} = \left(\mathbf{A}_{m,\text{scaled}}\right)^T \mathbf{1} \tag{8.5}$$

The life-cycle cost is then equivalent to vector $\mathbf{v}$. To illustrate this approach, we use the following example from Heijungs et al. (2013) and the subsequent reinterpretation by Moreau and Weidema (2015) which considers the life cycle of a chair. The system is shown in Fig. 8.1, the technology matrix ($\mathbf{A}_p$) is given in Table 8.1, while the associated market price

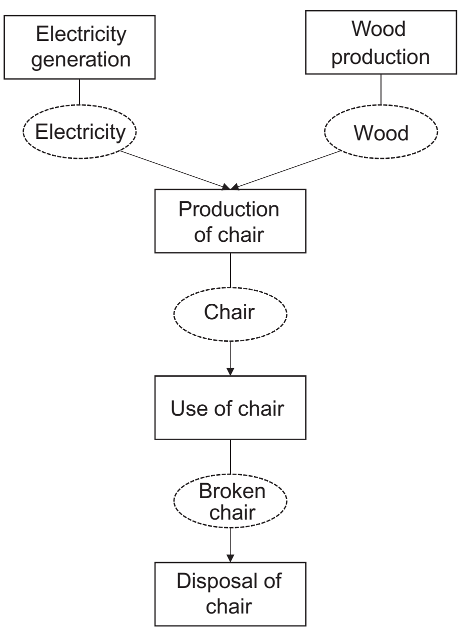


Fig. 8.1 Process flow diagram for chair life cycle. *(Modified from Heijungs, R., Settanni, E., Guinée, J., 2013. Toward a computational structure for life cycle sustainability analysis: unifying LCA and LCC. Int. J. Life Cycle Assess. 18, 1722–1733. doi:10.1007/s11367-012-0461-4.)*

Table 8.1 Technology matrix for chair life cycle ($\mathbf{A}_p$)

	Units	Electricity generation	Wood production	Production of chair	Use of chair	Chair disposal service
Electricity	MJ	1	0	−2	0	0
Wood	kg	0	1	−5	0	0
Chair	Piece	0	0	1	−5	0
Broken chair	Piece	0	0	0	5	−1
Sitting service	Years	0	0	0	10	0

Table 8.2 The price vector or the market value per unit of product or service (α)

Product	Market price
Electricity	5 €/MJ
Wood	1 €/kg
Chair	25 €/piece
Broken chair	−2 €/piece
Sitting service	13.5 €/year

for the identified products and services is presented in Table 8.2 and is equivalent to the price vector $\boldsymbol{\alpha}$. It is important to note that the negative market value for the broken chair is for the associated cost needed for its disposal. Also, note that, following the reinterpretation of Moreau and Weidema (2015), there is a market value assigned for sitting in the chair. While the user does not pay or get taxed for sitting, there is a value for the use of the chair and may be computed as the price of the chair (12.5 €) plus its disposal (1 €).

To calculate for the life-cycle cost associated with 1 year of sitting, the functional unit vector can be represented by Eq. (8.6). The scaling vector can then be obtained using Eq. (8.2) but for this example, the values are indicated in Eq. (8.7). This indicates that 1 MJ of electricity and 2.5 kg of wood are needed to produce 0.5 pieces of chairs to provide 1 year of sitting service.

$$\mathbf{f}_p = \begin{pmatrix} 0 \\ 0 \\ 0 \\ 0 \\ 1 \end{pmatrix} \tag{8.6}$$

$$\mathbf{s}_p = \mathbf{A}_p^{-1}\mathbf{f}_p = \begin{pmatrix} 1 & 0 & -2 & 0 & 0 \\ 0 & 1 & -5 & 0 & 0 \\ 0 & 0 & 1 & -5 & 0 \\ 0 & 0 & 0 & 5 & -1 \\ 0 & 0 & 0 & 10 & 0 \end{pmatrix}^{-1} \begin{pmatrix} 0 \\ 0 \\ 0 \\ 0 \\ 1 \end{pmatrix} = \begin{pmatrix} 1 \\ 2.5 \\ 0.5 \\ 0.1 \\ 0.5 \end{pmatrix} \tag{8.7}$$

The scaled technology matrix is then obtained using Eq. (8.3). The specific result for this example is shown in Eqs. (8.8a), (8.8b). This resulting scaled technology matrix clearly shows how materials and services flow through the different processes of the system such that a balanced system will only have a net output equivalent to the desired functional unit.

$$\mathbf{A}_{p,\text{scaled}} = \mathbf{A}_p \operatorname{diag}(\mathbf{s}_p) = \begin{pmatrix} 1 & 0 & -2 & 0 & 0 \\ 0 & 1 & -5 & 0 & 0 \\ 0 & 0 & 1 & -5 & 0 \\ 0 & 0 & 0 & 5 & -1 \\ 0 & 0 & 0 & 10 & 0 \end{pmatrix} \begin{pmatrix} 1 & 0 & 0 & 0 & 0 \\ 0 & 2.5 & 0 & 0 & 0 \\ 0 & 0 & 0.5 & 0 & 0 \\ 0 & 0 & 0 & 0.1 & 0 \\ 0 & 0 & 0 & 0 & 0.5 \end{pmatrix} \tag{8.8a}$$

$$\mathbf{A}_{p,\text{scaled}} = \begin{pmatrix} 1 & 0 & -1 & 0 & 0 \\ 0 & 2.5 & -2.5 & 0 & 0 \\ 0 & 0 & 0.5 & -0.5 & 0 \\ 0 & 0 & 0 & 0.5 & -0.5 \\ 0 & 0 & 0 & 1 & 0 \end{pmatrix} \tag{8.8b}$$

The equivalent monetary matrix is then obtained using Eq. (8.4), with the results for this example shown in Eqs. (8.9a), (8.9b). This shows the material costs and value added into goods and services. The third column of $\mathbf{A}_{m,\text{scaled}}$, for example, demonstrates how value is added to the chair which has a value of € 12.5 for 0.5 pieces, considering that only electricity (€ 5 for 1 MJ) and wood (€ 2.5 for 2.5 kg) amounting to a total of € 7.5 were utilized as input streams.

$$\begin{aligned} \mathbf{A}_{m,\text{scaled}} &= \operatorname{diag}(\boldsymbol{\alpha})\mathbf{A}_{p,\text{scaled}} \\ &= \begin{pmatrix} 5 & 0 & 0 & 0 & 0 \\ 0 & 1 & 0 & 0 & 0 \\ 0 & 0 & 25 & 0 & 0 \\ 0 & 0 & 0 & -2 & 0 \\ 0 & 0 & 0 & 0 & 13.5 \end{pmatrix} \begin{pmatrix} 1 & 0 & -1 & 0 & 0 \\ 0 & 2.5 & -2.5 & 0 & 0 \\ 0 & 0 & 0.5 & -0.5 & 0 \\ 0 & 0 & 0 & 0.5 & -0.5 \\ 0 & 0 & 0 & 1 & 0 \end{pmatrix} \end{aligned} \tag{8.9a}$$

$$\mathbf{A}_{m,\text{scaled}} = \begin{pmatrix} 5 & 0 & -5 & 0 & 0 \\ 0 & 2.5 & -2.5 & 0 & 0 \\ 0 & 0 & 12.5 & -12.5 & 0 \\ 0 & 0 & 0 & -1 & 1 \\ 0 & 0 & 0 & 13.5 & 0 \end{pmatrix} \tag{8.9b}$$

The life-cycle cost is then obtained using Eq. (8.5) while the results for our example are shown in Eq. (8.10). The corresponding life-cycle cost of 1 year of sitting is the sum of the elements of vector, $\mathbf{v}$. The total life-cycle cost of € 13.50 accounts for the cost associated with the purchase of 0.5 pieces of chair (€ 12.50) and its disposal (€ 1.00). Note that the costs associated with electricity and wood use have already been integrated in the cost of the chair. Note that the value added for sitting in the chair, element $\mathbf{v}_4$, is reflected as zero, which is more intuitively satisfying.

$$\mathbf{v} = \begin{pmatrix} 5 & 0 & -5 & 0 & 0 \\ 0 & 2.5 & -2.5 & 0 & 0 \\ 0 & 0 & 12.5 & -12.5 & 0 \\ 0 & 0 & 0 & -1 & 1 \\ 0 & 0 & 0 & 13.5 & 0 \end{pmatrix}^T \begin{pmatrix} 1 \\ 1 \\ 1 \\ 1 \\ 1 \end{pmatrix} = \begin{pmatrix} 5 \\ 2.5 \\ 5 \\ 0 \\ 1 \end{pmatrix} \tag{8.10}$$

This example of the application of eLCC shows how eLCC can expand beyond the fLCC and account for costs that may not necessarily be borne by the user or the producer. In the case of the chair, the eLCC is the value added by the individual providers of the electricity, wood chair, and disposal services. Thus all actors along the supply and use chain are accounted for.

Finally, the third type of LCC, sLCC extends the LCC concept to consider all social costs and benefits (Hoogmartens et al., 2014). Computationally, the sLCC would follow the same structure as the one for the eLCC described earlier. The sLCC would thus have to identify and monetize such issues as the equitable distribution of wealth, protection of vulnerable groups like migrants, indigenous groups, children and the elderly, and many others. Since these are not tied to any physical entities, it can easily be seen that even merely identifying the important impacts to be considered is extremely difficult, much less obtaining determining the appropriate characterizations and the required data. Fig. 8.2, adapted from UNEP 2011, shows how the boundaries of each of the three different LCCs may overlap each other. Inevitably, for a full LCSA, there would be some processes that cannot be monetized.

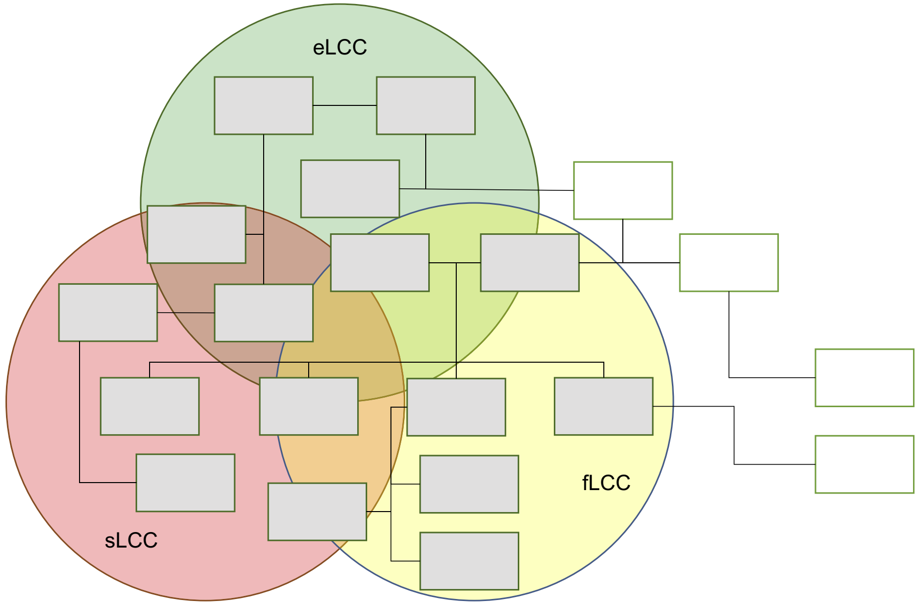


Fig. 8.2 The overlapping system boundaries of the three different types of LCC. *(Modified from UNEP/SETAC, 2011. Life Cycle Initiative. Towards a Life Cycle Sustainability Assessment.)*

1.2 Uncertainty in life-cycle costs

These costs are based on predicted and actual assumptions. Many LCC studies have assumed that all input parameters of the LCC model are deterministic (Tang et al., 2015). However, LCC calculation could involve many uncertainties, even at the lowest level of fLCC. The methods to address them have yet to be developed systematically (Ilg et al., 2017) The problem is even more pronounced for prospective systems like proposed biofuel systems using novel feedstocks which strongly affect the operating cost (Myint and El-Halwagi, 2009) or new process equipment that strongly affect the investment cost (Brownbridge et al., 2014). For novel processes under development, effects of technology maturity and scale-up are also difficult to gauge (Chan et al., 2018). Data is usually difficult to obtain because of corporate confidentiality concerns and they are also time sensitive. A good example of a time-sensitive parameter that strongly influences the LCC is interest rate. Since the randomness and the uncertainties behind these parameters are uncontrollable, there is a need to systematically assess the impacts of these parameters on the cost of the product.

There are many methods to assess the uncertainties in LCA or LCC including interval analysis (Chevalier and Le Téno, 1996), probabilistic (Kennedy et al., 1996), and fuzzy numbers methods (Tan, 2008). The uncertainty of single parameters and their effects on the final results can be assessed through sensitivity and matrix perturbation analysis (Heijungs, 2010). In Heijungs (1996), the key issues analysis method was used to guide improvements in data collection.

The simultaneous analysis of the effects of multiple parameters can be done via a method that has been traditionally used guiding experiments in the physical world: design of experiments (DOE). Long used for the analysis of complex systems like those found in agriculture and other living systems, DOE has also been used for designing computational experiments (Giunta et al., 2003). That is, DOE is used to guide how parameter values may be systematically varied such that the effects of individual parameters may be assessed simultaneously, the overall uncertainty may also be gauged. This is also known as global sensitivity analysis (GSA). Among the more popular alternative or competing methods is Monte Carlo simulation which has been used to determine lumped uncertainty in LCA (Ciroth et al., 2004). The Monte Carlo method, however, requires a large amount of calculation (Lloyd and Ries, 2008) although there have been algorithms proposed to reduce computational times (Peters, 2007). Other types of DOE are quasi-Monte Carlo sampling, pseudo-Monte Carlo sampling, Hammersley sequence sampling, and Latin hypercube sampling (Giunta et al., 2003).

When applying DOE to the computational results of models that result from LCA or LCC, the input parameters are considered to be the independent variable of the experiment and combination of levels in the input parameters in the model are considered as experimental "treatments." The purpose of the analysis is the attribution of uncertainty of final results to various sources of error in the inputs. Typically, the input parameters chosen to be used as the independent variables in the DOE are those whose estimates the model creator expects to be most unpredictable or the least reliable.

Another advantage of DOE for the analysis of models is that it allows the quantification of factor independence with a rigorous statistical framework. Specifically, the Latin hypercube method estimates linear and quadratic effects and bilinear interactions (Ye, 1998) to obtain low-order polynomial models. These polynomial models serve as approximations of the underlying, more complex model and are easier to understand on an intuitive level. On the other hand, while the traditional local or one-at-a-time sensitivity analysis is easier to implement, the polynomial models provide more information including interaction effects and confidence intervals for the polynomial coefficients.

In the next section, we show an example wherein a simple model for the financial life-cycle cost of biodiesel in Vietnam and analyze the sensitivity of the life-cycle cost to selected parameters. This is a more detailed exposition of results previously published in Khang et al. (2017, 2018).

2 Example of sensitivity analysis of biodiesel via DOE

2.1 Financial LCC of biodiesel

Ong et al. (2012) proposed a simple model for estimating the financial life-cycle cost of a palm oil biodiesel. It is easily translated to other feedstocks as will be done here. For a project life of n years, the life-cycle cost, here taken as the present value of the project, can be defined as

$$\mathrm{LCC} = \mathrm{CC} + \mathrm{OC} + \mathrm{MC} + \mathrm{FC} - \mathrm{SV} - \mathrm{BP} \tag{8.11}$$

where

CC = capital cost

OC = operating cost (US$) $= \sum_{i=l}^{n} \frac{\mathrm{OR} \times \mathrm{PC}}{(l+r)^{i.}}$

OR = operating rate (US$/t)
PC = annual biodiesel production capacity (t/y)
r = interest rate
MC = maintenance cost (US$) $= \sum_{i=l}^{n} \frac{MR \times CC}{(l+r)^i}$
MR = maintenance ratio (%), the ratio of maintenance cost to capital cost
FC = feedstock cost (US$) $= \sum_{i=l}^{n} \frac{FP \times FU}{(l+r)^i}$
FP = feedstock price (US$/ton)
FU = feedstock utilization = PC/CE
CE = biodiesel conversion efficiency (%)
SV = salvage value (US$) $= RC \times (l\text{–}d)^{n-1}$
RC = replacement cost
d = depreciation rate
BP = by-product credit (US$) $= \sum_{i=l}^{n} \frac{GP \times GM}{(l+r)^i}$
GP = glycerol price (US$/kg)
GM = PC × GCF
GCF = glycerol conversion factor

Each of the costs is accounted for in the conventional manner. Capital cost includes process equipment, infrastructure, and land while operation cost includes labor, utilities, and waste treatment. Maintenance cost is assumed to be 2% of the total capital cost over the course of the entire project. Feedstock cost includes all raw materials including oil, methanol, and catalysts. The salvage value is the remaining value of capital cost at the end of the project, assuming a depreciation of 10%. Finally, a credit is given to subtract the cost of the glycerin by-product.

2.2 System description

In the following example, we assess the cost of three feedstocks that are being considered as potential feedstocks for biodiesel production in Vietnam. These are jatropha oil, waste cooking oil, and fish oil. The jatropha oil is oil obtained from the seeds of *Jatropha curcas*. The fish oil is residual oil recovered from fish processing plants while the waste cooking oil is obtained from restaurants, hotels, and households. The system boundary for jatropha oil is shown in Fig. 8.3 while the system boundary for the waste cooking oil and the fish oil is shown in Fig. 8.4. It can be seen that the system for the jatropha oil includes cultivation and extraction of the oil. On the other hand, the systems for the waste cooking oil and the fish oil only include gathering of the oil.

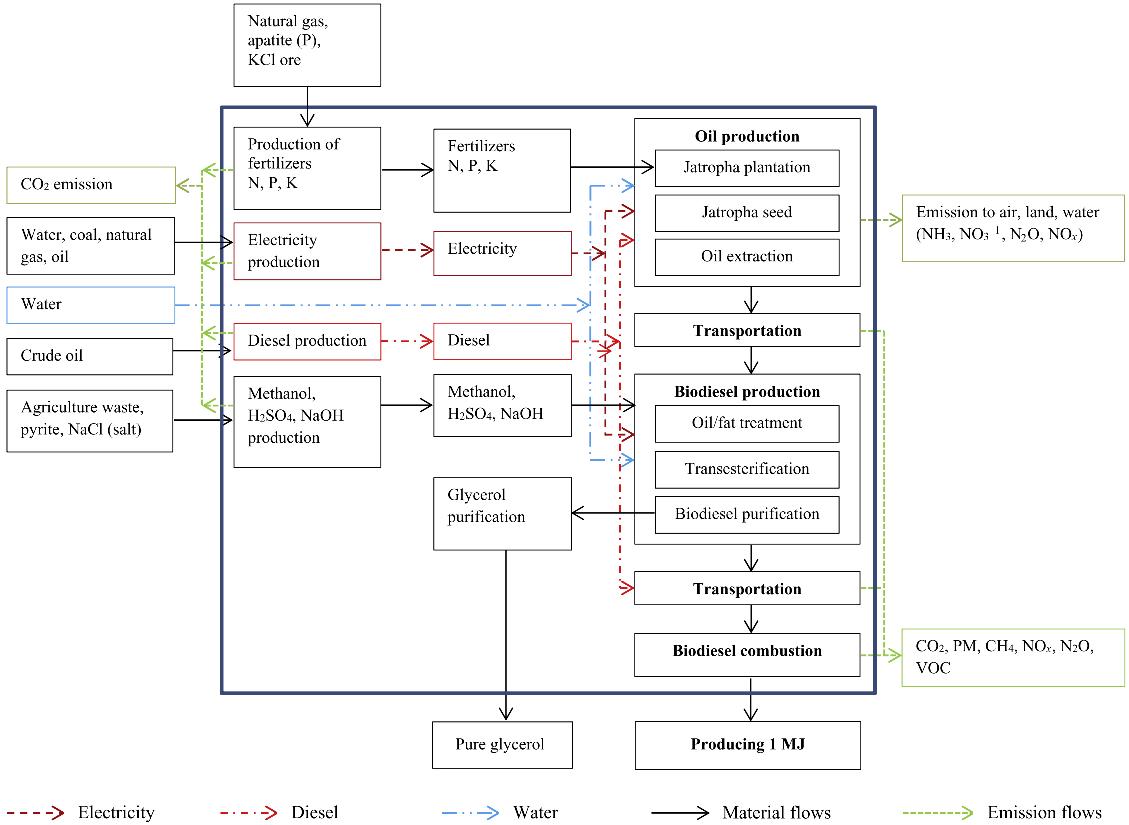


Fig. 8.3 System boundary for jatropha oil.

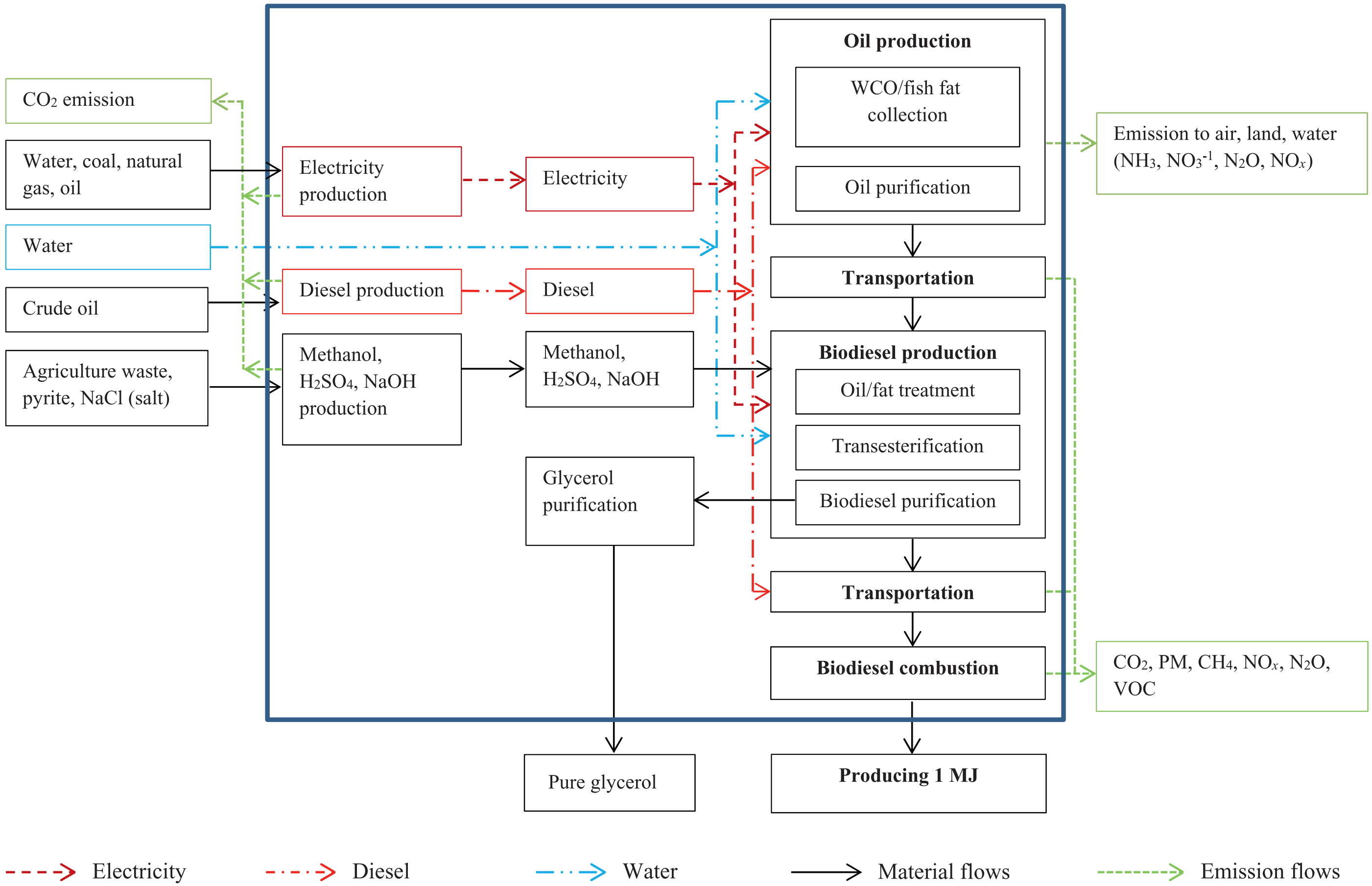


Fig. 8.4 System boundary for waste cooking oil and fish oil.

2.3 Global sensitivity analysis by Latin hypercube design of experiments

In the computations shown later, parameters were used as the independent variables in the DOE: plant capacity, interest rate, feedstock (fish oil, waste cooking oil, or jatropha oil) price, operating rate, biodiesel conversion. The range of the plant capacity was taken to be 50–100 kt/year. The interest rate was taken to be between 3% and 5%. The cost of one ton each of Jatropha oil, fish oil, and waste cooking oil was assumed to vary between US$400 and US$500, US$100 and US$400, and US$0 and US$300, respectively. Biodiesel conversion efficiency was taken to range between 90% and 98%. The operating rate was varied from US$190 to US$400 per ton.

Baseline cost distributions are shown in Fig. 8.5. To generate this figure, the midpoints of the range of the six selected parameters were chosen. It can be seen that feedstock cost, the operating cost, and the capital cost constitute the largest bulk of the costs albeit to varying degrees depending on the feedstock. Though technically negative values, the absolute values of salvage value and by-product credit are included to provide an idea of their relative magnitudes. It can be seen that these and the maintenance cost are relatively small.

For the sensitivity analysis of the LCC model, JMP software was used to determine 50 sampling points for each feedstock via the Latin hypercube experimental design. These are presented in Tables 8.3–8.5

Table 8.6 presents the results from the sensitivity analysis by DOE of the LCC of biodiesel from jatropha oil. In this table, the factors that are significant at a 95% level of confidences are marked with an asterisk. It can be seen that 9 factors and factor interactions are significant. These include each of the 5 factors: operating rate, feedstock price, capacity, interest rate, and biodiesel conversion efficiency and 4 interactions: the square of (capacity-75) and the products (interest rate-0.04) × (operating rate-295), (biodiesel conversion efficiency-0.94) × (feedstock price-450), and (feedstock price-450) × (interest rate-0.04). The linear effect of each of the 5 factors maybe expected but the design of experiments approach for sensitivity analysis provides additional information on the magnitude of each impact in relation to each other. The factor with the largest positive (unfavorable) impact on the LCC is the operating rate followed by the feedstock price and the interest rate. On the other hand, the capacity and the biodiesel conversion efficiency have negative (favorable) impacts on the life-cycle cost. A large capacity and a high biodiesel conversion efficiency would result in a lower cost. Similarly,

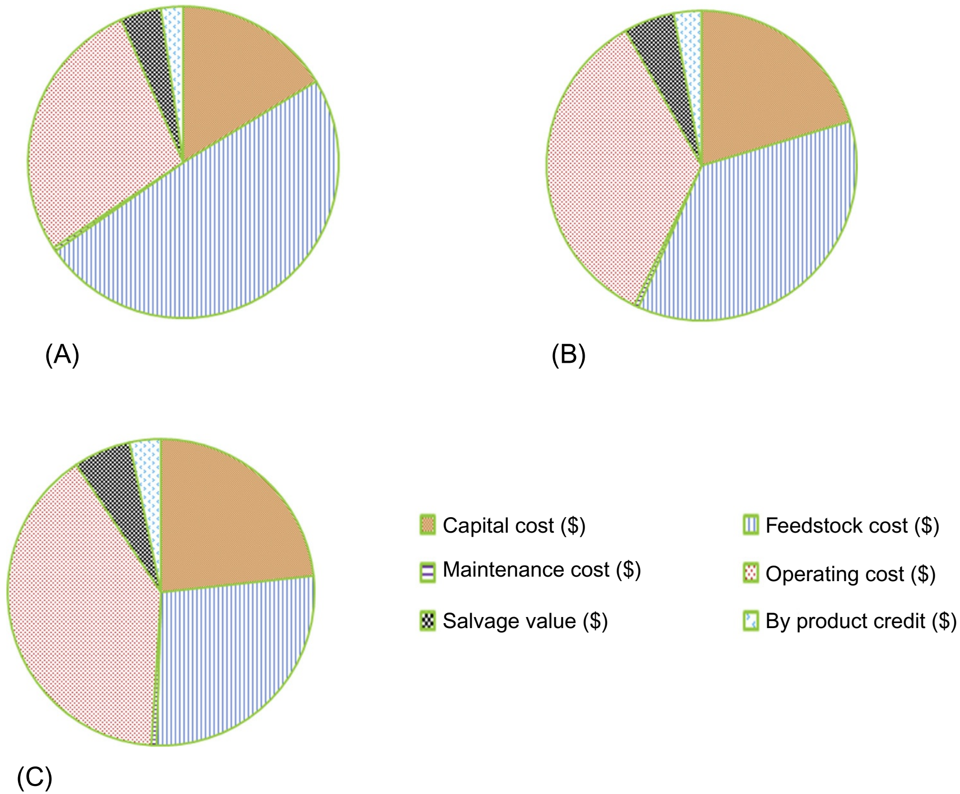


Fig. 8.5 Cost breakdown of biodiesel from (A) jatropha oil, (B) fish oil, and (C) waste cooking oil.

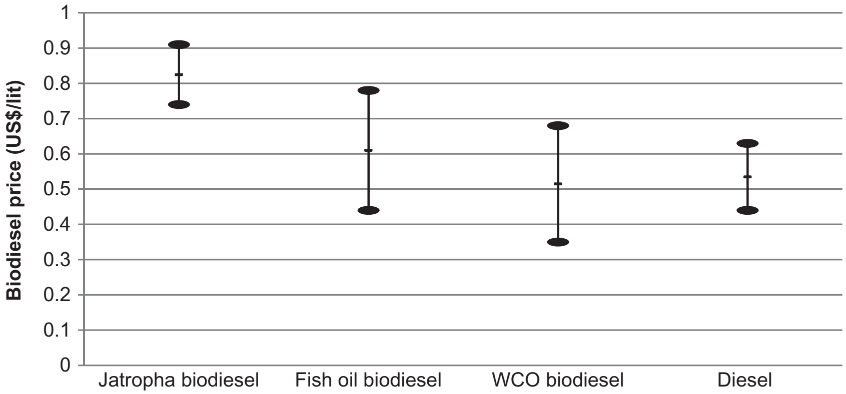


Fig. 8.6 Comparison of the cost of biodiesel from various feedstocks to the price of petroleum diesel.

while four interactions are identified as statistically significant, their influence on the overall cost is less.

The model obtained from the DOE sensitivity analysis is captured in a second-order regression (Eq. 8.12).

$$\begin{aligned} C_{\text{jatropha-biodiesel}} = {} & 0.5753 - 0.4996X_1 + 0.001047X_2 - 0.00136X_3 \\ & + 2.7154X_4 + 0.00098X_5 + 0.186 \times 10^{-5}(X_3 - 75)^2 \\ & + 0.00377(X_4 - 0.04)(X_5 - 295) - 0.001\,(X_1 \\ & - 0.94(X_2 - 450) + 0.00344(X_2 - 450)(X_4 - 0.04) \end{aligned} \tag{8.12}$$

where X_1 is the biodiesel conversion efficiency, X_2 is the feedstock price, X_3 is the capacity, X_4 is the interest rate, and X_5 is the operating rate.

Tables 8.7 and 8.8 present the sensitivity analyses for the fish oil and the waste cooking oil, respectively. Similar to jatropha oil, each of the five chosen parameters is found to be significant at a 95% confidence level. However, the relative magnitudes of the *t*-ratios are in a different order. It is the feedstock price that has the largest influence on the LCC followed by the operating rate and interest rate. This observation is rather counterintuitive because the feedstock price contributes a smaller share of the overall cost of the fish oil and waste cooking oil biodiesels. As with the jatropha oil, the negative *t*-ratios of capacity and biodiesel conversion efficiency indicate that increases in capacity and conversion efficiency reduce the LCC.

Table 8.3 Combinations of factor values provided by the Latin hypercube DOE for jatropha oil

Run	Biodiesel conversion efficiency	Feedstock price	Capacity	Interest rate	Operating rate	Biodiesel cost
1	0.93918	412.24	56.122	0.045102	374.29	0.95785
2	0.97510	469.39	76.531	0.036939	198.57	0.77114
3	0.96857	432.65	79.592	0.047143	232.86	0.79162
4	0.91796	455.10	51.020	0.042653	335.71	0.98154
5	0.97837	438.78	91.837	0.043469	357.14	0.89601
6	0.92939	410.20	66.327	0.030816	327.14	0.85540
7	0.90327	457.14	88.776	0.038980	280.00	0.86684
8	0.95061	416.33	81.633	0.037347	382.86	0.90315
9	0.93429	451.02	86.735	0.030408	275.71	0.81951
10	0.96204	422.45	87.755	0.034082	207.14	0.72022
11	0.94408	481.63	60.204	0.037755	395.71	1.0182
12	0.93592	442.86	58.163	0.031224	245.71	0.82440
13	0.95224	491.84	69.388	0.030000	271.43	0.86692
14	0.90490	418.37	55.102	0.038163	241.43	0.83352
15	0.98000	424.49	97.959	0.032857	322.86	0.81399
16	0.94898	489.80	74.490	0.046735	215.71	0.84938
17	0.95714	402.04	78.571	0.049592	340.00	0.88067
18	0.93755	477.55	62.245	0.050000	314.29	0.97044
19	0.92776	459.18	82.653	0.041837	400.00	0.98822
20	0.90653	436.73	59.184	0.047551	194.29	0.82007
21	0.93102	497.96	94.898	0.042245	288.57	0.90761
22	0.97020	440.82	70.408	0.032041	310.00	0.84837
23	0.91959	500.00	67.347	0.041020	318.57	0.97312
24	0.95388	430.61	53.061	0.033673	387.14	0.95690
25	0.96367	475.51	89.796	0.032449	378.57	0.93029

Continued

Table 8.3 Combinations of factor values provided by the Latin hypercube DOE for jatropha oil—cont'd

Run	Biodiesel conversion efficiency	Feedstock price	Capacity	Interest rate	Operating rate	Biodiesel cost
26	0.97184	453.06	65.306	0.043061	361.43	0.94828
27	0.95551	487.76	80.612	0.048776	391.43	1.0199
28	0.94735	408.16	98.980	0.041429	284.29	0.79530
29	0.94571	400.00	63.265	0.036122	211.43	0.74472
30	0.92449	479.59	52.041	0.043878	224.29	0.89440
31	0.97673	463.27	57.143	0.044694	228.57	0.84420
32	0.92612	444.90	92.857	0.044286	202.86	0.77551
33	0.95878	467.35	50.000	0.036531	297.14	0.92107
34	0.94082	448.98	73.469	0.039796	254.29	0.83232
35	0.91469	414.29	83.673	0.034898	220.00	0.75166
36	0.90163	483.67	61.224	0.035306	237.14	0.88097
37	0.91306	404.08	75.510	0.045510	258.57	0.81409
38	0.90000	465.31	72.449	0.047959	348.57	0.99047
39	0.90980	426.53	71.429	0.039388	344.29	0.91394
40	0.96041	461.22	96.939	0.038571	262.86	0.81697
41	0.97347	406.12	64.286	0.040612	292.86	0.82720
42	0.91633	471.43	68.367	0.031633	331.43	0.92795
43	0.96531	473.47	95.918	0.049184	267.14	0.86063
44	0.92122	485.71	85.714	0.035714	190.00	0.79432
45	0.91143	428.57	93.878	0.033265	352.86	0.88241
46	0.90816	434.69	100.00	0.046327	365.71	0.93684
47	0.94245	420.41	54.082	0.045918	250.00	0.84756
48	0.93265	446.94	84.694	0.048367	301.43	0.89154
49	0.96694	493.88	77.551	0.040204	305.71	0.91169
50	0.92286	495.92	90.816	0.034490	370.00	0.96948

Table 8.4 Combinations of factor values provided by the Latin hypercube DOE for fish oil

Run	Biodiesel conversion efficiency	Feedstock price	Capacity	Interest rate	Operating rate	Biodiesel cost
1	0.97673	265.31	77.551	0.047959	288.57	0.67538
2	0.94571	204.08	100.00	0.039796	335.71	0.63003
3	0.95388	240.82	91.837	0.030816	245.71	0.56900
4	0.96367	332.65	57.143	0.047143	378.57	0.87145
5	0.97510	167.35	87.755	0.040612	271.43	0.53489
6	0.90653	369.39	76.531	0.045510	224.29	0.74336
7	0.94082	363.27	81.633	0.046327	314.29	0.80832
8	0.90163	308.16	89.796	0.039388	292.86	0.71876
9	0.90980	124.49	63.265	0.041020	267.14	0.52854
10	0.91959	234.69	62.245	0.044286	190.00	0.57607
11	0.93755	106.12	83.673	0.038571	241.43	0.44945
12	0.96857	351.02	94.898	0.040204	348.57	0.79126
13	0.93429	246.94	74.490	0.038980	301.43	0.66685
14	0.91306	185.71	95.918	0.032449	280.00	0.55429
15	0.90816	253.06	69.388	0.050000	297.14	0.70625
16	0.96204	136.73	86.735	0.048367	352.86	0.60045
17	0.92776	100.00	80.612	0.048776	284.29	0.50312
18	0.97020	277.55	50.000	0.045102	250.00	0.69632
19	0.93265	179.59	70.408	0.030000	254.29	0.54275
20	0.98000	222.45	96.939	0.032857	331.43	0.62587
21	0.97184	295.92	73.469	0.038163	194.29	0.60132
22	0.96041	148.98	64.286	0.049184	207.14	0.49596
23	0.96531	118.37	60.204	0.030408	322.86	0.55923
24	0.95224	393.88	82.653	0.035306	262.86	0.75709

Continued

Table 8.4 Combinations of factor values provided by the Latin hypercube DOE for fish oil—cont'd

Run	Biodiesel conversion efficiency	Feedstock price	Capacity	Interest rate	Operating rate	Biodiesel cost
25	0.91796	387.76	66.327	0.032041	357.14	0.86909
26	0.95551	375.51	72.449	0.046735	215.71	0.72821
27	0.94898	259.18	51.020	0.034082	340.00	0.74696
28	0.97347	197.96	68.367	0.042245	391.43	0.71060
29	0.92612	314.29	92.857	0.041837	395.71	0.82078
30	0.92286	130.61	79.592	0.043061	382.86	0.62772
31	0.91633	338.78	67.347	0.042653	374.29	0.86090
32	0.91469	191.84	55.102	0.044694	370.00	0.72589
33	0.93918	228.57	85.714	0.047551	228.57	0.57673
34	0.94245	173.47	84.694	0.031224	365.71	0.62526
35	0.91143	381.63	65.306	0.034898	237.14	0.75922
36	0.90000	161.22	90.816	0.043469	232.86	0.50567
37	0.90490	271.43	52.041	0.036122	318.57	0.75371
38	0.92939	344.90	54.082	0.045918	275.71	0.79816
39	0.94408	289.80	53.061	0.033265	202.86	0.64099
40	0.92122	216.33	93.878	0.049592	344.29	0.68188
41	0.94735	142.86	58.163	0.043878	305.71	0.59473
42	0.96694	400.00	56.122	0.037347	258.57	0.79854
43	0.95878	320.41	98.980	0.041429	220.00	0.63654
44	0.93102	357.14	97.959	0.031633	327.14	0.76779
45	0.93592	112.24	59.184	0.036939	387.14	0.63074
46	0.92449	302.04	88.776	0.036531	198.57	0.60747
47	0.95714	155.10	61.224	0.037755	211.43	0.49812
48	0.95061	326.53	78.571	0.033673	400.00	0.82206
49	0.90327	210.20	75.510	0.034490	361.43	0.68419
50	0.97837	283.67	71.429	0.035714	310.00	0.69767

Table 8.5 Combinations of factor values provided by the Latin hypercube DOE for waste cooking oil

Run	Biodiesel conversion efficiency	Feedstock price	Capacity	Interest rate	Operating cost	Biodiesel cost
1	0.98000	146.94	93.878	0.040204	322.86	0.55817
2	0.96857	220.41	55.102	0.031224	344.29	0.69148
3	0.94571	251.02	90.816	0.038980	395.71	0.74288
4	0.95061	110.20	75.510	0.044694	391.43	0.61858
5	0.95224	287.76	50.000	0.043469	301.43	0.76034
6	0.96531	159.18	67.347	0.041837	292.86	0.57597
7	0.96204	67.347	57.143	0.034898	241.43	0.44312
8	0.97673	97.959	60.204	0.037347	400.00	0.62282
9	0.90980	122.45	53.061	0.042653	357.14	0.63949
10	0.93755	36.735	95.918	0.043061	327.14	0.45557
11	0.90327	189.80	92.857	0.030816	258.57	0.54018
12	0.92122	238.78	83.673	0.030408	378.57	0.70817
13	0.95714	226.53	63.265	0.032857	237.14	0.58418
14	0.92286	263.27	69.388	0.047959	352.86	0.76498
15	0.96041	134.69	77.551	0.031633	314.29	0.54313
16	0.97510	177.55	91.837	0.032041	224.29	0.48352
17	0.95388	300.00	71.429	0.042245	215.71	0.64225
18	0.90490	42.857	94.898	0.041429	245.71	0.38207
19	0.90816	73.469	62.245	0.045102	207.14	0.42019
20	0.93265	128.57	98.980	0.034082	340.00	0.54929
21	0.92612	116.33	65.306	0.033673	190.00	0.43026
22	0.97837	55.102	80.612	0.047551	288.57	0.45382
23	0.97020	275.51	82.653	0.036122	297.14	0.66671
24	0.94245	153.06	54.082	0.049184	318.57	0.63746

Continued

Table 8.5 Combinations of factor values provided by the Latin hypercube DOE for waste cooking oil—cont'd

Run	Biodiesel conversion efficiency	Feedstock price	Capacity	Interest rate	Operating cost	Biodiesel cost
25	0.91959	293.88	66.327	0.033265	284.29	0.70412
26	0.97347	30.612	84.694	0.036939	250.00	0.37548
27	0.92776	165.31	52.041	0.030000	310.00	0.61761
28	0.94898	232.65	89.796	0.045918	305.71	0.64937
29	0.93102	171.43	79.592	0.038163	271.43	0.55172
30	0.91469	6.1224	73.469	0.046735	382.86	0.50638
31	0.93429	48.980	72.449	0.050000	280.00	0.45410
32	0.90000	269.39	85.714	0.040612	361.43	0.75093
33	0.92449	208.16	70.408	0.048367	232.86	0.58216
34	0.95878	24.490	88.776	0.035306	387.14	0.49669
35	0.93592	104.08	87.755	0.045510	198.57	0.40898
36	0.92939	0.00	74.490	0.038571	267.14	0.37458
37	0.91306	85.714	76.531	0.036531	365.71	0.55669
38	0.94082	195.92	64.286	0.037755	370.00	0.69185
39	0.91796	202.04	51.020	0.041020	254.29	0.62499
40	0.96367	61.224	56.122	0.046327	194.29	0.40358
41	0.97184	183.67	81.633	0.049592	220.00	0.52036
42	0.94735	244.90	96.939	0.039388	202.86	0.54326
43	0.95551	18.367	58.163	0.044286	331.43	0.48986
44	0.90163	79.592	59.184	0.035714	275.71	0.49042
45	0.90653	281.63	78.571	0.039796	211.43	0.62068
46	0.91143	214.29	100.000	0.043878	262.86	0.58460
47	0.93918	91.837	97.959	0.034490	228.57	0.40501
48	0.91633	140.82	86.735	0.048776	335.71	0.59550
49	0.96694	257.14	68.367	0.047143	374.29	0.76674
50	0.94408	12.245	61.224	0.032449	348.57	0.48013

Table 8.6 The estimated coefficients and *P*-values: LCC of jatropha biodiesel

Term	Estimate	*t* Ratio	Prob>|*t*|
Operating rate	0.0009815		<.0001†
Feedstock price	0.001047		<.0001†
Capacity	–0.001364		<.0001†
Interest rate	2.7154324		<.0001†
Biodiesel conversion efficiency	–0.499616		<.0001†
(Capacity-75)*(Capacity-75)	1.8646e-5		<.0001†
(Interest rate-0.04)*(Operating rate-295)	0.0037701		<.0001†
(Biodiesel conversion efficiency-0.94)*(Feedstock price-450)	–0.001009		<.0001†
(Feedstock price-450)*(Interest rate-0.04)	0.0034463		<.0001†
(Biodiesel conversion efficiency-0.94)*(Interest rate-0.04)	–1.430167		0.1245
(Feedstock price-450)*(Feedstock price-450)	–2.308e-7		0.1781
(Capacity-75)*(Interest rate-0.04)	–0.001528		0.3109
(Interest rate-0.04)*(Interest rate-0.04)	3.4349643		0.3975
(Biodiesel conversion efficiency-0.94)*(Biodiesel conversion efficiency-0.94)	0.2010329		0.4580
(Biodiesel conversion efficiency-0.94)*(Operating rate-295)	3.4984e-5		0.6901
(Capacity-75)*(Operating rate-295)	–4.377e-8		0.7605
(Operating rate-295)*(Operating rate-295)	8.7365e-9		0.8055
(Feedstock price-450)*(Capacity-75)	7.1853e-8		0.8095
(Feedstock price-450)*(Operating rate-295)	–1.053e-8		0.8749
(Biodiesel conversion efficiency-0.94)*(Capacity-75)	–0.000018		0.9606

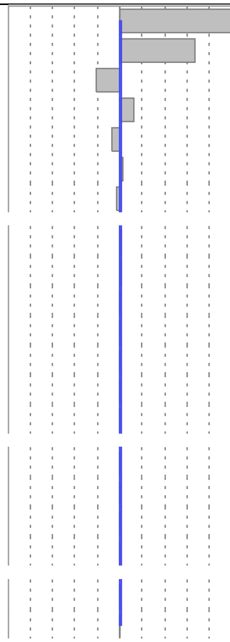
†Significant at a 95% degree of confidence.

Similarly, the LCC of fish oil and waste cooking oil biodiesel have a set of statistically significant interactions identical to that of jatropha oil biodiesel. The *t*-ratios of these interactions are of small magnitude like those of observed in the jatropha biodiesel sensitivity analysis but the order of their relative magnitudes is different.

The second-order regression equations that arise from the DOE analysis are captured in Eqs. (8.13), (8.14). The variables $X_1, \ldots, X_5$ are defined in the same manner as Eq. (8.12).

$$\begin{aligned} C_{\text{FO-biodiesel}} = {} & 0.398 - 0.2773X_1 + 0.00104X_2 - 0.00136X_3 + 1.952X_4 \\ & + 0.00098X_5 - 0.00109(X_1 - 0.94)(X_2 - 250) \\ & + 1.836 \times 10^{-5}(X_3 - 75)^2 + 0.0037(X_2 - 250)(X_4 - 0.04) \\ & + 0.0034(X_4 - 0.04)(X_5 - 295) \end{aligned} \tag{8.13}$$

Table 8.7 Estimated coefficients and *P*-values: LCC of fish oil biodiesel

Term	Estimate	*t* Ratio	Prob>|*t*|
Feedstock price	0.0010451		<.0001[†]
Operating rate	0.0009817		<.0001[†]
Capacity	−0.001365		<.0001[†]
Interest rate	1.9522147		<.0001[†]
Biodiesel conversion efficiency	−0.277292		<.0001[†]
(Capacity-75)*(Capacity-75)	1.8359e-5		<.0001[†]
(Biodiesel conversion efficiency-0.94)*(Feedstock price-250)	−0.001095		<.0001[†]
(Feedstock price-250)*(Interest rate-0.04)	0.0037242		<.0001[†]
(Interest rate-0.04)*(Operating rate-295)	0.0034087		<.0001[†]
(Feedstock price-250)*(Feedstock price-250)	−3.065e-8		0.1101
(Feedstock price-250)*(Capacity-75)	−1.461e-7		0.1574
(Operating rate-295)*(Operating rate-295)	−4.329e-8		0.2499
(Biodiesel conversion efficiency-0.94)*(Capacity-75)	−0.000357		0.3011
(Biodiesel conversion efficiency-0.94)*(Interest rate-0.04)	−0.768785		0.3944
(Feedstock price-250)*(Operating rate-295)	−1.707e-8		0.4448
(Capacity-75)*(Interest rate-0.04)	0.0008807		0.5326
(Interest rate-0.04)*(Interest rate-0.04)	1.9215701		0.6342
(Biodiesel conversion efficiency-0.94)*(Biodiesel conversion efficiency-0.94)	0.0703377		0.7846
(Capacity-75)*(Operating ratet-295)	−3.513e-8		0.7964
(Biodiesel conversion efficiency-0.94)*(Operating crate-295)	−0.000018		0.8345

[†]Significant at a 95% degree of confidence.

$$C_{\text{WCO-biodiesel}} = 0.308 - 0.167X_1 + 0.00104X_2 - 0.00136X_3 + 1.573X_4 + 0.00098X_5 - 0.00115(X_1 - 0.94)(X_2 - 150) + 1.89\times10^{-5}(X_3 - 75)^2 + 0.0033(X_4 - 0.04)(X_5 - 295) \tag{8.14}$$

By plugging in extreme values of the parameters $X_1, \ldots, X_5$ into Eqs. (8.12)–(8.14), a range of values may be obtained for the LCC of each of the feedstocks. The estimated cost of jatropha biodiesel in Vietnam ranges from US\$0.72 to US\$1.02 per liter. The cost of fish oil biodiesel may range between US\$0.44 and US\$0.87 per liter and, finally, the cost estimate of waste cooking oil biodiesel is between US\$0.37 and US\$0.77 per liter. The cost data was not made available because of confidentiality. So, these are compared to the *selling price* of petroleum diesel, which, in 2015, was selling at US\$0.42 to US\$0.61 per liter (Global Petrol Prices, 2015) in Fig. 8.6. Since this is the selling price of petroleum diesel, the actual cost of the diesel would be actually much less. It can be seen that the primary conclusion is

Table 8.8 The estimated coefficients and *P*-values: LCC of WCO biodiesel

Term	Estimate	*t* Ratio	Prob>|*t*|
Feedstock price	0.0010462		<.0001†
Operating rate	0.0009829		<.0001†
Capacity	–0.001366		<.0001†
Interest rate	1.5729815		<.0001†
Biodiesel conversion efficiency	–0.167171		<.0001†
(Capacity-75)*(Capacity-75)	0.0000189		<.0001†
(Biodiesel conversion efficiency-0.94)*(Feedstock price-150)	–0.001151		<.0001†
(Feedstock price-150)*(Interest rate-0.04)	0.0037928		<.0001†
(Interest rate-0.04)*(Operating rate-295)	0.0033239		<.0001†
(Feedstock price-150)*(Capacity-75)	–1.755e-7		0.0706
(Biodiesel conversion efficiency-0.94)*(Biodiesel conversion efficiency-0.94)	0.3412544		0.1324
(Biodiesel conversion efficiency-0.94)*(Operating rate-295)	–0.000105		0.1848
(Biodiesel conversion efficiency-0.94)*(Capacity-75)	0.0004336		0.1983
(Interest rate-0.04)*(Interest rate-0.04)	4.362823		0.2672
(Biodiesel conversion efficiency-0.94)*(Interest rate-0.04)	–0.482314		0.5538
(Operating rate-295)*(Operating rate-295)	–1.138e-8		0.7325
(Capacity-75)*(Interest rate-0.04)	–0.000309		0.8154
(Feedstock price-150)*(Feedstock price-150)	3.6136e-9		0.8325
(Feedstock price-150)*(Operating rate-295)	3.974e-9		0.8460
(Capacity-75)*(Operating rate-295)	1.4594e-8		0.9140

†Significant at a 95% degree of confidence.

actually negative. Within the range of values used for the LCC estimates, it is unlikely that the cost of jatropha oil will be less than the cost of petroleum diesel because the entire range of its *cost* is higher than the entire range of the petroleum diesel *selling price*. On the other hand, there is an overlap between the costs of fish oil and waste cooking oil biodiesels and the selling price of petroleum diesel. Depending then on prevailing conditions, such as government subsidies that may lower the cost of waste oil feedstock, it is easy to compute the cost for the feedstock and decide if the government subsidy would lower the cost sufficiently below the cost of petroleum diesel.

3 Concluding remarks

In this chapter, we have described how the life-cycle cost (LCC) of a development project may be used to contribute to a life-cycle sustainability assessment (LCSA). As the LCC expands from the financial LCC (fLCC) to the environmental LCC (eLCC) to the social LCC (sLCC), the system boundary expands and starts including activities whose financial equivalents are increasingly uncertain. The Design-of-experiments (DOE) approach to Global Sensitivity Analysis (GSA) provides a simple procedure to systematically determine the sensitivity of the LCC to key parameters, while taking into account potential interactions. By doing so, a more intuitive grasp of the relative importance of each of the parameters is provided to decision-makers. This approach has been applied here to the case of biodiesel production in Vietnam, but can be readily generalized to a wide range of LCC applications as well.

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CHAPTER 9

Social life cycle assessment of biofuel production

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1 Introduction

Sustainability is now an essential principle in the management of environmental resources where it is increasingly clear to society that the continued use of fossil fuels for energy purposes has become unsustainable. The increasing difficulties and costs of exploiting global oil reserves and the need to reduce greenhouse gas emissions associated with their use around the world are undermining the use of fossil fuels. First generation biofuels which derive from terrestrial crops put a lot of pressure on global food markets, contribute to water scarcity, and accelerate the destruction of forests. The sustainability of biofuels will depend on the development of advanced, sustainable, and commercially viable technologies. Several studies have been conducted on the technical feasibility of growing different types of organisms for the production of biofuels in laboratory, which have proven the absence of many of the major disadvantages associated with current biofuels. It is believed that economic viability is currently the main obstacle to be overcome by still immature biofuel technologies. The issue is not whether advanced biofuels are technically possible but focuses on whether they

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can be produced in a sustainable (environmental, economic, and social) manner and on a scale sufficient to help contribute to the global demand that will require innovative measures and of political and institutional cooperation to reach the solution to this complex challenge (Soares et al., 2018).

According to Živković et al. (2017) in developing any policy on any biofuel, the government should be aware of the socioeconomic, health, and environmental effects that its implementation will have on people because the efforts made for biofuel promotion can have feedback effects and other consequences which impose additional costs on society. In addition, increasing demand for biodiesel requires large arable areas for planting energy crops. However, a major change in land use can contribute to generating negative economic, social, and environmental impacts. While it appears that biofuel policies are effective in supporting domestic farmers, effectiveness in meeting climate change and energy security goals is under constant review. At the moment, it appears that biodiesel is an expensive form of greenhouse gas reduction, especially if all subsidies are considered. From the social point of view, several important issues should be carefully considered. As, for example, the connections of biodiesel with the energy and food markets, the possibility of improving working conditions and workers' rights, the effects of biodiesel policy, and the relations between social and economic impacts of its production with special attention to effects of feedback and other undesirable consequences of the biodiesel sector.

For Gonzalez-Salazar et al. (2016) among the different sustainable energy resources, a particularly interesting one for the industrialized countries as well as for the emerging and developing countries is biomass. Biomass is now the largest renewable resource and global interest in its sustainable use, as well as the potential to reduce dependence on fossil fuels and reduce greenhouse gas emissions continues to grow. In recent years, several industrialized countries and emerging economies have developed roadmaps for the exploitation of biomass resources and the deployment of bioenergy technologies. Examples include global technology roadmaps on: biofuels for transport and bioenergy for heat and energy (European Union); biomass and biofuel technologies for transport and biogas (United States); bioenergy and biofuels and biofuels for algae, sustainable aviation biofuels (Brazil), biomass energy technologies and rural biomass energy (China); a roadmap for biorefineries in Germany, among others. However, despite the vast potential and significant demand for bioenergy, the implementation of technological roadmaps for the exploitation of bioenergy in developing countries has been scarce. Providing bioenergy and biomaterials through biorefineries seems to be an inevitable approach in the coming decades. Therefore finding the right raw materials and adequate processing options is critical to future sustainable solutions (Khoshnevisan et al., 2018).

According to Valente et al. (2018), biofuels and biomaterials of the first generation of agricultural crops are strictly dependent on the costs of raw materials and the energy market and compete with other uses of biomass. Thus by acquiring the cheapest raw material, it is not uncommon to import from developing countries causing social, economic, and environmental problems. In contrast, second generation biofuels from lignocellulosic materials have the advantage of using low value raw materials such as waste, small diameter trees, or even dedicated crops. These materials consume less resources than those used by first generation processes, thus enabling a more sustainable supply chain.

2 Social life cycle assessment (SLCA)

The UNEP/SETAC (United Nations Environment Program)/Life Cycle Initiative began with a focus on the environmental LCA (Life Cycle Assessment) and continued its work on the perspective of sustainable development. A major initial contribution was the publication of the SLCA Guidelines (UNEP/SETAC, 2009). The need to integrate the LCA with the social aspects that led to the SLCA dates back 17 years. Since then, there is certainly much greater interest in the social impacts of products, in order to promote sustainability. SLCA can be defined as an engineering tool dedicated to the analysis and evaluation of the effects caused by changes in the life cycle of a product or service. The tool assesses the social impacts that are the subject of the study of sociology and, with the science of management, belong to the domain of human and social sciences and presented the state of the art of SLCA that intends to quantify the social impacts on the complete life cycle (Dreyer et al., 2006; Iofrida et al., 2016; Lehmann et al., 2011; Mattioda et al., 2015; Petti et al., 2018; Sonnemann et al., 2015; Valdivia et al., 2013).

In the LCA community, based on the context of the triple bottom line, Klöepffer (2008) basically stated that to achieve or assess sustainability, environmental, economic, and social aspects have to be adjusted and controlled against each other, and proposes the scheme LCSA (life cycle costing) + SLCA, where society depends on the economy, and the economy depends on the global ecosystem. The LCSA is an effective tool to support the product development process in order to consider all aspects of eco-design in order to reduce environmental, social, and economic impacts from a life cycle perspective.

According to Ekener-Petersen et al. (2014) in a study on biofuels on ethanol (Brazilian sugar cane, French wheat, French corn, and US corn), and rapeseed biodiesel originating in Lithuania, chosen as typical fuels used in the European Union and Sweden clearly shows that there are risks of substantial negative social impacts of fossil fuels at the same levels as biofuels. High or very high risks of negative social impacts are present for all fuel types included in this study. For Ren et al. (2015), SLCA aims to evaluate a multitude of impacts, ranging from direct impacts on workers to broader LCAs widely used by society. It investigates social performance at the sector/industrial level, and there is still little experience with its use and implementation.

In the systematic review of the literature by Petti et al. (2018), in which 35 case studies on SLCA were considered, in the period 2010 to 2015, publications were distributed in the following sectors: manufacturing (26%), agriculture (26%), waste management (21%), energy (24%)—including photovoltaic and biofuel—and tourism (3%). It was hoped that the scope of the most interested sectors would be high-risk social and socioeconomic problems, but the sectors analyzed seem to be areas with a strong environmental aspect. This is probably due to the fact that 48% of the SLCA case studies are implemented in developing economies (Africa 15%, Asia 25%, and South America 8%), while 46% in developed economies.

For Spierling et al. (2018) the SLCA is a fairly young field of research compared to the assessment of ecological impacts of value chains via LCA and has been less focused during the last decades of the assessment of life cycle sustainability. This can be explained by the perception of ecological aspects and by the complexity of social and economic issues and their interdependencies.

According to De Luca et al. (2017), SLCA is dedicated to assessing all kinds of life cycle impacts that affect people. This methodology has not yet been standardized. There is no consensus on the evaluation process and there are no unique definitions for SLCA and social impacts. This has led to a myriad of methodological proposals that differ in many respects, such as the evaluation perspective, the sources of impacts and what is worth evaluating (the "impact categories" as referred to in the ACL terminology), as well as the epistemological foundations.

For SLCA, data are collected through on-site observations and interviews with relevant stakeholders. Questionnaires are often used for data collection. In addition, some databases have been established, for example, the Global Trade Analysis Project database and the Social Hotspot Database (SHDB). However, results with generic data from statistical databases are approximate and site-specific data may reflect social impacts more accurately (Zhou et al., 2018). According to Živković et al. (2017), a comprehensive SLCA is not yet possible and improvement is suggested through the development of a universal set of indicators, databases for social aspects, and well-functioning models. Contreras-Lisperguer et al. (2018), indicate the indicators that may suggest a positive potential, depending on the introduction of a relevant organizational policy and/or the type of certification requested. These are (i) Number/percentage of injuries, illnesses, and fatal accidents in the organization by qualification of work within the company. (ii) Presence of formal policies on equal opportunities (working conditions). (iii) Lower paid workers compared to the country's minimum wage (Working Conditions). (iv) Has the organization developed a project-related infrastructure with mutual access and benefit to the community? (v) Strength of training and (re)qualification policies and practices (duration and type of training/qualifications plus eligibility by age, experience, qualifications, local life); (vi) strength of the organizational risk assessment in relation to the potential of conflict of material resources. (vii) The employment is not conditioned by any restrictions on the right to collective bargaining (Working Conditions). (viii) Workers voluntarily agree on the terms of employment. Employment contracts stipulate salary, working time, vacation, and terms of waiver. Work contracts are understandable to the workers and kept on file. (ix) Absence of underage child workers (Human Rights—Child Labor). (x) Policies/organizational efforts to reduce unpaid time spent by women and children who collect biomass. (xi) Local mortality rates and disease burden attributable to indoor smoke.

According to Valente et al. (2018), scientific research on the theoretical framework on SLCA has increased, but many methods and different approaches are currently available, leading to the subjective interpretation of the results. It is still difficult for practitioners to understand how to conduct an assessment and few empirical examples are available. SLCA is still in its infancy and needs to be applied to develop best practices. The two methodologies addressed the performance of bioethanol and biochemical production in two different dimensions (environmental and social), and their combination allows to achieve results that integrate the product-specific approach of the locality.

According to Di Cesare et al. (2018), social impact assessment is increasingly important in business and public policy contexts. Indeed, as key challenges for sustainable development, the United Nations Millennium Development Goals cover global social issues, from halving extreme poverty rates to halting the spread of HIV/AIDS and providing universal primary education. In addition, composite indicators such as the Human Development Index (HDI) and other sets of indicators are important for measuring progress toward sustainability, including social aspects, in order to understand how sociopolitical and economic systems are developing. However, the challenges in social impact assessment are related to the intrinsic difficulties of unanimously defining what is socially desirable and acceptable. According to the authors, focusing on social aspects, it is clear that not only the negative impacts are of interest, but also the positive impacts that can derive from a specific human intervention. When researching appropriate indicators for sustainability assessment, consideration should be given to its ability to guide policies and decisions at all levels of society (town, city, county, state, region, nation, continent, and world). Indeed, in the context of public policies, social indicators are an important tool for assessing the level of social development of countries and for assessing the current impact and where the research should focus.

2.1 Social aspects and stakeholders in the production of biofuels

According to Freeman (1984), over the years, companies have been changing their level of complexity and the external environment has become a concern as well. For the author, the managers need to define new strategies for each type of group that has influence on the income of the organizations, demanding a better understanding about the identity and characteristics of this group, known as stakeholders. As Moore (2001) reveals, in the midst of many concepts, the literature considers stakeholder employees, suppliers, shareholders, clients, and the community. For Donaldson and Preston (1995), stakeholders are people or groups of people with legitimate interests in the processes or impacts of the company's activities.

In this context, according to Iofrida et al. (2016), the SLCA was conceptualized in an engineering environment, since it is the main field of study for scholars and professionals of the environmental life cycle assessment (ELCA). However, the inherent nature of impacts on assessment is different in SLCA than in ELCA, since it is designed to analyze environmental impacts (linked to the natural sciences) and the first to analyze social impacts (belonging to the field of science social). The disciplinary and scientific heritage of SLCA has been discussed and tracked in sociology and administration science.

According to Zhou et al. (2018), life cycle assessment is usually adopted to assess the environmental burden associated with energy wastage initiatives. From a life cycle perspective, it is possible to attribute the SLCA to the disciplinary field of management sciences, since it can help organizations make decisions about how to organize their processes according to the social impacts of their products or services. This is confirmed in many SLCA studies, emphasizing the role of supporting management practices for sustainability at different levels: operational decisions, strategic decisions, and communication purposes.

Stakeholder theory proposes a rewriting of the traditional corporate purpose of providing dividends to shareholders and postulates that companies should address all stakeholder interests, such as "individuals and voters who voluntarily or involuntarily contribute to the capacity and wealth creation activities and thus are potential beneficiaries and/or risk-takers." Despite extensive research, stakeholders show a weakness in terms of applying an integrated vision for the analysis of environmental and social issues. Practical examples of the use of stakeholder theory to propose holistic solutions related to poverty and environmental deterioration are scarce. An interpretation of stakeholder theory is that any company will have a series of explicit or implicit claims from its many stakeholders. Emphasizing the importance of long-term success, stakeholder theory suggests that firms cultivate relationships with their stakeholders and integrate those relationships into a comprehensive management strategy. However, there are limits on the financial resources available to companies to meet the implicit claims of stakeholders (De-Burgos-Jiménez et al., 2011).

Stakeholder theory is a body of research that has emerged in the last 20 years through scholars in management, business and society, and business ethics in which stakeholder thinking plays a crucial role. Two basic assumptions are discussed by stakeholder theorists:

i. For good performance, managers need to pay attention to a wide variety of stakeholders.

ii. Managers are obligated to stakeholders, which include but go beyond shareholders.

The assessment of social impacts is very complicated and easily overlaps with environmental impacts (impacts on human health, for example) and economic impacts (job creation and income from work). Stakeholders in the SLCA can be workers /employees, local communities (national and international) companies, consumers (end-use or supply chain), value chain actors and other groups, such as nongovernmental organizations (NGOs) and public/state authorities. (Nguyen et al., 2017). However, in the literature we can find other stakeholders such as clients, shareholders/owners, creditors, resource agency and environment (environmental protection), competitors, trade unions (specific groups), media and consumer advocacy (Freeman, 1984; Zhao et al., 2012).

The social and socioeconomic subcategories were defined according to best practices at the international level: international instruments, initiatives of corporate social responsibility, legal framework model, and evaluation of the literature of social impacts. The following are the 31 subcategories associated with five categories of stakeholders: workers, local community, society, consumers, and actors of the value chain (Mattioda et al., 2017).

(a) Workers: Freedom of association and collective negotiation; child labor; fair wage; work hours; forced labor; equal opportunities/discrimination; health and safety; social benefits/social security.

(b) Local community: Access to material resources; access to immaterial resources; delocalization and migration; cultural heritage; safe and healthy living conditions; enforcement of indigenous rights; community participation; local employment; secure living conditions.

(c) Society: Public commitments to sustainability issues; contribution to economic development; prevention and mitigation of armed conflicts; development of technology; corruption.

(d) Consumer: Health and safety; feedback mechanism; consumer privacy; transparency; end-of-life responsibility.

(e) Actors of value chain: Fair competition; relationship with suppliers; enforcement of intellectual property rights.

The measurability of social conditions and socioeconomic impacts according to De Rosa (2018) is more imprecise than the quantification of physical phenomena. Current environmental impact assessments include indirectly some social aspects, such as the impact on human health and/or welfare (a social issue) caused by pollution (environmental problem). There are additional challenges to systematically quantify social impacts: LCA data at scale on social performance are rare; the LCIA of social issues involves, to a large extent, choices based on value. Although there are some guidelines and methodological structures the SLCA is currently not widely realized. Theoretically, the stakeholder categories (and their subcategories) evaluated by a SLCA by the UNEP/SETAC guidelines (UNEP, 2009) or the infringements list in Weidema (2006).

2.2 Cases of SLCA of biofuel

After a literature review of recent and relevant publications on the SLCA and biofuels themes, 9 articles were selected where their descriptive syntheses are as follows (Table 9.1):

- Contreras-Lisperguer et al. (2018): This paper builds on the findings of a case study on electricity generation through cogeneration in Jamaica in which it analyzes two scenarios: baseline assessing the impact of cogeneration technology already installed in a Jamaican and the second considers that the cogeneration technology is changed to a new biomass-based generation plant that updates the cogeneration stage in order to produce energy from the bagasse. The evaluation was carried out using a complete LCA, LCC, and SLCA. The results showed that the generation of electricity from bagasse-derived cogeneration is an adequate alternative aggregating economic, environmental, and social value.
- Ekener et al. (2018a): A systematic approach to capture positive social impacts of the methodology proposed in the SLCA guidelines has been developed and tested for vehicular fuels (fossil fuels and biofuels). The study addresses the positive social impacts in SLCA and answers the questions about the SLCA methodology, how it can be improved to systematically identify all possible positive impacts on the supply chain, and how positive and negative impacts can be considered. They conclude there are positive social impacts for fossil fuels and renewable for many social aspects in the literature, which can change the overall picture of the social impacts of vehicle fuels. In this way the authors propose a refinement of the SLCA methodology to better capture and add the positive impacts.
- Ekener et al. (2018b): Examine the potential to assess integrated product sustainability performance using the LCSA, including a broad range of social impacts, applying it to selected chains of transportation fuel supply. The methodology developed is tested on biomass-based fuels and fossil transport—ethanol produced from Brazilian sugarcane and US corn, and oil produced from Russian and Nigerian crude oil, where it outlines differences in sustainability performance between products evaluated. The main contribution is the measure taken to integrate the different perspectives of sustainability into a holistic result for sustainability, considering different stakeholder profiles (egalitarian, hierarchical, and individualistic) and negative and positive social impacts. It has been found that the order of classification of transport fuel chains included can change when three different "world views," representing different stakeholder profiles and different priorities between sustainability perspectives, are

Table 9.1 Application of SLCA to biofuels and stakeholders

Authors/ stakeholders	Workers/ employees	Local community/ communities	Society (national and international)	Consumers (in the state of end use or within the supply chain)	Actors of value chain (suppliers)
Contreras-Lisperguer et al. (2018)	X	X			
Ekener et al. (2018a)	X	X	X	X	X
Ekener et al. (2018b)			X		X
Nguyen et al. (2017)	X	X	X	X	X
Parada et al. (2017)	X		X	X	X
Rafiaani et al. (2018)	X	X	X	X	X
Ren et al. (2015)	X	X	X		X
Valente et al. (2018)	X		X	X	
Živković et al. (2017)	X	X	X	X	X

taken into account. This implies that there is no single answer to the more sustainable choice between different alternatives. On the contrary, this depends on different priorities maintained by different stakeholders. An important contribution of this work is the possibility of evaluating the sustainability of both chains of fossil and renewable fuels in the same tool.

– Nguyen et al. (2017): The article develops a precise methodological framework for the estimation of the inclusive impact index (Triple I) based on the current LCSA context. Triple I is a unique quantitative index for sustainability assessment, which is based on the ecological footprint, biocapacity, ecological risk, human risk, cost and benefit under the life cycle approach. It is applied to evaluate the trade-off between advantages and disadvantages of the biodiesel system. In general, this framework can promote the application of triple I in the field of biofuels as it provides several appropriate methods for estimating and suggests that several scenarios need to be taken into account.
– Parada et al. (2017): In this paper, sustainability methods and metrics in current biorefinery project practices are analyzed to identify challenges and opportunities for future improvements in the field. Generally, there is a need for comprehensive analysis that includes social impacts and go beyond the automatic use of metrics for predefined issues. While efforts have been made to develop more comprehensive sustainability analyzes for the biorefinery project, they are often challenged by disciplinary boundaries that generate a narrow scope of analysis and are blind to contextual configurations or stakeholder perspectives. Multi- and transdisciplinary, inclusive and context-aware approaches are identified as opportunities to overcome them in future developments.
– Rafiaani et al. (2018): This study proposes a modified systemic approach to a social sustainability impact assessment of the bio-based economy that serves industry and policy makers to gain a better insight into the importance of assessing the impacts of social sustainability within the economy. The proposed approach follows the four general iteractive SLCA steps and considers the possible social impacts on local communities, workers, and consumers as the three major stakeholder groups. The review shows that the most common social indicators for inventory analysis within the bio-based economy include health and safety, food security, income, employment, land and worker concerns, energy security, profitability, and gender issues. The proposed systemic approach allows integrating the social impacts that are highly valued by affected stakeholders in existing sustainability models that focus only on environmental and technological aspects.

- Ren et al. (2015): This study uses a MCDM methodology for LCSA applied to an illustrative case about three alternatives of bioethanol production in China (wheat-based, maize-based, and cassava-based). The proposed methodology has the following advantages: (i) The sustainability assessment is concluded from a life cycle perspective; (ii) LCSA integrates LCA, LCC, and SLCA methods that are used to obtain environmental, economic, and social criteria, respectively; (iii) social criteria can be quantified using fuzzy set theory; and (iv) MCDM is used to help decision makers/stakeholders make the right decision about the most sustainable scenario.
- Valente et al. (2018): The study uses the ELCA and SLCA methodologies to test environmental and social indicators related to a future biorefinery considering two possible hypothetical sites, Norway and the United States. Where we present the results of the analysis of social hotspots for the chemical, rubber, and plastic sector in Norway and the United States in a bar graph as a social hotspot index aggregated for five social categories: community infrastructure, governance, health and safety, human rights, and labor rights and decent work. According to the authors, there is a lack of LCA studies on the production of second generation biofuels in biorefineries, and no case study applying SLCA is currently available. Only a bibliographic review of Macombe et al. (2013) was found, considering the SLCA of the production of biofuels at the company, regional, and state levels. They conclude that the ELCA and SLCA allow to highlight the main environmental and social challenges in the production of biochemical compounds. The social hotspot database has potential as a social screening tool, although social indicators are still not well established. For this reason, a specific evaluation is necessary to validate the results in the social dimension.
- Živković et al. (2017) analyze the technological, technical, economic, environmental, social, toxicological, and human health risks of the production and use of biodiesel. They conclude that the environmentally sustainable production of biodiesel requires that sustainability standards cover direct and indirect impacts on the environment, that is, soil, water, and air. Combining technological, economic, social, and environmental issues will increase the benefits of biodiesel and may lead to integrated biorefineries to be able to produce sustainable biodiesel and other valuable chemicals. Government policies will be the main driving force for further increases in biodiesel production. There is a need to increase cooperation between governments and various stakeholders to develop and apply the corresponding sustainability criteria consistently around the world as quickly as possible.

3 Conclusions

The LCA and LCCA tools do not include the social dimension in the sustainability assessment. Because of this limitation, the SLCA method has recently emerged as a methodological approach aimed at assessing social aspects throughout a product's life cycle. This type of analysis is still new and there are few studies that apply SLCA in the biodiesel life cycle. According to Živković et al. (2017), it is not yet possible to conduct a comprehensive SLCA. They suggest its improvement through the development of a universal set of indicators, databases for social aspects, and well-functioning models. Ekener et al. (2018a) state that the most appropriate approach identified is multicriteria decision analysis (MCDA), which responds to several of the demands that SLCA places on the aggregation method and for Ren et al. (2015) MCDM for LCSA allows decision makers/stakeholders to identify the most sustainable scenarios to achieve their goals across multiple alternatives.

According to Ekener et al. (2018a,b), further development of the SLCA is required, and one of the challenges is to establish the corresponding scales for which indicators of positive impacts at different sites can be assessed. The use phase should be included in the SLCA assessments to better capture all relevant positive impacts. It is important to assess the impacts on the SLCA structure by defining them as positive at the beginning of the analysis. The aggregation methods for positive and negative social impacts found in the literature are mainly surveys, questionnaires, and monetization. Because these tools are inconsistent with the preconditions for the SLCA, their usefulness is limited. It is necessary to develop methodologies on how positive impacts can be taken into account, together with the negative impacts on SLCA. According to the authors it is important to evaluate the positive impacts separately in future efforts of the SLCA in order to clearly distinguish their contribution to the total social impact. This can inform future actions to improve these positive social impacts and not just to mitigate the negative impacts.

According to Rafiaani et al. (2018) there is still no methodology that covers all social aspects, as it depends on the scope of the study, the availability of data, and the priorities of the stakeholders. While growing, there is still a lack of research on the social impacts of innovative technologies within the biologically based economy. This requires more attention to the need for future direction of research and investments in the social concepts of biofuel supply chains.

In relation to biorefineries, Parada et al. (2017) and Valente et al. (2018) describe that sustainability has been increasingly incorporated into biorefinery design projects; however, social sustainability is often overlooked in project practices, while environmental sustainability is often reduced to analyzing the impacts of global warming, and macroeconomic effects are rarely taken into account. The incorporation of sustainability into the design of biorefinery projects faces the following challenges: inclusion of a comprehensive sustainability analysis that considers social impacts and goes beyond microeconomics and globalization; application during the early stages of the project when data availability is limited; disciplinary limits that limit the scope of the analysis; subjectivity of sustainability, usually disregarded by the use of normative approaches. Social and sustainability methods can be useful to consider the subjectivities of sustainability, particularly through the inclusion of stakeholder perspectives. It has been challenging to select the most appropriate indicators for biorefineries, especially in the social dimension, since social and socioeconomic indicators are not well established and are not specific to biorefineries. For the social dimension, the Social Hotspot Database (SHDB) illustrated its potential as a starting point for screening indicators, but showed some limitations in assessing social risks for assessed sites. It is recommended that stakeholder participation and detailed data for the validation of the results are included, since there are currently no specific factors of social impact characterization available for the biorefinery sector.

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CHAPTER 10

Key issue, challenges, and status quo of models for biofuel supply chain design

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1 Introduction

Biofuel is widely regarded as a sustainable alternative to fossil fuel for reducing Greenhouse Gas (GHG) emissions and enhancing energy security. However, the adoption of biofuel in the global transportation sector is still limited. For example, according to the International Energy Agency (IEA), 90% of transportation biofuel use happens in Brazil, the European Union (EU), China, and the United States (IEA, 2017). A much wider adoption of biofuel is needed to mitigate global warming while meeting the growing energy demand (e.g., high biofuel penetration is needed to achieve IEA 2°C

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scenario where at least a 50% chance of limiting the average global temperature increase to 2°C by 2100) (IEA, 2017).

To accelerate the adoption of biofuel, it is critical to produce and deliver biofuel in a cost-effective, robust, and sustainable way (Yue et al., 2014; Daoutidis et al., 2013; Marquardt et al., 2010). The economic feasibility, environmental impacts, and social implications of biofuel highly depend on the design and operation of the entire biofuel supply chain (BSC) (Awudu and Zhang, 2012; Sims et al., 2010; Iakovou et al., 2010). BSC usually involves a broad range of activities ranging from biomass production and transportation to biorefinery, and distribution to final end-use customers. Therefore it is challenging to design the entire BSC from a system perspective. Compared to traditional supply chains in the manufacturing industries, BSC is more complex given the large uncertainties and variabilities related to feedstocks (Awudu and Zhang, 2012; Santoso et al., 2005; Kim et al., 2011), conversion technologies (Iakovou et al., 2010; Kim et al., 2011; Rentizelas et al., 2009a; McKendry, 2002), and transportation network (An et al., 2011a).

Intensive efforts have been made in the past decades by researchers and the industry to develop optimization and simulation models for BSC to solve practical design problems in producing and delivering biofuel. This chapter reviews the literature related to BSC design to highlight the key issues, challenges, and status quo in BSC design.

2 Structure of BSC

BSC consists of three basic stages, including biomass production, biomass conversion and biofuel production, and distribution to customers. This section briefly discusses the major features and technology options in each stage of BSC.

2.1 Biomass production

Biomass production includes activities related to the cultivation and harvest of biomass, transportation and storage, and preprocessing. The first three activities are included in almost all BSC design cases, while preprocessing is optional. The common types of preprocessing include torrefaction and pelletization, and they have been explored in different BSC design cases (Bergman and Kiel, 2005; Pirraglia et al., 2013; Lamers et al., 2015; Kenney et al., 2013). Previous studies showed that adopting preprocessing sites in BSC may improve the logistic stability, feedstock quality, and biorefinery performance (Yue et al., 2014). The selection and design of different biomass production activities are highly subject to the types of biomass feedstocks given their differences in cultivation requirements (e.g., climate and soil) and regional availability. It is noticeable that one of the preprocessing intermediates, pellets, can be directly used as energy products (e.g., for combustion in cogeneration heat and power plant) (Cherubini, 2010; Lam et al., 2010). However, pellets from preprocessing were not considered as final products in most of previous studies.

Generally, biomass feedstocks can be categorized into five types: agriculture, forestry, industry (e.g., industrial waste), household (e.g., municipal wastes), and aquaculture (Cherubini, 2010). The types of biomass feedstock have been evolving in the past decades. The first-generation biofuel is mostly produced from agricultural biomass, such as ethanol from corn and sugarcane (Aden et al., 2002; Goldemberg et al., 2008), and biodiesel from vegetable oil (Mohan et al., 2006; Ekşioğlu et al., 2009). The first-generation biofuel has been commercialized and almost 50 billion liters are produced annually (Naik et al., 2010). The concerns related to the competition with food and relatively land use change (Dutta et al., 2014) lead to the development of the second-generation biofuel feedstocks that largely refer to lignocellulosic biomass (Zhang et al., 2013; Cambero et al., 2016). The common types of lignocellulosic feedstocks include by-products and wastes from the agriculture and forest sector (e.g., corn stover, bagasse, forest residues), wastes (e.g., municipal solid wastes), and dedicated feedstocks (e.g., energy crops) (Sims et al., 2010). The third-generation biofuel mainly uses algae, a feedstock that has potential advantages over previous generations of biomass feedstocks such as high lipid productivities, little competition for arable land, year-round cultivation in wastewater or sea (Moody et al., 2014). With the advancement in metabolic engineering, the fourth-generation focuses on producing biofuels from oxygenic photosynthetic organisms (Lü et al., 2011).

The type of biomass has a large impact on the overall BSC design. It determines the quality and quantity of biomass feedstocks at different regions and time, affecting many decisions related to planning, scheduling, and design of BSC. For example, preprocessing might be advantageous for biomass with high moisture content to reduce transportation cost. The quality and quantity of biomass fed into biorefinery need to be carefully investigated to ensure the overall effectiveness of BSC (Kenney et al., 2014; Wells et al., 2016).

2.2 Biomass conversion to biofuel

Biomass conversion is a critical stage in the BSC to convert biomass feedstocks to biofuels and/or biochemicals. Biomass conversion happens in biorefineries, which are analogous to petroleum refineries that typically produce multiple fuels and chemicals from petroleum crude. According to IEA Bioenergy Task 42, biorefining is "sustainable processing of biomass into a spectrum of marketable products and energy" (Cherubini, 2010; Cherubini et al., 2007). There is a wide range of technologies to convert different types of biomass feedstocks into building blocks and then into value-added products. Those technologies can be categorized into two main types: biochemical and thermochemical (Cherubini, 2010; An et al., 2011b).

Common biochemical processes in biofuel production include fermentation and anaerobic digestion. Fermentation process employs microorganisms to convert sugars and starch into recoverable products (e.g., ethanol). For sugar-based (e.g., sugarcane and sweet sorghum) and starch-based biomass (e.g., corn grain), minimum pretreatment is needed for size reduction and extraction. For lignocellulosic biomass (e.g., corn stover, woody biomass), pretreatment and enzymatic hydrolysis are necessary to obtain sugars (Cherubini, 2010; Humbird et al., 2017). Fermentation time and temperature can vary based on different microorganisms used (Dutta et al., 2011). After distillation, bioethanol usually can be either for further production (e.g., polyethylene, Mohsenzadeh et al., 2017) or sent to be blended with conventional gasoline [e.g., E15 (Romano and Zhang, 2008)]. Choosing the bioethanol pathway can influence BSC design in several aspects. On the supplier side, as sugar- and starch-based biomass are feedstocks in the food industry, biomass supply and price can be in equilibrium with the food market, which may bring in more uncertainties (Bai et al., 2012). On the end-use side, as bioethanol can be blended with gasoline, the market demand and price can relate to vehicle fuel market (Wang et al., 2013).

Anaerobic digestion breaks down biodegradable biomass by bacteria in anaerobic ambient (Cherubini, 2010; Sharma et al., 2013). The digestion temperature usually ranges from 30 to 65°C. One common product of these processes is biogas that can be upgraded to biomethane, an alternative to natural gas (Romano and Zhang, 2008). Biogas production can affect the BSC design mainly in two aspects. The transportation network and supply chain needs to be carefully designed for decentralized (e.g., farm-based) or centralized (e.g., industrial scale) biogas digestors (Ma et al., 2005). At the same time, biomethane after upgradation can be stored and distributed through existing natural gas infrustructure, which may influence the selection of biogas plant locations, inventory planning, and end distribution (Höhn et al., 2014).

Thermochemical conversion of biomass is a high-temperature process. Pyrolysis and gasification are the most common types of thermochemical conversion (Patel et al., 2016). Various types of feedstocks can be used, including energy crops, crop residues, woody biomass, municipal solid waste (MSW) (Dutta et al., 2011; Ringer et al., 2006). Pyrolysis is usually operated around 500°C in oxygen absence environment (Graham et al., 1984). For effective heat transfer during the pyrolysis, the size of biomass needs to be reduced depending on reactor size and design (Bridgwater et al., 1999) and the particles are dried to the moisture content of 5–10 wt% (Ringer et al., 2006). This requirement can affect the BSC configuration as decentralized preprocessing sites may be used to enhance feedstock quality and reduce transportation cost (Lamers et al., 2015; You et al., 2012). The primary products, bio-oil, can be used in combustion (Wornat et al., 1994) and biodiesel blending (Solantausta et al., 1993). Bio-oil can be further upgraded to drop-in biofuel by deoxygenation and reforming the remaining hydrocarbons (e.g., by catalytic cracking and hydrotreating) (Diebold and Scahill, 1987; Baker and Elliott, 1987). Some biofuels can be used as "drop-in" fuels that take advantage of existing petroleum infrastructure (An et al., 2011a; Tong et al., 2013).

Gasification process decomposes biomass in a limited oxygen environment to syngas (CO and H_2), tars, and char with a temperature range of 700–900°C (Dutta et al., 2011). Following is the gas cleanup process where tars, methane, and other hydrocarbons are reformed to CO and H_2 when particulates and other contaminants are quenched out (McKendry, 2002). Further processes lowering sulfur and acid levelmay also be used. The syngas in BSC can be further used by Fischer-Tropsch (FT) synthesis to produce hydrocarbon fuels and alcohol synthesis to produce bioethanol (Dutta et al., 2011; Liu et al., 2014; Tong et al., 2014). Gasification process and syngas cleaning up process can be designed at different locations or integrated into one biorefinery (Li and Hu, 2014).

In addition to two main types of biomass conversion technologies, several other processes have been developed in the past decades. One of the commercially successful processes is transesterification to produce biodiesel from oil crops, fat, waste oil, algae, and cyanobacteria, where fatty acids and triglycerides are chemically converted to fatty acid alkyl esters used as biodiesel (Fjerbaek et al., 2009; Ma and Hanna, 1999). With catalysts (e.g., acids, alkali, enzymes), reactions happen in the temperature of 35–90°C (Fjerbaek et al., 2009). For this conversion pathway, preprocessing sites are needed in BSC design. For example, oil crop feedstocks (e.g., soybean and canola) in grain form need to be preprocessed to extract the vegetable oil in crushers or extractors (Leão et al., 2011). Preprocessing may not be needed for algae as oil is exacted from algae in an extraction chamber with chloroform methanol before transesterification (Lü et al., 2011). Alternatively, direct transesterification can be used for either dry or wet algae with catalysts and solvent (e.g., chloroform, hexane, petroleum ether, methanol, H_2SO_4) (Johnson and Wen, 2009). With technology options for algae, optimizing the technology variables and considering the uncertainties existing in production stages are crucial (Wang et al., 2015).

Mechanical processes are typically processes without changing the state or components of materials. A common type of mechanical process is pelletization (Cherubini, 2010). Although pellet can be burned for energy, pellets were not considered as final products in most of previous studies for BSC design. Fig. 10.1 shows common biomass feedstock type, conversion technology, and end product categories in the BSC.

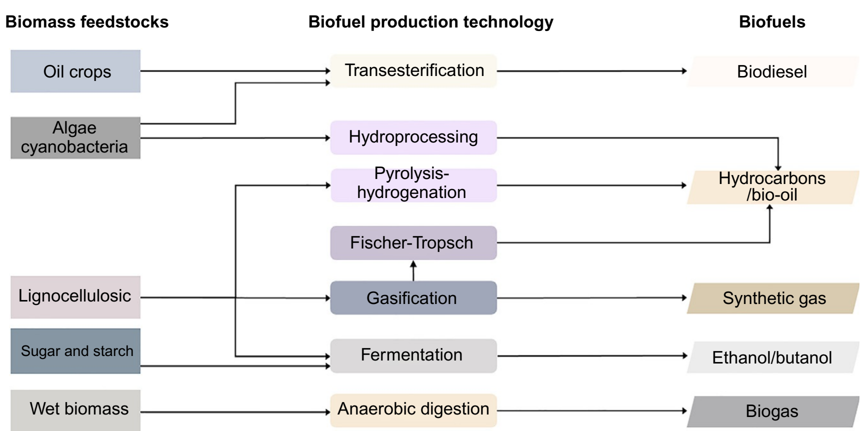

Fig. 10.1 Categories of main process technologies and biofuel types (Yue et al., 2014; Sharma et al., 2013).

2.3 Biofuel distribution and end use

Biofuel distribution and end use is a critical step for biofuel to reach the market and replace their fossil counterparts. In BSC design, this is the stage that involves distribution network and significantly affected by regional fuel demands (Yue et al., 2014).

Depending on the type of biofuels, infrastructures, and regional policies, biofuels are distributed to end-use customers through different infrastructures. For example, in the United States, B20 is the common biodiesel blending, B100 and other higher or lower blends are not common due to the lack of regulatory incentives and price (Alleman et al., 2016). Most of biodiesel can be distributed from biorefineries to fuel terminals and wholesalers by truck, train, or barge, while B5 is sometimes shipped by pipeline (Alleman et al., 2016; U.S. Department of Energy, n.d.-a). From an end-use perspective, not all fueling stations are able to fuel biodiesel (U.S. Department of Energy, n.d.-b). To promote the adoption of biofuel, it is expected to develop and blend biofuels in a way with minimum changes to vehicle stocks and distribution infrastructures. Table 10.1 presents an overview of different biofuels'

Table 10.1 Compatibility of biofuels to existing infrastructure (Eisentraut et al., 2011; EBTP, n.d.)

Biofuel	Blending compatibility features
Sugar-based ethanol Starch-based ethanol Cellulosic ethanol	E10–E15 (E25 in Brazil) in conventional gasoline vehicles; E85–E100 in flex-fuel vehicles or ethanol vehicles
Conventional biodiesel	Up to B20 in conventional diesel engines
Hydrotreated vegetable oil Fischer-Tropsch diesel Sugar-based diesel/jet fuel	Fully compatible
Algae oil based biodiesel/jet fuel	Fully compatible after hydrotreating
Biogas Bio synthetic gas	Fully compatible with natural gas vehicles and fueling infrastructure after upgrading
Bio-butanol	Use in gasoline vehicles in blends up to 85%
Dimethylether	Compatible with LPG infrastructure
Methanol	10%–20% blends in gasoline; blend up to 85% in flex-fuel vehicles

blending characteristics and their compatibility with the current infrastructure of fossil fuels (Eisentraut et al., 2011; European Biofuels Technology Platform (EBTP), n.d.). In BSC design, the compatibility of biofuel to existing infrastructure needs to be considered and investigated.

3 Multiple decision levels in BSC modeling

BSC design involves a large number of decisions related to biomass production, biomass conversion, and biofuel production and distributions to end-use customers. Depending on the complexity, scales (e.g., different geographic and temporal scales), objectives (e.g., maximize economic benefits), different decisions need to be made during BSC design. Based on decision timeframe and complexity, those decisions can be categorized into strategic, tactical, and operational decisions.

3.1 Strategic decisions

Strategic decisions commonly refer to long-term decisions that are hard to be changed or modified in a short period (e.g., supply chain network design, locations of biorefineries, the number of preprocessing plants) (Yue et al., 2014). Generally, strategic decisions in BSC design include the following aspects:

- *Resource utilization and allocation*. Selecting suitable types of biomass and allocating the resources to meet the demand of biorefineries are critical to design robust and cost-effective BSC (Sharma et al., 2013). Those decisions commonly involve selecting biomass types (Avami, 2013), allocating biomass resources (Akgul et al., 2012a), and selecting biomass supply sites (Lin et al., 2012; Palak et al., 2014). Those decisions are typically made based on factors related to biomass prices, biomass availability, and feedstock quality.
- *Supply chain network design*. Transporting low energy density biomass across large areas can be expensive and time-consuming, making the transportation network design crucial to overall effectiveness of BSC (Yue and You, 2016). For transportation network design, decisions need to be made on selecting biomass suppliers, locations, capacities of each operational facility (e.g., preprocessing plants and biorefineries), transportation modes, and related distribution channels. The design of a transportation network always needs to be tailored to regional contexts given the large variation in infrastructures (e.g., road routes) and vehicles (e.g., load limits).

- *Technology selection*. Given a large number of technology options available for biomass preprocessing and conversion, decisions need to made on selecting suitable technologies for specific biomass (Yue et al., 2014). Technology selections and following process design are often subject to economic and technical constraints such as biomass properties, budget limitations, and targeted products (e.g., upgrading to biofuels or high value-added biochemicals) (Kim et al., 2011; Leão et al., 2011; Parker et al., 2010; You and Wang, 2011; Zhang and Wright, 2014; Cambero and Sowlati, 2016).

Table 10.2 presents literature identified that have considered strategic decisions, and tactical and operational decisions (discussed in the next section). Almost all previous BCS design studies considered strategic decisions and different modeling approaches have been used. For example, Avami (2013) determines biomass source utilization and allocation decisions using linear programming (LP). Zhang and Wright (2014) selected different technology options and decided locations and capacities of biorefineries using mixed integer nonlinear programming (MINLP).

3.2 Tactical and operational decisions

Tactical and operational decisions are the medium- and short-term decisions which can be alternated annually, weekly, or even daily (Awudu and Zhang, 2012). They are made under the constrained structure of strategic decisions that are typically made before investigating tactical and operational decisions (De Meyer et al., 2014). Compared with strategic decisions, tactical and operational decisions are typically made at a smaller scale(e.g., biorefinery level or process level). Tactical and operational decisions may include the following aspects:

- *Production planning* determines detailed design and operations of unit processes included in BSC, such as supplying biomass and other raw materials (Zhang and Hu, 2013), process design (Tong et al., 2013; Kazemzadeh and Hu, 2013), and scheduling (Sharma et al., 2013; Beamon, 1998).
- *Inventory planning* determines the quantity and timing of materials or goods in stock (Cachon and Fisher, 2000; Stadtler, 2005; Min and Zhou, 2002) which needs to be aligned with production capacity, fuel distribution, and biomass supply (Tong et al., 2013; You and Wang, 2011; Azadeh et al., 2014). The storage contains raw materials for manufacturing, intermediate productions, and final product for distribution.

Table 10.2 The supply chain decisions in reviewed articles

References	Strategic decisions	Tactical and operational decisions
Bai et al. (2011)	Locations and capacities of processing sites	N/A
Cambero and Sowlati (2016)	Network design	
Leão et al. (2011)	Production technology selections; locations and capacities of preprocessing site	Production planning
Zhang and Hu (2013)	Locations and capacities of processing sites; biomass sourcing	Inventory planning; logistic management
Huang et al. (2014)	Locations and capacities of processing sites; locations and capacities of inventory sites	Production planning; inventory planning
Bowling et al. (2011)	Locations and capacities of processing sites	Production planning
Jonker et al. (2016)	Locations and capacities of processing sites	N/A
Huang et al. (2010)	Locations and capacities of processing sites; production technology selections; transportation mode; locations and capacities of inventory sites	
Avami (2013)	Biomass sourcing; production technology selections	
Cambero et al. (2016)	Network design	
Akgul et al. (2012a)	Locations and capacities of processing sites; transportation mode; biomass sourcing	
Zhang et al. (2013)	N/A	Logistic management
Awudu and Zhang (2013)		
Ren et al. (2015)		
Zhang and Wright (2014)	Locations and capacities of processing sites; production technology selections	
Akgul et al. (2012b)	Locations and capacities of processing sites; biomass sourcing; transportation mode	
Kazemzadeh and Hu (2013)	Locations and capacities of processing sites	

Kim et al. (2011)	Locations and capacities of processing sites; production technology selections	
Ekşioğlu et al. (2009)	Locations and capacities of processing sites; network design	
Papapostolou et al. (2011)	Network design	
Marufuzzaman et al. (2014)	Locations and capacities of processing sites; transportation mode	
Hajibabai and Ouyang (2013)	Locations and capacities of processing sites; transportation mode; network design	
Alex Marvin et al. (2012)	Locations and capacities of processing sites	
He-Lambert et al. (2018)	Locations and capacities of processing sites; locations and capacities of preprocessing site; transportation mode	
Kanzian et al. (2013)	Locations and capacities of preprocessing site; transportation mode	
Zhang et al. (2016a)	Locations and capacities of processing sites	
Martínez-Guido et al. (2016)	Locations and capacities of processing sites	
Tong et al. (2013)	Locations and capacities of processing sites; network design; production technology selections	Production planning; inventory planning; logistic management
Lin et al. (2014)	Locations and capacities of processing sites; locations and capacities of inventory sites; locations and capacities of preprocessing site	
Tong et al. (2014)	Locations and capacities of processing sites; network design; production technology selections	
An et al. (2011b)	Locations and capacities of processing sites; production technology selections; transportation mode; locations and capacities of inventory sites	
You et al. (2012)	Locations and capacities of processing sites; locations and capacities of preprocessing site; network design	
Parker et al. (2010)	Locations and capacities of processing sites; production technology selections; network design	

Continued

Table 10.2 The supply chain decisions in reviewed articles—cont'd

References	Strategic decisions	Tactical and operational decisions
Bai et al. (2012)	Locations and capacities of processing sites; network design	N/A
Palak et al. (2014)	Biomass sourcing; transportation mode	
Zhang et al. (2016b)	Locations and capacities of processing sites	
Höhn et al. (2014)		
Dal-Mas et al. (2011)		
Wang et al. (2013)		
Lin et al. (2015)	Locations and capacities of processing sites; locations and capacities of preprocessing site; biomass sourcing; locations and capacities of inventory sites	
Sukumara et al. (2014)	Locations and capacities of processing sites; network design	
Zhang et al. (2014)	Locations and capacities of preprocessing site; locations and capacities of processing sites; transportation mode	
Yilmaz Balaman and Selim (2014)	Biomass sourcing; locations and capacities of processing sites; locations and capacities of inventory sites	
Sammons et al. (2008)	Locations and capacities of processing sites	Inventory planning
Rincón et al. (2015)		Inventory planning; fleet management
Bernardi et al. (2013)	Biomass sourcing; production technology selections; network design	Production planning; logistic management
Zamboni et al. (2009)	Biomass sourcing; locations and capacities of processing sites; transportation mode	
Gebreslassie et al. (2012)	Network design; production technology selections;	
Corsano et al. (2011)	Locations and capacities of processing sites; locations and capacities of inventory sites	
Chen and Fan (2012)	Locations and capacities of processing sites; locations and capacities of distribution sites	

Santibañez-Aguilar et al. (2016) and Santibañez-Aguilar et al. (2014)	Biomass sourcing; production technology selections; locations and capacities of processing sites; network design	Inventory planning
Sammons Jr et al. (2007)	Production technology selections; network design	Production planning; inventory planning; logistic management
Azadeh et al. (2014)	Locations and capacities of processing sites	
Bairamzadeh et al. (2016)	Biomass sourcing; locations and capacities of processing sites;	
Li and Hu (2014)	Locations and capacities of processing sites; locations and capacities of preprocessing sites;	
Xie et al. (2014)	Locations and capacities of processing sites; locations and capacities of preprocessing sites; transportation mode	
Čuček et al. (2012)	Locations and capacities of processing sites; network design	
Wang et al. (2015)	N/A	
You and Wang (2011)	Network design; locations and capacities of processing sites; production technology selections	
Mele et al. (2011)	Locations and capacities of processing sites; locations and capacities of inventory sites; transportation mode	Production planning; logistic management; fleet management

- *Logistic management* refers to managing and implementing the sufficient and effective flows of materials (e.g., raw materials, intermediates, and products), goods, and information from original suppliers to end users (Bowersox, 1997; Lambert and Cooper, 2000; Sokhansanj et al., 2006).
- *Fleet management* decides the movements of materials between different BSC stages (Awudu and Zhang, 2012). Fleet management plays a crucial role in BSC as it directly affects the robustness of the transportation network (Ravula et al., 2008; Eriksson and Björheden, 1989; Van Wassenhove and Pedraza Martinez, 2012; Thomas and Griffin, 1996).

Depending on the predetermined strategic decisions and the objectives of BSC, different tactical and operational decisions mentioned were considered in previous BSC cases (see Table 10.2). Zhang and Hu (2013) built two models to optimize the strategic decisions of facility locations and capacity, then tactical and operational decisions such as monthly biorefinery production planning and inventory control were investigated and determined. Some studies developed optimization approaches to make strategic and tactical decisions simultaneously. For example, Lin et al. (2014) established a model to optimize the large-scale biomass-to-ethanol SC where the strategic (e.g., farm and facility locations and capacities) and tactical decisions (e.g., biomass production planning, plant operating schedules, and inventory control) were optimized simultaneously. An et al. (2011b) established a model considering multiple types of lignocellulosic biomass and the material flows in the BSC. This model could be used for both strategic and tactical decisions including facility locations and capacities, technology types, production plans, transportation strategies, and storage amount. Ekşioğlu et al. (2009) integrated the long-term decisions (e.g., capacity, location, and the number of biorefineries) and mid-term logistic decisions (e.g., biomass supply) (Ekşioğlu et al., 2009). As a BSC usually has a large number of components, determining tactical and operational decisions without considering the uncertainty related to each component may lead to poor performance of the whole SC (Awudu and Zhang, 2012). Different approaches (e.g., stochastic programming and fuzzy logic) have been developed to model and address uncertainties in BSC design, which are further discussed in the following sections.

4 Modeling approaches for BSC design

Three levels of decisions discussed previously are commonly modeled as decision variables; effective modeling techniques to identify optimal solutions of those decisions are critical for BSC design. With the development of computer science and mathematical programming theories, increasingly complex BSC models have been developed to address large-scale and multidimensional challenges in BSC design. In this section, two main aspects of modeling techniques for BSC design, optimization and simulation, are discussed and reviewed.

4.1 Optimization models

Optimization is a mathematical process aiming at finding the maximum or minimum value of objective functions which are subject to constraints (Dantzig, 2016; Fang and Puthenpura, 1993). An optimization model consists of three parts, decision variables, constraints, and objective functions. Generally, the constraints can be equalities, inequalities, and integer restrictions. Depending on the mathematical structure and the size of models, different algorithms are needed to solve optimization problems (Fang and Puthenpura, 1993; Rajasekera and Fang, 1991).

4.1.1 Types of BSC optimization objectives

The objective functions in the BSC optimization models are typically related to three aspects of sustainability: economic, environmental, and social implications. Economic effectiveness is the most common objective function in BSC optimization studies (Corsano et al., 2011). In recent years, environmental objective functions such as minimizing GHG emissions and energy footprints and social implications (e.g., job creation) have been included in more and more articles (You et al., 2012; Corsano et al., 2011). Previous studies are reviewed and categorized based on their considerations related to economic, environmental, and social aspects as shown in Table 10.3.

Economic viability is one of the most common objectives used in previous BSC optimization studies. Common indicators related to economic viability used in previous studies include expected net profit, Internal Rate of Return (IRR), and Net Present Value (NPV) (Leão et al., 2011; Palak et al., 2014; Alex Marvin et al., 2012; Dal-Mas et al., 2011). The economic objective function can be designed for whole or part of the BSC (e.g., profit of biorefinery or NPV of the entire supply chain) (Azadeh et al., 2014; Alex Marvin et al., 2012). Depending on the specific economic indicators chosen as the objective functions, economic and process data at different levels need to be collected. For example, using robust optimization, Lin et al. (2012) chose minimizing product unit cost as the objective function and collected the data of fuel price. Ren et al. (2015) took the life cycle cost of the BSC as

Table 10.3 The modeling features of reviewed articles in BSC optimization

References	Biomass type	Products	Model type	Objective/sustainability aspects			Region or country	GIS	Resolution
				Economic	Environmental	Social			
Avami (2013)	Biomass waste and residues	Bioethanol	LP	Min annual cost			Iran		
Jonker et al. (2016)	Sugarcane and eucalyptus	Bioethanol	LP	Min total production cost			Brazil	X	5 by 5 km
Ren et al. (2015)	Corn	Bioethanol	LP	Min life cycle cost			China		
Leão et al. (2011)	Castor oil	Biodiesel	MILP	Min investment cost and min logistical cost			Brazil		
Zhang and Hu (2013)	Corn stover	Drop-in fuel	MILP	Min total production cost			IA, United States	X	County level
Bowling et al. (2011)	Biomass (not specified)	Biodiesel	MILP	Max total profit					
Huang et al. (2010)	Cellulosic biomass and MSW	Bioethanol	MILP	Min total cost			CA, United States	X	County/city level
Zhang et al. (2013)	Switchgrass	Bioethanol	MILP	Min annual cost			ND, United States		
Tong et al. (2013)	Crop residues, energy crops, and wood residues	Gasoline and diesel and jet fuel	MILP and FMP	Min annual cost			IL, United States		
Lin et al. (2014)	Miscanthus	Bioethanol	MILP	Min annual cost			IL, United States	X	County level

Tong et al. (2014)	Crop residues, energy crops, and wood residues	Gasoline and diesel and jet fuel	MILP and FMP	Min unit cost			IL, United States		
An et al. (2011b)	Switchgrass	Bioethanol	MILP	Max the SC profit			TX, United States		
Akgul et al. (2012b)	Wheat, wheat straw, and miscanthus	Bioethanol	MILP	Min the SC cost			United Kingdom		
Kim et al. (2011)	Forestry biomass	Bio-oil	MILP	Max expected SC profit			United States		
Ekşioğlu et al. (2009)	Corn stover and wood residues	Bioethanol	MILP	Min the biofuel cost			MS, United States		
Papapostolou et al. (2011)	Vegetable oil	Biodiesel	MILP	Max the total value			Greece		
Parker et al. (2010)	Crop, forest biomass, animal fats, waste grease, and MSW	Bio-oil	MILP	Max SC profit			United States	X	County level
Palak et al. (2014)	Forestry biomass	Bio-oil	MILP	Min SC cost			United States		
Alex Marvin et al. (2012)	Crop residue	Bioethanol	MILP	Max NPV			United States	X	County level
Dal-Mas et al. (2011)	Corn	Bioethanol	MILP	Max NPV			Italy		
Marufuzzaman et al. (2014)	Biomass (not specified)	Bioethanol	MILP	Min the SC cost			United States	X	County level
Lin et al. (2015)	Miscanthus	Bioethanol	MILP	Min annual production cost			IL, United States	X	County level
He-Lambert et al. (2018)	Switchgrass	Biobutanol	MILP	Max NPV			TN, United States	X	13 km^2 hexagons
Zhang et al. (2016b)	Woody biomass	Biodiesel	MILP	Min the SC cost			MI, United States	X	County level

Continued

Table 10.3 The modeling features of reviewed articles in BSC optimization—cont'd

References	Biomass type	Products	Model type	Objective/sustainability aspects: Economic	Environmental	Social	Region or country	GIS	Resolution
Wang et al. (2015)	Energy crop	Bioethanol	MILP	Min SC cost			ND, United States		
Sammons et al. (2008)	Animal waste	Syngas/ hydrogen	MILP	Max SC profit					
Höhn et al. (2014)	Biowastes, sludges, agricultural residues, and energy crops	Biogas	MILP	Min total distance			Finland	X	<3 by 3 km
Sukumara et al. (2014)	Animal waste, corn stover, and forest residues	Natural gas and gasoline and diesel	MILP	Max profit			KY, United States		
Martínez-Guido et al. (2016)	Sugarcane bagasse	Bioethanol	MILP	Max profit	Min eco-points	Max job creations	Mexico		
Cambero and Sowlati (2016)	Wood residues	Bio-oil and pellets	MOLP	Max NPV	Max GHG saving	Max job creations	Canada		
Cambero et al. (2016)	Wood residues	Bio-oil and pellets	MOLP	Max NPV	Max GHG saving		Canada		
Akgul et al. (2012a)	Wheat, wheat straw, and miscanthus	Bioethanol	MOLP	Min total daily cost	Min GHG		United Kingdom		
You and Wang (2011)	Crop residues, energy crop, and wood residues	Gasoline and diesel	MOMILP	Min annual cost	Min GHG		IA, United States		
You et al. (2012)	Crop residues, energy crop, and wood residues	Bioethanol	MOMILP	Min annual cost	Min life cycle GHG	Max accrued jobs	IL, United States	X	County level

Kanzian et al. (2013)	Woody biomass	–	MOMILP	Max profit	Min CO_2 emissions		Austria	X	1.5 by 1.5 km
Zhang et al. (2014)	Woody biomass	Bio-oil	MILP/ MOMILP	Max 20-year profit	Min GHG		MN, United States	X	County level
Rincón et al. (2015)	Palm oil	Biodiesel	MONLP	Min SC cost	Min GHG		Colombia		
Bernardi et al. (2013)	Corn grain and corn stover	Bioethanol	MOMILP	Max NPV	Min water footprints/min GHG		Italy	X	Around 30 by 30 km
Zamboni et al. (2009)	Corn grain and corn stover	Bioethanol	MOMILP	Min SC cost	Min GHG		Italy	X	Around 30 by 30 km
Santibañez-Aguilar et al. (2016)	Woody biomass, sugarcane, corn grain, sorghum grain, sweet sorghum, african palm, jatropha, and safflower	Bioethanol and biodiesel and butanol and xylitol	MOMILP	Max SC profit	Min eco-points		Mexico		
Santibañez-Aguilar et al. (2014)	Same as above	Bioethanol and biodiesel	MOMILP	Max SC profit	Min eco-points	Max job creations	Mexico		
Mele et al. (2011)	Sugarcane	Bioethanol	MOMILP	Max NPV	Min Eco-indicator 99/GWP100				
Yilmaz Balaman and Selim (2014)	Animal manure and energy crop	Biogas	MOMILP and FMP	Max SC profit	Min weighted unused waste biomass		Turkey		
Bairamzadeh et al. (2016)	Lignocellulosic biomass	Bioethanol	MILP/MO Robust Possibilistic Programming	Max SC profit	Min Eco-indicator 99	Max job creations	Iran		

Continued

Table 10.3 The modeling features of reviewed articles in BSC optimization—cont'd

References	Biomass type	Products	Model type	Objective/sustainability aspects: Economic	Environmental	Social	Region or country	GIS	Resolution
Zhang et al. (2016a)	Waste cooking oil	Biodiesel	HEU	Max profit	Max carbon emission allowance	Max social responsibility	China		
Bai et al. (2011)	Corn and cellulosic biomass	Bioethanol	MILP and HEU	Min total system cost			IL, United States		
Kazemzadeh and Hu (2013)	Biomass	Gasoline	SP and MILP	Max annual profit			IA, United States		
Huang et al. (2014)	Corn stover and forest residues	Bioethanol	SP and MILP	Min expected total system cost			CA, United States	X	County/city level
Awudu and Zhang (2013)	Biomass (not specified)	Bioethanol	SP and MILP	Max expected SC profit			ND, United States		
Azadeh et al. (2014)	Biomass (not specified)	Bioethanol	SP and MILP	Max biorefinery profit			Iran	X	Not specified
Gebreslassie et al. (2012)	Agricultural residues, energy crops, and wood residues	Hydrocarbon fuels	SP and MILP	Min annual cost			IL, United States	X	County level
Chen and Fan (2012)	Legionellosis biomass, forest residues, and MSW	Bioethanol	SP and MILP	Min expected SC cost			CA, United States	X	County/city level

Li and Hu (2014)	Corn grain and corn residues	Gasoline	SP	Max annual profit			IA, United States	X	County level
Bai et al. (2012)	Corn	Bioethanol	MIQP	Max SC profit			IL, United States		
Wang et al. (2013)	Corn	Gasoline and diesel	MIQP with MPEC	Max SC profit			IL, United States		
Corsano et al. (2011)	Sugarcane	Bioethanol	MINLP	Max total net profit					
Zhang and Wright (2014)	Forest residues	Bio-oil	MINLP	Max annual profit			MN, United States	X	County level
Hajibabai and Ouyang (2013)	Biomass (not specified)	Bioethanol	MINLP	Min SC cost			IL, United States	X	County level
Sammons Jr et al. (2007)	Animal waste	Syngas/ hydrogen	MINLP	Max SC profit			United States		
Xie et al. (2014)	Corn stover and forest residues	Bioethanol	MINLP	Min the SC cost			CA, United States	X	Only road information
Čuček et al. (2012)	Corn, corn stover, wood chips, MSW, manure, and timber	Bioethanol	MOMINLP	Max SC profit	Min environmental footprints	Max social footprints	Central European		

the objective function and collected economic data related to each life-cycle stage such as grain cost, transportation cost, and production cost. Economic data collected include costs of cultivating activities (i.e., weeding, sowing, fertilization, pesticides, irrigation, and harvest), transportation, and production (Ren et al., 2015). Another common economic indicator widely used in BSC optimization is NPV (Alex Marvin et al., 2012; He-Lambert et al., 2018; Dal-Mas et al., 2011). Alex Marvin et al. (2012) adopted the NPV as the objective function to optimize the BSC for a biochemical pathway from crop residues to ethanol, where the NPV was calculated from revenue, feedstock cost, transportation expense, and capital investment. As the revenue of biorefineries is subject to fuel selling price, some studies took market equilibrium into consideration (Wang et al., 2013). Wang et al. (2013) used maximizing SC profit as the objective function and considered food market and biofuel market equilibrium as constraints that need data of both food market and blended fuel market. Their results showed that government mandates (e.g., US Energy Independent and Security Act of 2007 could boost biofuel production, while rigid mandates (e.g., mandates without equilibrium constraints in this study) on blenders might depresse the biofuel production in monopoly market (Wang et al., 2013).

As biofuel is considered as a sustainable alternative to fossil-based fuels, environmental benefits and trade-offs with economic objectives are considered in many studies. Life Cycle Analysis (LCA) is one of the most recognized tools to quantify environmental footprints of BSC (Lardon et al., 2009; Gnansounou et al., 2009; Cherubini and Strømman, 2011; Kim and Dale, 2005; Muench and Guenther, 2013; Wang et al., 2007; Singh et al., 2010; Hill et al., 2006). The system boundary of common biofuel LCAs is farm to wheel (adapted from "Well to Wheel," the common system boundary of fossil-based fuels), including biomass cultivation, transportation, production, and end use (in vehicle) (Muench and Guenther, 2013). In previous studies, common LCA indicators include GHG emissions (Hill et al., 2006; Tonini et al., 2016), environmental footprints [e.g., total energy consumption (Wang et al., 2007) and water footprints (Yang et al., 2011)], Life Cycle Environmental Impacts (LCIA) (e.g., eutrophication and acidification) (Lardon et al., 2009; Cherubini and Strømman, 2011), or normalized LCIA indicators such as Eco-indicator 99 (Santibañez-Aguilar et al., 2014).

In literature, LCA has been integrated with BSC optimization either as constraints or objective functions or both. In some studies (Bernardi et al., 2013; Sammons Jr et al., 2007), the results of LCA were used as constraints (e.g., the emissions of several life-cycle stages are set to not to exceed a certain cap). For example, Bernardi et al. set a GHG cap as a constraint based on the European Union's goal of GHG reduction (Bernardi et al., 2013). Sammons Jr et al. (2007) proposed a framework for biorefinery product allocation where the environmental impacts were modeled as constraints after the first stage optimization with an economic objective function. In some studies, the results of LCA were used as one of the multiple objective functions. For example, Akgul et al. (2012a) developed a multiobjective function model for wheat, wheat straw, and miscanthus to bioethanol in the United Kingdom, where two objective functions were minimizing GHG emissions and the SC daily cost. You and Wang (2011) developed an optimal design of the BSC with the economic and environmental criteria considering the variability and seasonality in feedstocks, biomass degradation, and geographic diversity (You and Wang, 2011). Some other studies used a single objective function that integrates LCA with economic analysis or other criteria using Multiple-Criteria Decision Analysis (MCDA) (Kanzian et al., 2013; Bernardi et al., 2013; Eskandarpour et al., 2015). For example, Bernardi et al. adopted the weighted summation of three objective functions, namely, NPV, GHG emissions, and water footprints (Bernardi et al., 2013; Eskandarpour et al., 2015).

As the development of biofuel has potential to create new jobs and thrive the economy in rural areas, social impacts such as job creation have been included in previous studies (Bamufleh et al., 2013; Lira-Barragán et al., 2013). Among different factors that have been used to quantify the social impacts (e.g., indicators on diversity, physical working condition, job creation, and local community acceptance) (Jørgensen et al., 2008), job creation is one of the most common indicators considered in previous optimization studies. The U.S. National Renewable Energy Laboratory (NREL) has developed the jobs and economic development impact (JEDI) models that can quantify the job creations due to the construction and operation of biofuels at local and state levels (NREL, 2012). JEDI was developed based on IMPLAN (economic impact analysis for planning) that uses the input-output method to evaluate three economic impacts of a specific activity, including direct impact (e.g., on-site labor), local revenue and supply chains, and induced effect (e.g., increasing local business due to the development of BSC) (Taylor et al., 1993; Rickman and Schwer, 1995). Some studies integrated JEDI and IMPLAN with optimization models in BSC design to couple the job creation with other objective functions related to economic and environmental benefits (Yue et al., 2014; Ayoub et al., 2009).

Some studies considered the impacts of the job creation at different locations [e.g., more develop regions versus less developed regions (Mota et al., 2015)] or the impacts of different types of jobs created (Cambero and Sowlati, 2016).

As discussed previously, many studies used multiple objectives in optimization models to design sustainable BSC. Most of them have employed MCDA to integrate multiple objective functions into a single objective function using assumed weighting factors (Kanzian et al., 2013; Bernardi et al., 2013; Eskandarpour et al., 2015) or use Pareto curve to explore trade-offs (Zhang et al., 2014; Sammons et al., 2008; Zamboni et al., 2009; Santibañez-Aguilar et al., 2016). Table 10.3 lists 61 studies of BSC optimization reviewed and their objective functions. Among those studies, GHG emissions reduction, job creations, and SC cost are the mostly used indicators representing environmental, social, and economic aspects in BSC optimization.

4.1.2 Types of BSC optimization models

The approaches to solve optimization problems vary largely according to the different types of optimization models. The common types of BSC optimization include LP, mixed integer linear programming (MILP), MINLP, multiobjective linear programming (MOLP), multiobjective mixed integer linear programming (MOMILP), mixed integer quadratic programming (MIQP), stochastic programming (SP), Fuzzy Programming (FMP), and heuristic algorithms (HEU) (Sharma et al., 2013; Mula et al., 2010).

LP has been widely used for optimization problems as a basic approach. LP is "concerned with problems in which a linear objective function in terms of decision variables is to be optimized while a set of linear equations, inequalities, and sign restrictions are imposed on the decision variables as requirements" (Fang and Puthenpura, 1993). Jonker et al. (2016) used the LP approach to optimize the locations and capacities of plants given the expansion of biomass supply regions. MILP is a more common format of BSC optimization models where binary variables are introduced for decisions such as selecting locations, technologies, and other options. Tong et al. (2014) employed a MILP model to integrate the existing petroleum refineries and biomass conversion facilities with considering the uncertainties in production. In this model, integer variables were introduced to represent decision selection, biorefinery property category, and other "whether or not" variables. In some cases, MILP can be computationally intensive, and thus in some studies, two-stage or multistage optimization frameworks were used to address this challenge. Kazemzadeh and Hu (2013) adopted stochastic programming to account for the uncertainties in fuel market price, feedstock, and logistic cost. In this model, two-stage programming with MILP approach was employed where the first stage decided capacity and location of biorefinery and then second stage determined the biomass and gasoline flow.

NLP is used in some BSC optimization models when nonlinear relationships are needed. In the work by Corsano et al. (2011) a MINLP model was established for the sugarcane-based bioethanol SC where the constraints related to fermentation, evaporation, drying, and distillation were modeled by nonlinear equations. Zhang and Wright (2014) proposed a MINLP model to make integrated decisions on production selection, production planning, and facility locations. In their model, technical constraints related to hydroprocessing and reforming processes were modeled as nonlinear. This model was solved in software GAMS/DICOPT with around 30 hours. To reduce computational time in some cases, constraints could be linearized (Zhang and Wright, 2014). For example, many studies divided nonlinear capital cost function into intervals and linearized in each interval (Bowling et al., 2011).

Among the NLP problems, Quadratic Programming (QP) is one special type where the objective function is in quadratic form (Frank and Wolfe, 1956; Imhof, 1961). A typical example is Bai et al. (2012) who developed a BSC design model in quadratic form with consideration of competitive agriculture land use and feedstock market equilibrium.

Stochastic programming (SP) is used in many studies to identify solutions given different sources of uncertainties along the BSC. Given the complexity of uncertainty and intensive computational loads, many studies used mixed-integer multistage stochastic programming. Chen and Fan (2012) established a mixed integer stochastic programming model with two stages: first stage decisions were planning decisions; second stage decisions were operational decisions that consider uncertainties. Gebreslassie et al. (2012) developed a multiperiod stochastic MILP to optimize the hydrocarbon biorefinery SC that modeled feedstock supply and biofuel demand uncertainties in the second stage. Kazemzadeh and Hu (2013) modeled uncertainties of fuel market price, feedstock, and logistic cost as discrete distributions after planning decisions in the first stage. Awudu and Zhang (2013) proposed a model considering the uncertainties in demand, production, and price by using the stochastic MILP that considered the quantity of final products and the initial quantity of feedstocks in the first stage. The product distribution were modeled in the second stage.

Other studies have used fuzzy mathematical programming (FMP) to address uncertainties (Mula et al., 2010). In SP, the uncertainty of independent variables is modeled using probability density function which could be hard to determine due to the lack of data. In FMP, uncertainties are modeled by fuzzy numbers with fuzzy intervals, which could be helpful to characterize uncertain parameters (Tong et al., 2013; Roubens and Teghem, 1991; Inuiguchi and Ramik, 2000; Lodwick et al., 2000). For example, in the MILP model developed by Tong et al. (2013), the data related to the conversion rate from intermediates to end products, capital costs, and operational costs were limited, and the probability functions of feedstock supply and product demand were unkown. In this situation, fuzzy numbers could be helpful in modeling those uncertain parameters with limited data. Another example the MILP model developed by Yilmaz Balaman and Selim (2014) that incorporated fuzzy constraints and fuzzy goal programming to address uncertainty.

4.1.3 Geographical information system (GIS)-based BSC design

Given the intensive computation loads of most optimization models, it is challenging to design BSC at a high spatial resolution (e.g., 10 km by 10 km). Many previous optimization models used low-resolution spatial data (e.g., county level without considering real routes of different transportation infrastructure), as presented in Table 10.3. Geographic Information System (GIS) is a powerful tool specifically used to process, visualize, and analyze geographic data, and it has been used and integrated with optimization models for BSC design. In most of previous studies, GIS has been used to provide geographic information that is needed for parameters in optimization models (Kim et al., 2011; Parker et al., 2010; Bowling et al., 2011; Nardi et al., 2007).

Lin et al. (2015) developed a GIS-enabled model to optimize the BSC using GIS to generate data on biomass availability, production costs, and distances for the optimization model. He-Lambert et al. (2018) proposed a GIS-MILP combined model to decide the biomass supply and facility locations at a high spatial resolution where GIS provided the information of cropland locations and areas, potential plant sites, and road transportation network. Zhang et al. (2016b) developed a GIS and optimization model where GIS was employed to choose biofuel facility locations by inputting the geospatial information including county boundaries, transportation network, water body dispersion, and city locations. In other studies, GIS provided information of biomass availabilities that were used as biomass resource constraints in optimization models (Höhn et al., 2014; Parker et al., 2010).

A few studies used GIS alone as a decision supporting tool for the BSC design. Beccali et al. (2009) used GIS to identify the most exploitable biomass resources in Sicily by analyzing regional economic, agriculture, climate, and infrastructure data. Thomas et al. (2013) employed GIS to assess the spatial supply and demand balance in England at a national scale.

4.2 Simulation-based BSC models

SC simulation is a basic method for SC prediction, management, and improvement (Zhao et al., 2011). Simulation models can support BSC design by employing different performance indicators such as available regional forest fuel potential (NAP), GHG emissions, and SC energy consumption (Zhang et al., 2012; Gronalt and Rauch, 2007). Unlike optimization that determines the optimal values of a set of decision variables, simulation is to model the presence of a system in order to predict the behavior of the system under a given set of conditions (Wurbs, 1993). In some cases, simulation can work as a means of BSC optimization by conducting a large number of BSC design scenarios (Agusdinata et al., 2014).

Compared to BSC optimization models, simulation models can provide a better understanding of the impacts of specific design parameters and strategies. For example, Gronalt and Rauch (2007) proposed a scenario evaluation model to study the regional forest BSC in Austria, where different scenario parameters such as varied transportation distances and varied demands were evaluated by simulating a number of system configurations. Zhang et al. (2012) established a simulation model of converting low-value pulpwood into biofuels and evaluated the impacts of spring break-up on delivered biomass cost, GHG emissions, and energy consumption. They also included other varied design parameters such as cost coefficients, energy coefficients (Btu/ton-mile), biofuel facility locations, and size options. By changing these parameters, a case study was conducted in the lower peninsula of Michigan and showed that simulation model was useful in BSC management, the selection of facility mode, logistic design, inventory management, and information exchange (Zhang et al., 2012). Sokhansanj et al. (2006) simulated the dynamic biomass logistics to predict the transportation cost between operations such as collection, storage, and biorefinery to understand the impacts of biomass availability, moisture content, weather factors, transportation equipment performance, and dry matter loss on the transportation cost.

A powerful simulation tool for BSC design emerged in recent decades is agent-based modeling (ABM). ABM is capable of modeling a system with individuals who have autonomous decision-making abilities (Bonabeau, 2002). Wooldridge (1997) and Jennings (2000) defined ABM as, "an agent is an encapsulated computer system that is situated in some environment and that is capable of flexible, autonomous action in that environment in order to meet its design objectives" (Jennings, 2000). ABM has been largely used in SC management and design (Zhao et al., 2011; Giannakis and Louis, 2011; Nissen, 2001; Lou et al., 2004; Kaihara, 2003; Julka et al., 2002a; Garcia-Flores et al., 2000; Gjerdrum et al., 2001). For example, Zhao et al. (2011) established an ABM model for multistage SC where SC members (e.g., order agent, inventory agent, and distribution agent) were operating autonomy cooperations with each other. The information flow, product flow, and SC member relationship were modeled to observe the dynamics of the SC system. The simulation results were helpful in understanding the impacts of individual members' behavior and organization strategies on the SC. Julka et al. (2002b) applied the ABM technique in traditional refinery SC to support decision-making of refinery manager. Five departments in the refinery were modeled as agents: procurement, sales, operations, storage, and logistics. "What-if" scenarios were developed to understand the behavior rules for each department. Another advantage of ABM in modeling SC is the capability of modeling emergent phenomena under extreme or disruptive events, which can enhance risk management of SC. Giannakis and Louis (2011) investigated the risk management of SC and developed a multiagent model to simulate the operational level SC. Five different agents played varied roles in information integration, coordination, monitoring, and risk management.

In recent decades, ABM has been applied to BSC design to analyze different policy options, design strategies, as well as many "what-if" scenarios. Moncada et al. (2017) developed an agent-based model to study the impacts of agriculture and bioenergy policy in Germany, and they investigated the BSC design strategies such as liberalization of the farmer EU agricultural market, energy tax act, and biofuel quota act. Beck et al. (2008) established a model combining ABM and optimization to study the bioenergy network in South Africa with different policies to understand the trade-offs among economic, social, and environmental aspects. Shastri et al. (2011) applied

ABM to model the system dynamic with farms' and biorefineries' adaptive decisions in the BSC network. By simulating different scenarios (e.g., different policy incentives), the results showed the dynamics of biomass resource adaption and corresponding effects on the biorefinery and crop contracts. Agusdinata et al. (2014) applied ABM to BSC network simulation to study system behaviors of users, biorefineries, and farmers. The results showed that the network was sensitive to the information time delay between different stakeholders. Kempener et al. (2009) used ABM to simulate a complex adaptive system for designing the bioenergy network with combined optimization and simulation approach. This model provided decisions on network performance and impacts of policy enforcement on the BSC. Scheffran and BenDor (2009) developed an ABM model with spatial dynamic powered by GIS to study the BSC and corresponding land use pattern change in Illinois. Results showed that with expanding demand for bioenergy, farmers adopted more energy crops, which further changes the local land use pattern. As ABM can simulate the decision-making of each component of BSC (e.g., farms, inventory sites, biorefineries) and other organizations (e.g., government agencies), the BSC evolution, network performance, trade-offs, and complex system adaptation can be evaluated under different policies, economic, and technical conditions. So far most of previous studies focused on stakeholders such as farms and biorefineries while some included policy incentives from government. Given the complexity of BSC, stakeholders at multiple scales and levels (e.g., government, food companies, manufacturers, farms, customers) may need to be included to better address BSC design problems through ABM. In addition, most of previous ABM studies focused on understanding the economic implications of BSC, a potential use of ABM in the future is to take sustainability of BSC, especially environmental and social sustainability, into consideration (Gold and Seuring, 2011).

5 Challenges and issues in BSC design

5.1 Technical challenges and issues related to BSC component

Although intensive efforts have been made on BSC design and modeling in the past decades, many challenges still exist. One major challenge is the uncertainty. For example, choosing the locations and capacities of preprocessing and processing sites is a widely studied topic. However, location selections and configuration design (e.g., centralized or decentralized) still needs case-by-case analysis that depends on the availability of different biomass feedstocks at different geospatial and temporal scales. The uncertainty, or say the inconsistency, of biomass quality and quantity also has large impacts on biorefinery operations. How to factor those uncertainties into BSC design at an early stage needs to be carefully addressed (Kudakasseril Kurian et al., 2013; Mafakheri and Nasiri, 2014).

Another challenge associated with BSC design is the coordination among different components of BSC. Obtaining accurate information for each BSC component is important to develop effective strategies for the overall BSC. For example, logistic management is critical to link different parts of BSC and is tightly related to strategic decisions and tactical decisions. Inefficient design of transportation network due to limited information of transportation routes and costs may greatly affect the overall performance of BSC. Inventory planning is another critical part in BSC to link biomass production and biomass conversion. Different strategies need to be developed based on the type and characterization of biomass. Otherwise, a significant amount of biomass could be lost during the storage stage, leading to economic loss. Another example is the evolving technologies of biomass conversion. Many emerging technologies have not been commercialized yet, how to design effective BSC for those emerging technologies and ensure reliable performance in the future is an open question (Rentizelas et al., 2009b; Sims and Venturi, 2004).

5.2 Challenges and issues related to BSC modeling and decision-making

How to effectively quantify and model uncertainty always present as a challenge for BSC. In general, two types of uncertainties have been considered in previous studies, one is the parameter uncertainty, the other is the methodological uncertainty. Parameters uncertainties are those related to fluctuation and variations of specific parameters in BSC, such as biomass supply (Nagel, 2000), climate (An et al., 2011b), feedstock quality (Dautzenberg and Hanf, 2008), feedstock cost (Bai et al., 2012), transportation (Ekşioğlu et al., 2009), biofuel demand and price (Markandya and Pemberton, 2010), policy incentive (Parker et al., 2010), and regulatory changes (Palak et al., 2014). The challenges of addressing parameter uncertainties are (1) limited information on data ranges and probability density function and (2) long computational time when solving problems with uncertainty. Many efforts have been made on biomass data collection, especially on collecting data with uncertainties. Examples are the U.S. Department of Energy (DOE) Billion Ton Study that estimated future potential of supplying at least one billion dry tons biomass resources in the United States (Langholtz et al., 2016). Other examples include the U.S. National Biomass Estimator Library (NBEL) (Wang, 2014), USDA Bioenergy Statistics (2018), IEA Energy Access Database (IEA, 2017), USDA Wood2Energy Database (USDA, 2014), and USDA Forest Service Timber Product Output (TPO) database (US Department of Agriculture Forest Service, 2012). All of those are good resources to bound uncertainty, and it will be subject to BSC designers to choose appropriate data sources based on their projects. With respect to modeling parameter uncertainty, previous studies tried to use stochastic programming and fuzzy programming to develop optimization models for BSC design uncertainty as discussed in previous sections. Most of those models are computationally intensive; the challenge is to solve those models in a reasonable time with robust and reliable results.

Methodological uncertainties are those brought in by different methodological options. For example, when quantifying environmental impacts of BSC, different allocation methods [e.g., mass allocation versus energy allocation (Lardon et al., 2009; Wiloso et al., 2012)], environmental footprints [e.g., GHG and water footprints (Bernardi et al., 2013; Yang et al., 2011)], and LCIA methods [e.g., TRACI, Eco-indicator 99 (Cherubini and Strømman, 2011; Kim and Dale, 2005; Morales et al., 2015; Neupane et al., 2011)] may lead to different conclusions. It may be hard to completely address methodological, but it is always helpful to recognize such uncertainty sources and include sensitivity analysis to test the robustness of the results. Another challenge is meaningful quantification of different aspects related to sustainability. For example, job creation is widely used in BSC design to represent social impacts of BSC. However, there are many other social implications such as food security, environmental justice, and social welfare benefits (Dauvergne and Neville, 2010; Bringezu et al., 2009; de Gorter and Just, 2010). Social LCA has been developed in recent decades to evaluate the social and socioeconomic impacts of products and their life cycles (UNEP and SETAC Life Cycle Initiative, 2009). Similar as environmental LCA, social LCA needs to collect intensive inventory data (e.g., number of working hours), which is challenging for biofuels that have not been industrially commercialized in many regions. Even for environmental LCA or techno-economic analysis that have relatively more developed methodologies and tools than social LCA, intensive data needs, especially the need of the process-based inventory data, are always challenging. Some researchers have used process-based simulation models [e.g., Aspen Plus (You et al., 2012; Zhang and Wright, 2014; Sukumara et al., 2014; Zhang et al., 2014;

Sammons et al., 2007, 2008; Pérez et al., 2016)] to generate process-based inventory data. However, most of previous studies developed process-based models in separate environment with BSC design models, making the system optimization challenging. In the future, more efforts should be made on developing effective and robust methods to generate and collect data to better quantify the environmental, economic, and social impacts of BSC. Some new modeling techniques such as machine learning and big data analytics could be a possible solution. Given the potential trade-offs among different aspects of sustainability, how to better understand and integrate those aspects into BSC design and modeling is another area that needs more efforts. As discussed previously, many studies used MCDA to integrate multiple objective functions into a single objective using weighting factors that are subject to stakeholder preferences and socioeconomic and regional contexts. Given the large impacts of weighting factors on the results, it is critical for researchers and BSC designers to provide transparent documentation and interpretation.

6 Conclusions and future directions

In this chapter, a comprehensive review was conducted for BSC design to present its status quo, issues, and challenges. Based on the literature included in this review, infrastructure location, capacity selection, and network design are the top three strategic decisions that have been mostly investigated by previous studies. Regarding tactical and operational decisions, logistic management related decisions are most investigated by previous studies. Two types of approaches, optimization and simulation, are commonly used to support decision-making in BSC design. Between the two modeling approaches, optimization is used in more studies based on the papers included in this review.

For optimization studies, economic objective functions such as maximizing NPV and profit or minimizing the cost at different levels are commonly investigated in most of BSC design cases. Environmental and social objective functions are also considered in many studies to address more sustainability issues in the BSC design. Several environmental indicators are commonly used in reviewed studies such as GHG or different LCIA indicators. These indicators can be modeled as either objective functions or constraints. Job creations are the most common indicators used in BSC optimization models for social sustainability. As the increasing awareness of sustainability, it is expected that more efforts will be made in better understanding and incorporating sustainability related aspects into BSC design.

Different modeling approaches have been used to solve BSC optimization models. Among those approaches, MILP is found to be the most common type of modeling technique. Binary variables are useful for modeling decisions related to locations or technologies. NLP can be employed when nonlinearity exists in models (e.g., production constraints). Both MILP and NLP are used for deterministic BSC. However, there are many sources of uncertainties in BSC, such as biomass availability, feedstock price, fuel demand, and selling price. To address uncertainty, SP and FMP are used in many previous studies. Uncertainties related to tactical and operational decisions were typically modeled in the second stage of multistage optimization. Besides uncertainties in BSC, BSC design with high geographic resolution can be challenging given the intensive need of geospatial information. GIS has been used by previous studies to process and provide the spatial data need by BSC design.

As another modeling approach for BSC design, simulation can offer a better understanding of the dynamic effects of design strategies and parameter settings. In most of previous studies reviewed in this chapter, BSC design decisions was made by developing "what-if" scenarios in simulation models. Among different simulation techniques, ABM is a powerful technique that has been employed by researchers to support BSC decision-making with a consideration of individual stakeholder behaviors and to understand emergent phenomena for risk management.

The uncertainty of different components in BSC design is challenging from both technical and modeling perspective. How to coordinate different component in BSC with a consideration of uncertainty is challenging and needs more efforts on the data collection and decision-making tool development. Effective algorithms are also needed for complex BSC models that take uncertainty into consideration. Some uncertainties are brought in by choices in methodology (e.g., LCA allocation methods). Those uncertainties are hard to be fully addressed, but transparent documentation and sensitivity analysis could be helpful.

Based on the review and challenges identified, future directions that need more efforts are summarized:

(1) BSC design needs more integrated modeling tools to enhance decision-making toward sustainable production and delivery of biofuels, especially on understanding and quantifying environmental and social implications of different BSC design strategies.

(2) Uncertainty challenges need to be addressed from technical, data, and methodological perspectives. More advanced algorithms and computing techniques are needed, as well as more data and standardization on selecting and documenting different methodological options.

(3) The conflicts and relationships between stakeholders at varied scales and levels in BSC need a better understanding to support effective BSC design at an early stage.

(4) In addition to optimization, which has been widely used in BSC design, other modeling tools such as ABM and GIS demonstrate a strong capability in supporting BSC decision-making. More case studies will be needed to explore the broader use and effectiveness of different modeling techniques for BSC design.

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CHAPTER 11

Fuzzy multicriteria decision making on ranking the biofuels production pathways

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1 Introduction

The development of renewable energy sources (biofuels including biogas, bioethanol, and biodiesel) has been recognized as a promising way for emissions reduction and substituting the fossil fuels (Ren et al., 2014). There are various kinds of biomass resources such as soybean, rapeseed, *jatropha* seeds, palm fruit, and sunflower seeds which can be used for biofuel production, and different pathways for biofuel production perform different in economic, environmental, and social aspects (Liang et al., 2016).

Previous studies have shown that biofuels have significant advantages in economy, environment, and society when compared with fossil fuels. Hill et al. (2006) analyzed the economic competiveness and net social benefits of a biofuel and pointed out that with proper policy support and large subsidies biofuel production could be profitable. Phalan (2009) agreed with the opinion that biofuels could provide economic benefits, decrease emissions, and

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contribute to the society to some extent under suitable situation. Job creation through the central policy of biofuel also could make a difference toward the development of the economy and society (Demirbas, 2009).

Accordingly, different pathways for biofuel production have different sustainability performances, and developing a method for helping the stakeholders/decision-makers to select the most sustainable pathway for biofuel production is of vital importance.

The selection of the most sustainable pathway for biofuel production usually involves a set of criteria in multiple aspects including economic, environmental, technological, and social-political aspects, thus, it is usually a multicriteria decision making (MCDM) problem. Multicriteria decision making methods aim at ranking a set of alternatives with the considerations of multiple criteria, and the combination of sustainability assessment and MCDM method can achieve sustainability ranking of alternatives (An et al., 2016). Scott et al. (2012) revealed that the most popular application of MCDM methods used in the area of bioenergy is technology selection. Perimenis et al. (2011) developed a multicriteria analysis method as the decision support tool for the assessment of biofuels with the considerations of economic, environmental, and social aspects along the biofuel production chain. Cobuloglu and Büyüktahtakın (2015) developed a stochastic analytic hierarchy process (AHP) method for the selection of sustainable biomass crop for biofuel production by considering economic, environmental, and social dimensions.

Uncertainty, which results from language description and probability and statics, generally exists in practical problems. Fuzzy set theory was thus introduced to MCDM field named Fuzzy Multiple Criteria Decision Making (FMCDM), aiming at handing the problems with linguistic variables properly. Previous studies have shown the feasibility of the application of FMCDM in various fields, including engineering, technology, science, management, and economy (Mardani et al., 2015; Behzadian et al., 2010). MCDM and fuzzy MCDM have been divided into many domains and methods by a group of researchers (Mardani et al., 2015; Baležentis et al., 2010; Liou, 2013). Generally speaking, fuzzy MCDM methods can be classified as fuzzy multiobjective decision making (DMODM) accesses and fuzzy multiattribute decision making (FMADM) approaches (Mardani et al., 2015; Kadane, 2011; Liou and Tzeng, 2012). Actually, there exist various classifications of FMCDM tools based on different principles. For instance, Peneva and Popchev (2008) pointed out that the problems with real numbers as weight, the methods, like Weighted Mean

(Chiclana et al., 1998), Weighted MaxMin and Weighted MinMax (Fodor and Roubens, 1995), and Weighted Geometric (Chiclana et al., 2000) can be used for the aggregation of fuzzy relations. On the other hand, Hwang et al. (1992) proposed two other categories that are made up of the approaches aiming to find a ranking, such as optimal degree, linguistic ranking methods, and comparison functions (Mardani et al., 2015). Further information can be found in the study of Mardani et al. (2015), who make a comprehensive summary of the development and enrichment of theory and applications for FMCDM from 1994 to 2014.

Due to the flexibility of FMCDM dealing with practical problems, it has been widely applied in plenty of fields, especially engineering, including civil engineering (Bagočius et al., 2014), computer science (Kaya and Kahraman, 2014; Yazdani-Chamzini, 2014), industrial engineering (Avikal et al., 2014; Keskin, 2014), electrical engineering (Kurt, 2014), and mechanical engineering (Azadnia et al., 2014; Mardani et al., 2015). Sadeghzadeh and Salehi (2011) used TOPSIS to determine the optimal alternative from the list of strategic technologies for fuel cell. Kucukvar et al. (2014) applied F-ENROPY and TOPSIS to evaluate the performance of life cycle sustainability for pavements. It is obvious to see the huge potential and application value of FMCDM in solving engineering problems.

Based on the literature reviews, however, there is a research gap in the current studies about multicriteria decision making on selecting the best or the most sustainable biofuel production pathway: it is usually difficult or even impossible for the users to obtain the data of the alternative biofuel production pathways with respect to the evaluation criteria. In order to overcome this research gap, this study aims at developing a fuzzy multicriteria decision making approach for sustainability ranking of alternative pathways for biofuel production.

Besides the introduction section, the remainder part of this chapter has been organized as follows: the fuzzy multicriteria decision making method for sustainability ranking of alternative pathways for biofuel production is presented in Section 2; an illustrative case is studied in Section 3; sensitivity analysis and validation are carried out in Section 4; and finally, this chapter is concluded in Section 5.

2 Fuzzy multicriteria decision making method

Some frequently used MADM methods under fuzziness include fuzzy analytic hierarchy process (AHP), fuzzy TOPSIS, fuzzy outranking methods (fuzzy ELECTRE and fuzzy PROMETHEE), and fuzzy weighting methods. MODM techniques combined with fuzzy set theory are also a major part of FMCDM, such as fuzzy multiobjective linear programming, quasiconcave and nonconcave fuzzy multiobjective programming, interactive fuzzy stochastic linear programming, fuzzy multiobjective integer goal programming, gray fuzzy multiobjective optimization, and fuzzy multiobjective geometric programming (Kahraman, 2008). The method applied in this research is a kind of fuzzy multiobjective programming approach. Framework for the FMCDM method is shown in Fig. 11.1.

2.1 Fuzzy concept

Fuzzy set theory had been introduced by Zadeh (1965), and many improved fuzzy methods have been developed to be used in many fields such as optimization of multiobjective problem and multicriteria decision making (Wang and Chen, 2011; Chen et al., 2011; Bajpai et al., 2010).

Definition 1 Fuzzy sets (Khrais et al., 2011)
Assume that X is a collection of objects presented by x, a fuzz set α in X is a set of ordered pairs defined as shown in Eq. (11.1), and the bigger the value of the membership function, it will be more certain that x belongs to α.

$$\alpha = \{(x, \mu_\alpha(x)) | x \in X\} \tag{11.1}$$

where $\mu_\alpha(x)$ is the membership function of x in α.

Definition 2 Triangular fuzzy numbers (Tsai and Hsiao, 2004)
The triangular Fuzzy number is usually used in fuzzy study, and $\widetilde{a}$ can be defined by a triplet (a^L, a^M, a^U). Its mathematical and graphic concepts are shown in Eq. (11.2) and Fig. 11.2, respectively.

$$\mu_{\widetilde{a}}(x) = \begin{cases} 0 & x \leq a^L \\ \dfrac{x - a^L}{a^M - a^L} & a^L < x \leq a^M \\ \dfrac{x - a^U}{a^M - a^U} & a^M < x \leq a^U \\ 0 & x > a^U \end{cases} \tag{11.2}$$

Definition 3 Arithmetic operations (Chang, 1996; Yuen and Lau, 2011; Chen, 2000)

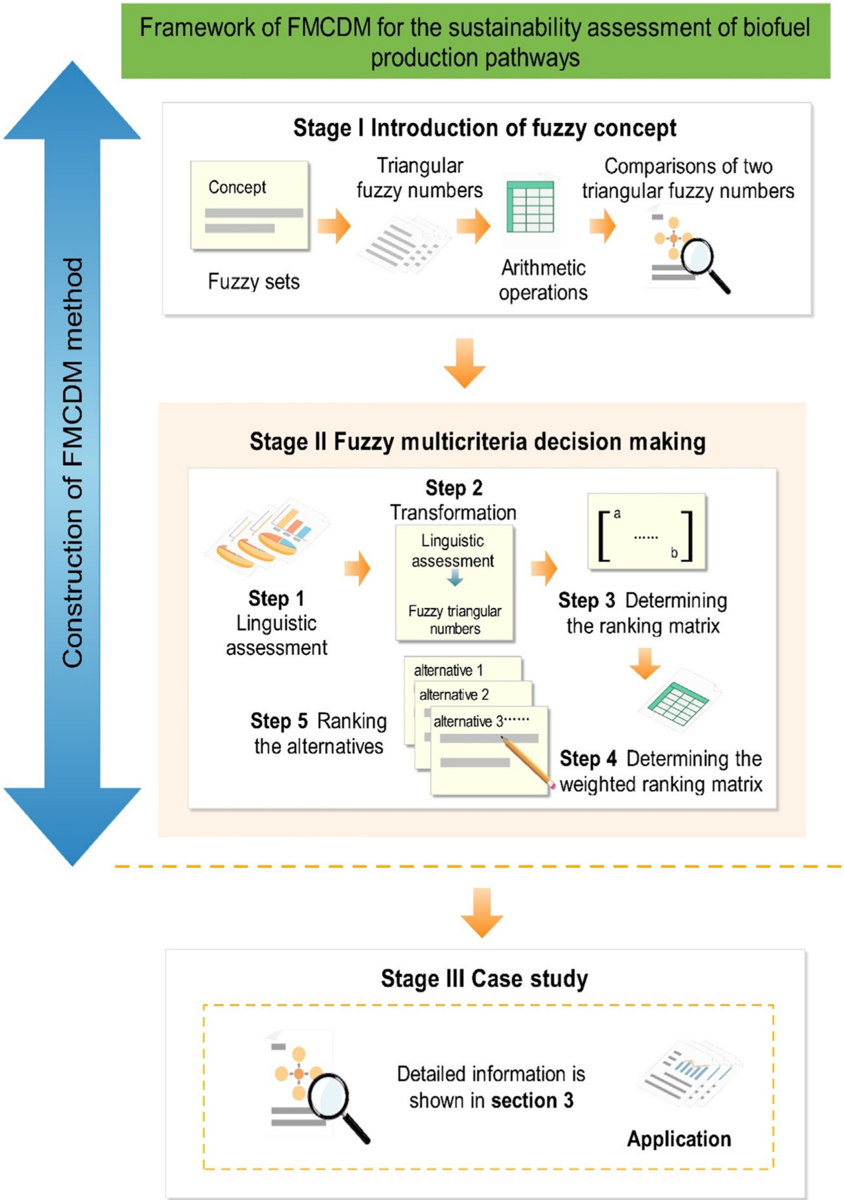

Fig. 11.1 Framework of FMCDM method for the sustainability assessment of biofuel production pathways.

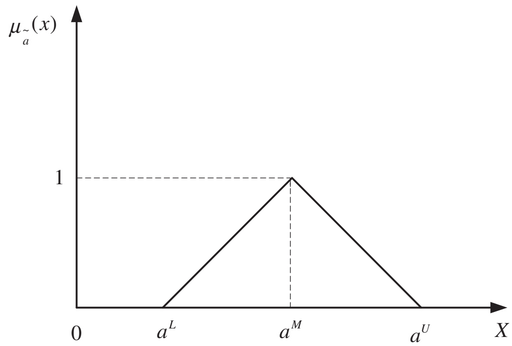

Fig. 11.2 Triangular fuzzy number (a^L, a^M, a^U).

The arithmetic operations between the triangular fuzzy numbers are presented in Table 11.1.

Definition 4 Comparisons of two triangular fuzzy numbers (Xu, 2002; Wei, 2010)
The probability of $\widetilde{x}_1 = \left(x_1^L, x_1^M, x_1^U\right)$ being greater than $\widetilde{x}_2 = \left(x_2^L, x_2^M, x_2^U\right)$ can be calculated by Eq. (11.10).

$$P(\widetilde{x}_1 \geq \widetilde{x}_2) = \lambda \max\left\{1 - \max\left[\frac{x_2^M - x_1^L}{x_1^M - x_1^L + x_2^M - x_2^L}, 0\right], 0\right\} + (1-\lambda)\max\left\{1 - \max\left[\frac{x_2^U - x_1^M}{x_1^U - x_1^M + x_2^U - x_2^M}, 0\right], 0\right\} \tag{11.10}$$

where $P(\widetilde{x}_1 \geq \widetilde{x}_2)$ represents the probability of $\widetilde{x}_1 = \left(x_1^L, x_1^M, x_1^U\right)$ be greater than $\widetilde{x}_2 = \left(x_2^L, x_2^M, x_2^U\right)$, and λ represents the attitudes of the decision-makers on the risk, and it usually takes the value of 0.50 which means that decision-makers are neutral to the risk. While a value which is greater than 0.50 should be assigned to λ if the decision-makers tend to pursue the risks, and a value which is smaller than 0.50 should be assigned to λ when the decision-makers intend to reject the risks.

2.2 Fuzzy multicriteria decision making

A fuzzy Multicriteria Decision Making (FMCDM) method which allows the stakeholders and decision-makers using linguistic terms/variables to participate in the decision-making process has been developed for sustainability assessment of biofuel production pathways. The procedures of FMCDM are illustrated as follows based on the work of Li (2003). This method has been used for sustainability ranking of biomass-based technologies for hydrogen

Table 11.1 The arithmetic operations between the triangular fuzzy numbers

$\widetilde{A}=(a^1,a^2,a^3)$ and $\widetilde{B}=(b^1,b^2,b^3)$ are two triangular fuzzy numbers, and $\lambda>0$, $\lambda\in R$		
Addition	$\widetilde{A}+\widetilde{B}=(a^1,a^2,a^3)+(b^1,b^2,b^3)=(a^1+b^1,a^2+b^2,a^3+b^3)$	(11.3)
Subtraction	$\widetilde{A}-\widetilde{B}=(a^1,a^2,a^3)-(b^1,b^2,b^3)=(a^1-b^3,a^2-b^2,a^3-b^1)$	(11.4)
Multiplication	$\widetilde{A}\otimes\widetilde{B}=(a^1,a^2,a^3)\otimes(b^1,b^2,b^3)=(a^1b^1,a^2b^2,a^3b^3)$	(11.5)
Scalar	$\lambda\widetilde{A}=\lambda(a^1,a^2,a^3)=(\lambda a^1,\lambda a^2,\lambda a^3)$	(11.6)
Division	$\widetilde{A}\div\widetilde{B}=(a^1,a^2,a^3)\div(b^1,b^2,b^3)=\left({}^{a^1}/_{b^3},{}^{a^2}/_{b^2},{}^{a^3}/_{b^1}\right)$	(11.7)
Reciprocal	$\frac{1}{\widetilde{A}}=\frac{1}{(a^1,a^2,a^3)}=\left(\frac{1}{a^3},\frac{1}{a^2},\frac{1}{a^1}\right)$	(11.8)
Euclidean distance	$d\left(\widetilde{A},\widetilde{B}\right)=\left[\left(a^1-b^1\right)^2+\left(a^2-b^2\right)^2+\left(a^3-b^3\right)^2\right]^{1/2}$ where $0<a^1\leq a^2\leq a^3$ and $0<b^1\leq b^2\leq b^3$	(11.9)

Table 11.2 Linguistic variable for assigning weights to evaluation criteria and rating alternative

Number	Linguistics for weights	Linguistics for performance	Fuzzy scale
1	Lowest (LT)	Worst (WT)	(0,0,0.1)
2	Lower (LR)	Worse (WE)	(0,0.1,0.3)
3	Low (L)	Bad (B)	(0.1,0.3,0.5)
4	Medium (M)	Medium (M)	(0.3,0.5,0.7)
5	High (H)	Good (G)	(0.5,0.7,0.9)
6	Higher (HR)	Better (BR)	(0.7,0.9,1.0)
7	Highest (HT)	Best (BT)	(0.9,1.0,1.0)

production in the work of Ren et al. (2013). This FMCDM method was presented as follows (Li, 2003; Ren et al., 2013):

Step 1: Linguistic assessment.

Assume that a set of the stakeholders and decision-makers have been invited to participate in the decision-making process, M alternatives have been assessed, and N criteria have been used to assess the alternatives. The decision-makers are asked to assign the importance of the criteria and rate the alternatives using the linguistic variables (see Table 11.2).

Step 2: Transformation.

Transfer the linguistic assessment into fuzzy triangular numbers according to Table 11.2. Let ω_j be the weight of the (j)th criterion by the stakeholders and decision-makers and $\widetilde{x}_{ij}$ be the assessment on the performance of the ith alternative with respect to the jth criterion. Assume the fuzzy decision-making matrix determined by the decision-makers is presented in Eq. (11.11).

$$\begin{array}{ccccc} & C_1 & C_2 & \cdots & C_n \\ & \widetilde{\omega}_1 & \widetilde{\omega}_2 & \cdots & \widetilde{\omega}_n \\ A_1 & \widetilde{x}_{11} & \widetilde{x}_{12} & \cdots & \widetilde{x}_{1n} \\ A_2 & \widetilde{x}_{21} & \widetilde{x}_{22} & \cdots & \widetilde{x}_{2n} \\ \vdots & \vdots & \vdots & \vdots & \vdots \\ A_m & \widetilde{x}_{m1} & \widetilde{x}_{m2} & \cdots & \widetilde{x}_{mn} \end{array} \tag{11.11}$$

where A_i represents the ith alternative, C_j represents the jth criterion, $\widetilde{\omega}_j = \left(\omega_j^L\ \omega_j^M\ \omega_j^U\right)$ is the weight of the jth criterion, and $\widetilde{x}_{ij}$ represents the performance of the ith alternative with respect to the ith.

Step 3: Determining the ranking matrix.

Rank the alternatives corresponding to each criterion according to Eq. (11.10), for the (j)th criterion, the ranking matrix can be obtained with the following method.

$$\varphi_{it}^{j} = \begin{cases} 1, & \text{if the } i\text{th alternative has been ranked at the } t\text{th place} \\ 0, & \text{if the } i\text{th alternative has not been ranked at the } t\text{th place} \end{cases} \tag{11.12}$$

$$\varphi^{j} = \left\{ \varphi_{it}^{j} \right\}_{m \times m} \tag{11.13}$$

where φ^j is the ranking matrix corresponding to the (j)th criterion and φ_{it}^j is the element in the ranking matrix corresponding to the (j)th criterion.

Step 4: Weighted ranking matrix.

The weighted ranking matrix R can be obtained by Eq. (11.14) and the elements in the weighted ranking matrix can be calculated by Eqs. (11.15), (11.16).

$$R = \begin{array}{cccccc} & 1 & 2 & \cdots & m \\ A_1 & r_{11} & r_{12} & \cdots & r_{1m} \\ A_2 & r_{21} & r_{22} & \cdots & r_{2m} \\ \vdots & \vdots & \vdots & \vdots & \vdots \\ A_m & r_{m1} & r_{m2} & \cdots & r_{mm} \end{array} \tag{11.14}$$

$$r_{it} = \sum_{j=1}^{n} \varphi_{it}^{j} \omega_j \tag{11.15}$$

$$\omega_j = \left(\frac{\omega_j^L + 2\omega_j^M + \omega_j^U}{4} \right) / \sum_{j=1}^{n} \frac{\omega_j^L + 2\omega_j^M + \omega_j^U}{4} \tag{11.16}$$

Step 5: Ranking the sequence of the alternatives.

Use the following linear 0–1 programming to rank the alternatives, and the solutions of this programming are the elements in the final ranking matrix Z, as shown in Eq. (11.21). If $z_{it} = 1$, it means that the (i)th alternative has been ranked at the (t)th place.

$$\text{Max } S = \sum_{j=1}^{m} \sum_{t=1}^{m} r_{it} z_{it} \tag{11.17}$$

$$z_{it} = \begin{cases} 1, & \text{if the } i\text{th alternative has been ranked at the } t\text{th place} \\ 0, & \text{if the } i\text{th alternative has not been ranked at the } t\text{th place} \end{cases} \tag{11.18}$$

$$\sum_{t=1}^{m} z_{it} = 1, \quad i = 1, 2, \ldots, m \tag{11.19}$$

$$\sum_{i=1}^{m} z_{it} = 1, \quad t = 1, 2, \ldots, m \tag{11.20}$$

$$Z = \{ z_{it} \}_{m \times m} \tag{11.21}$$

3 Case study

In order to illustrate the developed fuzzy multicriteria decision making method for sustainability ranking of biofuel production pathway, three scenarios for bioethanol were investigated by the developed fuzzy multicriteria decision making method, and they are corn-, wheat-, and cassava-based technologies for bioethanol production. A total of nine criteria in economic, environmental, technological, and social-political aspects were employed to assess these three biofuel production pathways, and they are life cycle cost (LCC) in economic aspect, climate change (CC), terrestrial acidification (TA), human toxicity (H.Tox), particulate matter formation (PMF) in environmental aspect, technology maturity (TM) in technological aspect, and social benefits (SB), contribution to economic development (CED), and food security (FS) in social-political aspect.

In order to determine the fuzzy multicriteria decision-making matrix, a focus group was held in China. Three full professors whose research focused on bioenergy, three senior engineers who are skilled bioethanol production, three PhD students who are working in the field of renewable energy, and three administrators were invited to participate in the decision-making process. One of the authors is the coordinator of this focus group meeting, and he is responsible to achieve a consensus among these experts when there are different opinions among them. After this focus group meeting, the experts employed the linguistic terms presented in Table 11.2 to assign the weights of the nine criteria and rate the three alternative pathways for bioethanol production, and the results are presented in Table 11.3.

Table 11.3 The weights of the nine criteria and the performances of the three pathways for bioethanol production with respect to each criterion using linguistic terms

	Wheat-based	Corn-based	Cassava-based	Relative importance
LCC	B	M	G	HT
CC	B	BT	G	HR
TA	WE	BR	M	H
H.Tox	B	BT	G	M
PMF	WT	BT	M	M
TM	G	G	M	HR
SB	M	M	G	L
CED	M	G	BR	M
FS	WT	B	BT	H

Table 11.4 The weights of the nine criteria and the performances of the three pathways for bioethanol production with respect to each criterion using fuzzy numbers

	Wheat-based	Corn-based	Cassava-based	Relative importance
LCC	(0.1,0.3,0.5)	(0.3,0.5,0.7)	(0.5,0.7,0.9)	(0.9,1.0,1.0)
CC	(0.1,0.3,0.5)	(0.9,1.0,1.0)	(0.5,0.7,0.9)	(0.7,0.9,1.0)
TA	(0,0.1,0.3)	(0.7,0.9,1.0)	(0.3,0.5,0.7)	(0.5,0.7,0.9)
H.Tox	(0.1,0.3,0.5)	(0.9,1.0,1.0)	(0.5,0.7,0.9)	(0.3,0.5,0.7)
PMF	(0,0,0.1)	(0.9,1.0,1.0)	(0.3,0.5,0.7)	(0.3,0.5,0.7)
TM	(0.5,0.7,0.9)	(0.5,0.7,0.9)	(0.3,0.5,0.7)	(0.7,0.9,1.0)
SB	(0.3,0.5,0.7)	(0.3,0.5,0.7)	(0.5,0.7,0.9)	(0.1,0.3,0.5)
CED	(0.3,0.5,0.7)	(0.5,0.7,0.9)	(0.7,0.9,1.0)	(0.3,0.5,0.7)
FS	(0,0,0.1)	(0.1,0.3,0.5)	(0.9,1.0,1.0)	(0.5,0.7,0.9)

The linguistic terms presented in Table 11.3 can be transported into triangular fuzzy numbers and the results are presented in Table 11.4. It is worth pointing out that there are multiple different stakeholders/decision-makers participating in the decision-making process, and different multicriteria decision-making matrices were provided by them, the users can use the average value of the alternatives with respect to each evaluation criterion to determine the sustainability sequence of the alternative pathways for bioethanol production.

After this, the ranking matrix with respect to each criterion can be determined according to Eqs. (11.12, 11.13). Taking the criterion-LCC as an example, the data of wheat-, corn-, and cassava-based technologies with respect to LCC are (0.1,0.3,0.5), (0.3,0.5,0.7), and (0.5,0.7,0.9), respectively. These three triangular fuzzy numbers can be ranked according to Eq. (11.10), and λ takes the value of 0.50 in this study, and the ranking matrix with respect to LCC can be then determined, as presented in

$$\varphi^{LCC} = \begin{array}{c} \\ \text{Wheat}-\text{based} \\ \text{Corn}-\text{based} \\ \text{Cassava}-\text{based} \end{array} \begin{array}{ccc} 1 & 2 & 3 \\ 0 & 0 & 1 \\ 0 & 1 & 0 \\ 1 & 0 & 0 \end{array} \tag{11.22}$$

Similarly, the ranking matrices with respect to the other eight criteria can also be calculated determined, and the results are presented in Table 11.5.

Meanwhile, the fuzzy weights of these nine criteria can be defuzzied by Eq. (11.16), and the results are presented in Table 11.6.

Table 11.5 The ranking matrices with respect to the other eight criteria

CC	1	2	3	TA	1	2	3
Wheat-based	0	0	1	Wheat-based	0	0	1
Corn-based	1	0	0	Corn-based	1	0	0
Cassava-based	0	1	0	Cassava-based	0	1	0
H.Tox	1	2	3	PMF	1	2	3
Wheat-based	0	0	1	Wheat-based	0	0	1
Corn-based	1	0	0	Corn-based	1	0	0
Cassava-based	0	1	0	Cassava-based	0	1	0
TM	1	2	3	SB	1	2	3
Wheat-based	1	0	0	Wheat-based	0	1	0
Corn-based	1	0	0	Corn-based	0	1	0
Cassava-based	0	1	0	Cassava-based	1	0	0
CED	1	2	3	FS	1	2	3
Wheat-based	0	0	1	Wheat-based	0	0	1
Corn-based	0	1	0	Corn-based	0	1	0
Cassava-based	1	0	0	Cassava-based	1	0	0

Then, the weighted ranking matrix can be obtained by Eqs. (11.14–11.16), as presented in Eq. (11.23).

$$R = \begin{array}{cccc} & 1 & 2 & 3 \\ \text{Wheat}-\text{based} & 0.1477 & 0.0506 & 0.8017 \\ \text{Corn}-\text{based} & 0.5823 & 0.4177 & 0 \\ \text{Cassava}-\text{based} & 0.4177 & 0.5823 & 0 \end{array} \tag{11.23}$$

The final ranking matrix can be determined by the fuzzy linear 0–1 programming according to Eqs. (11.17)–(11.21), as shown in Eq. (11.24).

$$\begin{aligned} &\text{Max}\, S = 0.1477z_{11} + 0.0506z_{12} + 0.8017z_{13} + 0.5823z_{21} \\ &\qquad + 0.4177z_{22} + 0.4177z_{31} + 0.5823z_{32} \\ &z_{it} \in \{0, 1\}, i = 1,2,3; j = 1,2,3 \\ &z_{11} + z_{12} + z_{13} = 1 \\ &z_{21} + z_{22} + z_{23} = 1 \\ &z_{31} + z_{32} + z_{33} = 1 \\ &z_{11} + z_{21} + z_{31} = 1 \\ &z_{12} + z_{22} + z_{32} = 1 \\ &z_{13} + z_{23} + z_{33} = 1 \\ &Z = \begin{vmatrix} z_{11} & z_{12} & z_{13} \\ z_{21} & z_{22} & z_{23} \\ z_{31} & z_{32} & z_{33} \end{vmatrix} \end{aligned} \tag{11.24}$$

Table 11.6 The defuzzied weights of these nine criteria for sustainability assessment of biofuel production pathways

Criteria	LCC	CC	TA	H.Tox	PMF	TM	SB	CED	FA
Fuzzy weights	(0.9,1.0,1.0)	(0.7,0.9,1.0)	(0.5,0.7,0.9)	(0.3,0.5,0.7)	(0.3,0.5,0.7)	(0.7,0.9,1.0)	(0.1,0.3,0.5)	(0.3,0.5,0.7)	(0.5,0.7,0.9)
Crisp weights	0.1646	0.1477	0.1181	0.0844	0.0844	0.1477	0.0506	0.0844	0.1181

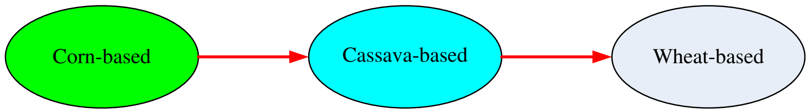

Fig. 11.3 The sustainability sequence of the three alternative pathways for bioethanol production.

After solving programming (11.24), the matrix Z can be determined, as presented in Eq. (11.25).

$$Z = \begin{array}{lccc} & 1 & 2 & 3 \\ \text{Wheat}-\text{based} & 0 & 0 & 1 \\ \text{Corn}-\text{based} & 1 & 0 & 0 \\ \text{Cassava}-\text{based} & 0 & 1 & 0 \end{array} \tag{11.25}$$

According to the meaning of the elements in the final ranking matrix, the sustainable sequence of the three alternative pathways for bioethanol production can be determined, as shown in Fig. 11.3.

4 Discussion

The sustainability of the three alternative pathways for bioethanol production from the most sustainable to the least is corn-based, cassava-based, and wheat-based. It is worth pointing out that the ranking of these three alternative pathways for bioethanol production was determined based on the opinions and preferences of the selected experts, and the results may change when the stakeholders/decision-makers have been changed. In order to investigate the influences of the weights of the criteria on the sustainability sequence of the three alternative pathways for bioethanol production, the following eleven cases were studied for sensitivity analysis by changing the weights of the nine criteria:

Case 0: Equal weights—an equal weight (0.1111) was assigned to the nine criteria;

Case 1–9: a dominant weight (0.3600) was assigned to one of the nine criteria, and an equal weight (0.0800) was assigned to all the other eight criteria.

The results of sensitivity analysis are presented in Fig. 11.4. It is apparent that the sustainability sequence of these three alternative pathways for bioethanol production may change with the change of the weights of the criteria for sustainability assessment. In other words, the sustainability sequence of the three pathways for bioethanol production may change when

Fig. 11.4 The results of sensitivity analysis.

changing the preferences of the stakeholders and decision-makers. Accordingly, the sustainability sequence may change when the stakeholders and decision-makers have been changed and there are two main reasons: (i) the weights of the criteria may change when the stakeholders and decision-makers have been changed and (ii) the relative performances of the three alternative pathways for bioethanol production may change with respect to the criteria that may change when the stakeholders and decision-makers have been changed.

In order to validate the developed fuzzy multicriteria decision making method for sustainability assessment and ranking of alternative pathways for biofuel production, the fuzzy sum weighted method (SWM) was also employed to determine the sustainability sequence of these three pathways for bioethanol production. These two methods were specified as follows.

As for the fuzzy SWM, the integrated priority of each alternative pathway for biofuel production can be determined after determining the fuzzy multicriteria decision-making matrix (see Eq. 11.11) by Eq. (11.26).

$$\widetilde{P}_i = \sum_{j=1}^{n} \widetilde{x}_{ij}\widetilde{\omega}_j \quad i = 1, 2, \ldots, m \tag{11.26}$$

where $\widetilde{P}_i$ represents the integrated priority of the ith alternative for biofuel production.

According to the data presented in Table 11.4, the integrated priority of each alternative pathway for biofuel production can be determined, as presented in Eqs. (11.27)–(11.29)

$$\begin{aligned} P_{\text{Wheat-based}} &= (0.1, 0.3, 0.5) \times (0.9, 1.0, 1.0) + (0.1, 0.3, 0.5) \\ &\quad \times (0.7, 0.9, 1.0) + \cdots + (0, 0, 0.1) \times (0.5, 0.7, 0.9) \\ &= (0.6600, 1.8200, 3.5200) \end{aligned} \tag{11.27}$$

$$\begin{aligned} P_{\text{Corn-based}} &= (0.3, 0.5, 0.7) \times (0.9, 1.0, 1.0) + (0.9, 1.0, 1.0) \\ &\quad \times (0.7, 0.9, 1.0) + \cdots + (0.1, 0.3, 0.5) \times (0.5, 0.7, 0.9) \\ &= (2.3700, 4.3700, 6.3300) \end{aligned} \tag{11.28}$$

$$\begin{aligned} P_{\text{Cassava-based}} &= (0.5, 0.7, 0.9) \times (0.9, 1.0, 1.0) + (0.5, 0.7, 0.9) \\ &\quad \times (0.7, 0.9, 1.0) + \cdots + (0.9, 1.0, 1.0) \times (0.5, 0.7, 0.9) \\ &= (2.1100, 4.0900, 6.3000) \end{aligned} \tag{11.29}$$

Then, the integrated priorities of the three alternative pathways for biofuel production can be ranked according to Eq. (11.10), and it could be obtained that $P_{\text{Corn-based}} \succ P_{\text{Cassava-based}} \succ P_{\text{Wheat-based}}$. Therefore corn-based pathway was recognized as the most sustainable, followed by cassava- and wheat-based pathways, and the results determined by the fuzzy SWM are consistent to that determined by the proposed fuzzy multicriteria decision making in this study.

5 Conclusion

A fuzzy multicriteria decision making method was developed for sustainability ranking of biofuel production pathways, and the stakeholders/decision-makers are allowed to use linguistic variables to weigh the relative importance of the criteria for sustainability assessment and rate the alternative biofuel production pathways, and the opinions and preferences of the stakeholders/decision-makers can be effectively expressed by using fuzzy numbers. However, there are also some weak points in this study:

(1) Some useful information and data cannot be effectively used, because the relative performances of the alternative biofuel production pathways with respect to the evaluation were merely determined according to the judgments of the stakeholders/decision-makers;
(2) The relative importance of the evaluation criteria was assigned by the stakeholders/decision-makers directly rather than in a comparison way, thus, this may lead to some inaccurate judgments.

The future work of the authors is to develop a multicriteria decision making method which can overcome the abovementioned two weak points for sustainability ranking of biofuel production pathways.

Acknowledgments

This method used in this study was based on Ren, J., Fedele, A., Mason, M., Manzardo, A., Scipioni, A., 2013. Fuzzy multi-actor multi-criteria decision making for sustainability assessment of biomass-based technologies for hydrogen production. Int. J. Hydrog. Energy 38, 9111–9120.

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CHAPTER 12

Prioritization of biofuels production pathways under uncertainties

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1 Introduction

Biofuel has been recognized as a promising solution to sustainable and low-carbon transport, because it can be produced from various types of biomass to substitute the petroleum-derived fuel and reduce emissions from transport sector (Hao et al., 2018). Biofuel is one of the most promising renewable energy carriers, and it can be produced from various feedstocks such as biomass, algal, and some other nonstaple food crops (Liang et al., 2016). However, there are also some challenges for promoting the development of biofuels including the relatively higher production, the negative impacts on food security, and competition with some other renewable energy resources (Ren et al., 2015a). Meanwhile, there are also some emissions in the whole life cycle of biofuels, because there is also some consumption for cropping, transportation of biomass resources, biofuel production, transportation of biofuels, and so on (Ren et al., 2014). Therefore the

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development of biofuel industry is still "in debate," and people are usually puzzled with two questions: (i) Is biofuel really sustainable? (ii) Which is the most sustainable pathway for biofuel production?

As for the question: is biofuel really sustainable? There are many studies for answering this question. The most typical is to use life cycle tools to analyze the sustainability of biofuel production pathways from cradle to grave. For instance, Ou et al. (2009) employed life cycle assessment (LCA) to investigate the energy consumption and GHG emissions of six biofuel pathways in China. Yang et al. (2011) used life cycle thinking to analyze the water footprint and nutrients balance of biodiesel from microalgae. Requena et al. (2011) employed LCA to study the environmental impacts of biofuels from sunflower oil, rapeseed oil, and soybean oil. However, all these studies can only answer the first question. As for the second question: which is the most sustainable pathway for biofuel production? It is usually different and even the stakeholders/decision-makers know the performances of different biofuel production pathways and there are two main reasons: (i) there are usually various conflict criteria for sustainability assessment of biofuel production pathways; (ii) there are various data uncertainties, and the data of the alternative biofuel production pathways with respect to the evaluation criteria usually varies. Therefore this study aims at developing an interval multicriteria decision making (MCDM) method for sustainability prioritization of biofuel production pathways under uncertainties.

MCDM is a widely used decision-making tool that allows for scientific and comprehensive analysis based on multiple data and decision-maker's preferences (Triantaphyllou, 2000). MCDM methods commonly used include Analytic Hierarchy Process (AHP) (Saaty, 1980), Technique for Order Preference by Similarity to an Ideal Solution (TOPSIS) (Hwang and Yoon, 1981), Gray Relational Analysis (GRA) (Deng, 1989), The Preference Ranking Organization Method for Enrichment of Evaluations (PROMETHEE) (Brans et al., 1986), and Elimination and Choice Expressing Reality (ELECTRE) (Benayoun et al., 1966). The decision-making method based on uncertain data is an extension of MCDM, which is used to analyze the ranking, selection, and classification of more than one evaluation criterion with uncertain or fuzzy information in the data (Ho et al., 2010). Due to the uncertainty of subjective judgment, the normal fluctuation caused by environmental factors, and the uncertainty caused by knowledge limitations, the application of MCDM under uncertainties plays a significant role in strategy establishment (Ren and Toniolo, 2018). MCDM dealing with uncertain data can be classified into interval MCDM, fuzzy

MCDM, intuitional fuzzy MCDM, and stochastic MCDM. Interval MCDM takes the interval number as the input data in the decision process, so that the value of each criterion reflects its maximum and minimum values (Tsaur, 2011). For example, Giove (2002) and Jahanshahloo et al. (2006) have extended TOPSIS into interval TOPSIS. Fuzzy MCDM is the MCDM that bringing fuzzy number such as triangular fuzzy number and trapezoidal fuzzy number into consideration (Pohekar and Ramachandran, 2004; Kahraman, 2008). The fuzzy number consists more information than interval number to express the uncertainty (Pohekar and Ramachandran, 2004; Kahraman, 2008). For instance, Sevkli (2010) extended ELECTRE into fuzzy ELECTRE, Shemshadi et al. (2011) extended VIKOR into fuzzy VIKOR, and Liu and Wang (2007) developed a MCDM based on intuitionistic fuzzy. The fuzzy MCDM has been further extended into intuitional fuzzy MCDM which adds degree of nonmembership to better express the uncertainty (Liu and Wang, 2007). Different from fuzzy MCDM, stochastic MCDM uses randomness to reveal the uncertainty sets (Ramanathan, 1997). For example, Xiong and Qi (2010) have extended TOPSIS into stochastic multicriteria decision making method. Furthermore, many researchers have also tried to combine different MCDM methods under uncertainty to form hybrid MCDM. For example, Büyüközkan and Çifçi (2012) have proposed a hybrid MCDM approach combining fuzzy Decision making trial and evaluation laboratory (DEMATEL), fuzzy (Analytic Network Process) ANP, and fuzzy TOPSIS together.

Some researchers have done studies to compare biofuel production processes sustainably (e.g., Ou et al., 2009; Yang et al., 2011; Larkum et al., 2012), but these researches cannot show the direct priority through the comparisons. Some people helped to make choices in this selection problem (e.g., Schaidle et al., 2011; Vlysidis et al., 2011; Sharma et al., 2011), but they did not consider uncertainties in the biofuel production processes. Furthermore, some scholars have studied energy selection problem but have not specifically scoped to biofuel such as Sadeghi et al. (2012) provided a fuzzy MCDM approach for renewable electricity production and Lee et al. (2009) have conducted a fuzzy AHP method for energy technology prioritization. Therefore we shall bring uncertainties into consideration and provide a scientific and comprehensive analysis for biofuel production pathways prioritization. Because the interval number represents the uncertainty of the data biofuel production process and leads to few obstacles in data collection, so the data type of interval MCDM is more in line with the operation of sustainable biofuel production selection. The interval AHP which allows the users to use interval numbers rather than the basketball numbers to establish the comparison matrix was employed for determining the weights of the criteria for sustainability assessment of biofuel production pathways, and the interval GRA method was employed to determine the sustainability sequence of alternative biofuel production pathways under uncertainties.

2 Interval multicriteria decision making method

Interval multicriteria decision making method is a series of MCDM adapting interval numbers as criteria inputs and weights, which helps to deal with complex decision-making cases with vagueness of language and uncertainty of criteria values. The interval VIKOR raised by Sayadi et al. (2009) supports decision-makers to rank the alternatives with regards to its performances comparing to the best and the worst alternative. Giove (2002) and Jahanshahloo et al. (2006) have extended TOPSIS into interval TOPSIS, respectively. The interval TOPSIS considers the balance between needs fulfillment and lost compromise. Luo et al. (2015) developed an interval GRA method based on the grey theory invented by Deng (1989). Xu and Da (2003) have extended the AHP method into the interval analytic hierarchy process (IAHP), which assistant to quantify the criteria values and weights according to quality values. In addition, interval PROMETHEE (Le Téno and Mareschal, 1998) and interval Best Worst Method (BWM) (Rezaei, 2016) are methods for dealing qualified inputs as well. Among these, the IAHP is one of the most commonly used weighting methods in MCDM for its advantage of addressing the hesitations and ambiguity existing in human's judgments.

In this section, interval multicriteria decision making method for biofuel production pathways selection is described. Criteria system establishment, criteria weighting, and aggregating are three steps for MCDM method whose methods are designed as criteria system establishment, interval AHP, and interval GRA as shown in Fig. 12.1.

2.1 Interval numbers

The basic information of interval numbers was presented in this section based on the work of He et al. (2017), Zhang et al. (2005), Bohlender and Kulisch (2011), Moore (1966), Dymova et al. (2013), and Xu and Da (2003).

Let $x=[x^-,x^+]=\{x|x^-\le x\le x^+, x^-\le x^+, x^-,x^+\in R\}$. Here $x=[x^-,x^+]$ is called an interval number and is a positive interval number

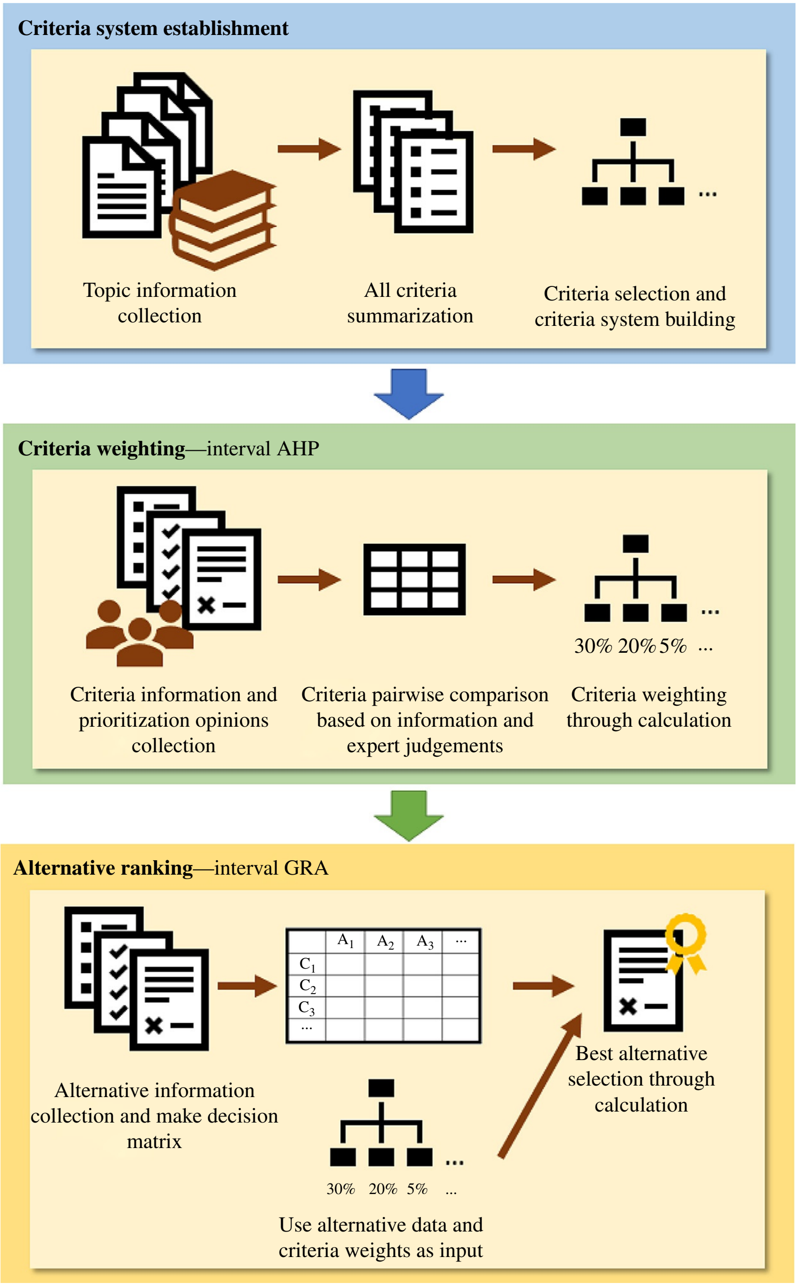

Fig. 12.1 Methodology of biofuel production pathways selection.

if $0 \le x^- \le x^+$ [see the work of Zhang et al. (2005)]. It is apparent that an interval number can take an arbitrary value between its lower bound and upper bound.

Definition 1 Distance between two interval numbers (Yue, 2011)
If $x = [x^-, x^+]$ and $y = [y^-, y^+]$ are two arbitrary interval numbers, the distance from $x = [x^-, x^+]$ to $y = [y^-, y^+]$ can be determined by

$$|x - y| = \sqrt{(x^- - y^-)^2 + (x^+ - y^+)^2} \tag{12.1}$$

Definition 2 Number product between a positive real number and an interval number (Zhang et al., 2005)
The number product of a positive real number λ and an interval number $x = [x^-, x^+]$ is defined as

$$\lambda \cdot x = \lambda[x^-, x^+] = [\lambda x^-, \lambda x^+] \tag{12.2}$$

Note that a crisp number λ could also be transformed into an interval number $\lambda = \lambda, \lambda]$.

Definition 3 Addition (Bohlender and Kulisch, 2011)
If $x = [x^-, x^+]$ and $y = [y^-, y^+]$ are two arbitrary interval numbers, the sum of the two interval numbers can be obtained by

$$x + y = [x^-, x^+] + [y^-, y^+] = [x^- + y^-, x^+ + y^+] \tag{12.3}$$

Definition 4 Subtraction (Moore, 1966; Dymova et al., 2013)
If $x = [x^-, x^+]$ and $y = [y^-, y^+]$ are two arbitrary interval numbers, the subtraction between two interval numbers can be determined by

$$x - y = [x^-, x^+] - [y^-, y^+] = [x^- - y^+, x^+ - y^-] \tag{12.4}$$

Definition 5 Multiplication (Zhang et al., 2005)
If $x = [x^-, x^+]$ and $y = [y^-, y^+]$ are two arbitrary interval numbers, the interval product can be obtained according to the following two cases:
(1) when $y^+ > 0$, the interval product can be determined by

$$x \times y = [x^-, x^+] \times [y^-, y] = [x^- y^-, x^+ y^+] \tag{12.5}$$

(2) when $y^+ < 0$, the interval product can be determined by

$$x \times x = [x^-, x^+] \times [y^-, y^+] = [x^+ y^-, x^- y^+] \tag{12.6}$$

Definition 6 The possibility of an interval being greater than another
For two arbitrary interval numbers $x=[x^-, x^+]$ and $y=[y^-, y^+]$, the possibilities that $x \geq y$ and that $b \geq a$ are defined in the following two equations, respectively (Xu and Da, 2003):

$$p_{xy} = p([x^-, x^+] \geq [y^-, y^+]) = \max\left\{1 - \max\left[\frac{y^+ - x^-}{y^+ - y^- + x^+ - x^-}\right], 0\right\} \tag{12.7}$$

2.2 Interval AHP

The interval analytic hierarchy process (IAHP) developed by Xu and Da (2003) has been widely used in various fields for its advantage of address ambiguity and hesitation existing in human judgments. Ren (2018) summarized the IAHP method developed by Xu and Da (2003) in the following three steps:

Step 1: Establishing the interval pair-wise comparison matrix.

Assuming that there are n criteria $(C_1, C_2, \ldots, C_n)$ which need the stakeholders/decision-makers to determine the relative importance (weights), the stakeholders/decision-makers were asked to use the Saaty's nine-scale system (Saaty, 2008) to establish the pair-wise comparison matrix (see Table 12.1). However, it is different from the traditional AHP method which relies on employing the numbers(from 1 to 9) and their corresponding reciprocals to establish the pair-wise comparison matrix, the users of the interval numbers rather than the single numbers which sometime cannot

Table 12.1 Saaty scales for establishing the pair-wise comparison matrix (Saaty, 2008)

Scales	Definition	Explanation
1	Equal importance	Two elements perform equally
3	Moderate importance	Experience and judgment slightly favor one element over another
5	Essential importance	Experience and judgment strongly favor one element over another
7	Very strong importance	An element is favored very strongly over another; its dominance demonstrated in practice
9	Absolute importance	The evidence favoring one element over another is of the highest possible order of affirmation
2,4,6,8	Intermediate value	Intermediate value

depict the relative importance/preference of one criterion over another accurately. For instance, it is difficult or even impossible to depict the relative weight/priority of a criterion over another when the stakeholders/decision-makers think that the relative importance of a criterion over another is between "moderate importance" (corresponding to number 3) and "essential importance" (corresponding to number 5). Accordingly, the interval number [3 5] should be used to depict this judgment. In a similar way, the interval comparison matrix for determining the relative importance (weights) of the n metrics can be established:

$$M^{\pm} = \begin{array}{c} \\ C_1 \\ C_2 \\ \vdots \\ C_n \end{array} \begin{array}{cccc} C_1 & C_2 & \cdots & C_n \\ 1 & \left[m_{12}^L, m_{12}^U\right] & \cdots & \left[m_{1n}^L, m_{1n}^U\right] \\ \left[m_{21}^L, m_{21}^U\right] & 1 & \cdots & \left[m_{2n}^L, m_{2n}^U\right] \\ \vdots & \vdots & \ddots & \vdots \\ \left[m_{n1}^L, m_{n1}^U\right] & \left[m_{n2}^L, m_{n2}^U\right] & \cdots & 1 \end{array} \tag{12.8}$$

where $M^{\pm}$ represents the interval pair-wise matrix for determining the relative weights of the n criteria, $[m_{ij}^L, m_{ij}^U]$ which is an interval number represents the relative preference of the ith criterion over the jth criterion, and m_{ij}^L and m_{ij}^U are the lower and upper boundary of the interval number $[q_{ij}^L, q_{ij}^U]$.

The relative preference of the jth criterion over ith metric can be determined by Eq. (12.9).

$$\left[m_{ji}^L, m_{ji}^U\right] = \frac{1}{\left[m_{ij}^L, m_{ij}^U\right]} = \left[\frac{1}{m_{ij}^U}, \frac{1}{m_{ij}^L}\right], \quad i, j = 1, 2, \ldots, n \tag{12.9}$$

Step 2: Decomposing the interval pair-wise comparison matrix into two crisp nonnegative matrices.

The interval pair-wise comparison matrix in Eq. (12.8) can be decomposed into two crisp nonnegative matrices, as presented in Eqs. (12.10), (12.11), respectively.

$$M_L = \begin{vmatrix} 1 & m_{12}^L & \cdots & m_{1n}^L \\ 1/m_{21}^U & 1 & \cdots & m_{2n}^L \\ \vdots & \vdots & \ddots & \vdots \\ 1/m_{n1}^U & 1/m_{n2}^U & \cdots & 1 \end{vmatrix} \tag{12.10}$$

$$M_U = \begin{vmatrix} 1 & m_{12}^U & \cdots & m_{1n}^U \\ 1/m_{21}^L & 1 & \cdots & m_{2n}^U \\ \vdots & \vdots & \ddots & \vdots \\ 1/m_{n1}^L & 1/m_{n2}^L & \cdots & 1 \end{vmatrix} \tag{12.11}$$

The geometric mean method (Ren, 2018) can be used to determine the weights according to the matrices presented in Eqs. (12.10), (12.11), and the weight vectors determined by these two matrices are presented in Eqs. (12.12), (12.13), respectively.

$$W_L = \begin{bmatrix} \omega_1^L & \omega_2^L & \cdots & \omega_n^L \end{bmatrix} \tag{12.12}$$

$$W_U = \begin{bmatrix} \omega_1^U & \omega_2^U & \cdots & \omega_n^U \end{bmatrix} \tag{12.13}$$

where W_L and W_U represent the weight vectors determined by the matrices presented in Eqs. (12.10), (12.11), respectively. ω_j^L and ω_j^U are the weights of the jth metric in W_L and W_U, respectively.

Step 3: Determining the interval weights. The interval weights of each metric can be determined by Eqs. (12.14)–(12.16).

$$k = \sqrt{\sum_{j=1}^{n} \frac{1}{\sum_{i=1}^{n} q_{ij}^+}} \tag{12.14}$$

$$m = \sqrt{\sum_{j=1}^{n} \frac{1}{\sum_{i=1}^{n} q_{ij}^-}} \tag{12.15}$$

It is worth pointing out that if k and m satisfy $0 < k \le 1 \le m$, then the users can use Eq. (12.16) to determine the interval weight of the jth metric, or the users should modify the interval pair-wise comparison matrix to make k and m satisfy this condition.

$$\omega_j^\pm = \begin{bmatrix} \omega_{j,L}^\pm & \omega_{j,U}^\pm \end{bmatrix} = \begin{bmatrix} k\omega_j^L & m\omega_j^U \end{bmatrix} \tag{12.16}$$

where $\omega_j^\pm$ represents the interval weight of the jth criterion and $\omega_{j,\ L}^\pm$ and $\omega_{j,\ U}^\pm$ are the lower and upper bounds of $\omega_j^\pm$, respectively.

2.3 Interval gray relational analysis

The interval gray relational analysis (GRA) method was presented in the following six steps based on the work of Zhang (2005), Wang et al. (2017), and Manzardo et al. (2012).

Step 1: Establishing the interval decision-making matrix. This step is to determine the weights of the decision criteria by using the IAHP method and to collect the data of the alternatives with respect to the decision criteria.

Assuming that there are a total of n decision attributes $C_1, C_2, \ldots, C_n$ to assess the n alternatives, namely, $A_1, A_2, \ldots, A_m$, then, the interval decision-making matrix can be determined, as presented in Eq. (12.17).

$$X = \left|x_{ij}^{\pm}\right|_{m\times n} = \left|\begin{array}{ccccc} & C_1 & C_2 & \cdots & C_n \\ A_1 & x_{11}^{\pm} & x_{12}^{\pm} & \cdots & x_{1n}^{\pm} \\ A_2 & x_{21}^{\pm} & x_{22}^{\pm} & \vdots & x_{2n}^{\pm} \\ \vdots & \vdots & \cdots & \ddots & \vdots \\ A_m & x_{m1}^{\pm} & x_{m2}^{\pm} & \cdots & x_{mn}^{\pm} \end{array}\right| \tag{12.17}$$

where $x_{ij}^{\pm} = \left[x_{ij}^{-}\ x_{ij}^{+}\right]$ represents the value of the ith alternative with respect to the jth attribute.

Step 2: Normalizing the decision-making matrix. In order to avoid the effects caused by the unit gaps existing in the data of the interval decision-making matrix, all the data determined by step 1 can be normalized according to Eqs. (12.18), (12.19).

As for the benefit-type criteria,

$$r_{ij}^{\pm} = \left[r_{ij}^{-}\ r_{ij}^{+}\right] = \begin{cases} \dfrac{x_{ij}^{-}}{\sqrt{\dfrac{1}{n}\sum\limits_{j=1}^{m} x_{ij}^{+}2}} \\ \dfrac{x_{ij}^{+}}{\sqrt{\dfrac{1}{n}\sum\limits_{j=1}^{m} x_{ij}^{+}2}} \end{cases} \tag{12.18}$$

As for the cost-type criteria,

$$r_{ij}^{\pm} = \left[r_{ij}^{-}\ r_{ij}^{+}\right] = \begin{cases} \dfrac{1/x_{ij}^{+}}{\sqrt{\sum\limits_{j=1}^{m} \dfrac{1}{n}\left(1/x_{ij}^{-}\right)^2}} \\ \dfrac{1/x_{ij}^{-}}{\sqrt{\sum\limits_{j=1}^{m} \dfrac{1}{n}\left(1/x_{ij}^{-}\right)^2}} \end{cases} \tag{12.19}$$

Step 3: Determining the weighted normalized decision-making matrix. After determining the normalized decision-making matrix, the weighted normalized decision-making matrix, Eq. (12.20), can be obtained by incorporating the criterion weights.

$$V = \left| v_{ij}^{\pm} \right|_{m \times n} = \begin{vmatrix} & C_1 & C_2 & \cdots & C_n \\ A_1 & \omega_1^{\pm} r_{11}^{\pm} & \omega_2^{\pm} r_{12}^{\pm} & \cdots & \omega_n^{\pm} r_{1n}^{\pm} \\ A_2 & \omega_1^{\pm} r_{21}^{\pm} & \omega_2^{\pm} r_{22}^{\pm} & \vdots & \omega_n^{\pm} r_{2n}^{\pm} \\ \vdots & \vdots & \cdots & \ddots & \vdots \\ A_m & \omega_1^{\pm} r_{m1}^{\pm} & \omega_2^{\pm} r_{m2}^{\pm} & \cdots & \omega_n^{\pm} r_{mn}^{\pm} \end{vmatrix} \tag{12.20}$$

where $\omega_1^{\pm}$ represents the interval weight of the *j*th criterion determined by the interval AHP method.

Step 4: Determining the reference series. The reference series (RS) $v_j^{RS\pm} = \left[v_j^{RS-}\ v_j^{RS+} \right]$, $j = 1, 2, \ldots, n$ can be determined by Eqs. (12.21)–(12.23).

$$RS = \left(v_1^{RS\pm}, v_2^{RS\pm}, \ldots, v_n^{RS\pm} \right) \tag{12.21}$$

$$v_j^{RS+} = \max_{i=1,2,\ldots,m} v_{ij}^{+}, j = 1, 2, \ldots, n \tag{12.22}$$

$$v_j^{RS-} = \max_{i=1,2,\ldots,m} v_{ij}^{-}, j = 1, 2, \ldots, n \tag{12.23}$$

Step 5: Calculating the correlation coefficients of the series of each alternative to the reference series with respect to each criterion. ρ takes the value of 0.5 in this study.

$$\xi_i(j) = \frac{\min_i \min_j d\left(v_{ij}^{\pm}, v_j^{RS\pm}\right) + \rho \max_i \max_j d\left(v_{ij}^{\pm}, v_j^{RS\pm}\right)}{d\left(v_{ij}^{\pm}, v_j^{RS\pm}\right) + \rho \max_i \max_j d\left(v_{ij}^{\pm}, v_j^{RS\pm}\right)} \tag{12.24}$$

$d(v_{ij}^{\pm}, v_j^{RS\pm})$ represents the distance between $v_{ij}^{\pm}$ and $v_j^{RS\pm}$, and it can be determined according to the work of Wang et al. (2017).

Step 6: Determining the correlation degree of the series of each alternative to the reference series.

$$r_i = \sum_{j=1}^{n} \xi_{ij}, i = 1, 2, \ldots, m \tag{12.25}$$

After determining the correlation degree of the series of each alternative to the reference series, the priority sequence of the alternatives can be determined according to the rule that the greater the value of r_i, the more superior the alternative will be. Accordingly, the greater the value of r_i, the more sustainable the corresponding pathway for biofuel production will be.

3 Case study

In order to illustrate the developed interval multicriteria decision making method for sustainability prioritization of biofuel production pathways under uncertainties, three alternative bioethanol production pathways including wheat-, corn-, and cassava-based technologies for bioethanol production were studied by the developed method. Eight criteria including life cycle cost (LCC) in economic aspect, climate change (CC), terrestrial acidification (TA), human toxicity (H.Tox), particulate matter formation (PMF) in environmental aspect, and social benefits (SB), contribution to economic development (CED), and food security (FS) in social-political aspect were used for sustainability assessment of these three bioethanol production pathways, and the data were modified from the work of Ren et al. (2015b), as presented in Table 12.2.

The interval decision-making matrix can be determined by changing the data with 10% positive/negative derivations, and the results are presented in Table 12.3.

Table 12.2 The data of the alternative bioethanol production pathways with respect to the eight criteria (Ren et al., 2015b)

Criteria	Unit	Wheat-based	Corn-based	Cassava-based
CC	kg CO_2eq	5.746	0.461	1.662
TA	kg SO_2eq	2.806	0.166	0.834
H.Tox	kg1,4-dB eq	1.619	0.096	0.481
PMF	Kg PM_{10} eq	0.342	0.017	0.105
LCC	RMB Yuan	5220	4937	4259
SB	–	8.75	8.75	9.75
CED	–	7	8.75	9.75
FS	–	0.25	1.25	9.75

Table 12.3 The interval multicriteria decision making matrix for ranking the three alternative bioethanol production pathways

Criteria	Unit	Wheat-based	Corn-based	Cassava-based
CC	kg CO_2eq	[5.171 6.321]	[0.415 0.507]	[1.496 1.828]
TA	kg SO_2eq	[2.525 3.087]	[0.149 0.183]	[0.751 0.917]
H.Tox	kg1,4-dB eq	[1.457 1.781]	[0.086 0.106]	[0.433 0.529]
PMF	Kg PM_{10} eq	[0.308 0.376]	[0.015 0.019]	[0.095 0.116]
LCC	RMB Yuan	[4698 5742]	[4443.3 5430.7]	[3833.1 4684.9]
SB	–	[7.875 9.625]	[7.875 9.625]	[8.775 10.725]
CED	–	[6.3 7.7]	[7.875 9.625]	[8.775 10.725]
FS	–	[0.225 0.275]	[1.125 1.375]	[8.775 10.725]

The interval AHP was then employed to determine the interval weights of the three dimensions (environmental, economic, and social aspects) and the local interval weights of the criteria in each dimension. Taking the weights of the three dimensions as an example, the interval comparison matrix was first determined, as presented in Eq. (12.26).

$$\begin{array}{cccc} & \text{Environmental} & \text{Economic} & \text{Social} \\ \text{Environmental} & 1 & [3 \quad 5] & [2 \quad 4] \\ \text{Economic} & \left[\frac{1}{5} \quad \frac{1}{3}\right] & 1 & [1 \quad 2] \\ \text{Social} & \left[\frac{1}{4} \quad \frac{1}{2}\right] & \left[\frac{1}{2} \quad 1\right] & 1 \end{array} \tag{12.26}$$

The interval pair-wise comparison matrix (presented in Eq. 12.26) into two crisp nonnegative matrices, as presented in Eqs. (12.27), (12.28), respectively.

$$\begin{array}{cccc} & \text{Environmental} & \text{Economic} & \text{Social} \\ \text{Environmental} & 1 & 3 & 2 \\ \text{Economic} & \frac{1}{5} & 1 & 1 \\ \text{Social} & \frac{1}{4} & \frac{1}{2} & 1 \end{array} \tag{12.27}$$

$$\begin{array}{cccc} & \text{Environmental} & \text{Economic} & \text{Social} \\ \text{Environmental} & 1 & 5 & 4 \\ \text{Economic} & \frac{1}{3} & 1 & 2 \\ \text{Social} & \frac{1}{2} & 1 & 1 \end{array} \tag{12.28}$$

According to the geometric mean method, W_L and W_U can be determined, and the results are presented in Eqs. (12.29), (12.30), respectively.

$$W_L = [0.6262 \quad 0.2015 \quad 0.1723] \tag{12.29}$$

$$W_U = [0.6195 \quad 0.1994 \quad 0.1811] \tag{12.30}$$

According to Eqs. (12.14), (12.15), the values of m and k can be determined, and $m = 1.0779$, and $k = 0.9117$. It can satisfy $0 < k \leq 1 \leq m$, thus, the interval weights of the three dimensions can be determined by Eq.(12.16), and the results are presented in Eq. (12.31).

$$W^{\pm} = [[0.5709 \quad 0.6678], [0.1837 \quad 0.2149], [0.1571 \quad 0.1953]] \tag{12.31}$$

Table 12.4 The local weights of the four criteria in environmental aspect and the three criteria in social-political aspect

	CC	TA	H.Tox	PMF
CC	1	[1 3]	[3 5]	[3 5]
TA	[1/3 1]	1	[2 3]	[2 3]
H.Tox	[1/5 1/3]	[1/3 1/2]	1	[1 2]
PMF	[1/5 1/3]	[1/3 1/2]	[1/2 1]	1
	[0.4051 0.5414]	[0.2513 0.3187]	[0.1188 0.1398]	[0.0999 0.1176]
	SB	**CED**	**FS**	
SB	1	[1 2]	[1/3 1]	
CED	[1/2 1]	1	[1/5 1/3]	
FS	[1 3]	[3 5]	1	
Local weights	[0.2315 0.3190]	[0.1550 0.1756]	[0.4816 0.6244]	

Table 12.5 The global weights of the criteria for sustainability assessment of bioethanol production pathways

Dimension	Weights	Criteria	Local weights	Global weights
Environmental	[0.5709 0.6678]	CC	[0.4051 0.5414]	[0.2313 0.3615]
		TA	[0.2513 0.3187]	[0.1435 0.2128]
		H.Tox	[0.1188 0.1398]	[0.0678 0.0934]
		PMF	[0.0999 0.1176]	[0.0570 0.0785]
Economic	[0.1837 0.2149]	LCC	1	[0.1837 0.2149]
Social	[0.1571 0.1953]	SB	[0.2315 0.3190]	[0.0364 0.0623]
		CED	[0.1550 0.1756]	[0.0244 0.0343]
		FS	[0.4816 0.6244]	[0.0757 0.1219]

In a similar way, the local weights of the four criteria in environmental aspect and the three criteria in social-political aspect can also be determined, and the results are presented in Table 12.4.

After this, the global weights of the eights criteria for sustainability assessment of bioethanol production pathways can be determined, and the global weight of each criterion = the local weights of the each criterion × the weight of the corresponding dimension to which it belongs to. It is worth pointing out that there is only one criterion (LCC) in economic aspect, and the local weight of LCC is 1. The global weights of these eight criteria are presented in Table 12.5.

Then, the presented interval GRA method can be used to rank these three alternative bioethanol production pathways. According to Eqs. (12.18), (12.19), the data presented in Table 12.3 can be normalized. It is worth pointing out that the first five criteria including CC, TA, H. Tox, OMF, and LCC are cost-type criteria, thus, the data of the three biofuel production pathways with respect to these five criteria can be normalized by Eq. (12.19), and the other three criteria including SB, CEM, and FS are benefit-type criteria, thus, the data of the three biofuel production pathways with respect to these three criteria can be normalized by Eq. (12.18). The normalized decision-making matrix is presented in Table 12.6.

According to Eq. (12.20), the weighted normalized decision-making matrix can be obtained. Taking the data of wheat-based pathway for bioethanol production with respect to CC can be obtained by Eq. (12.32).

$$[0.1092\ 0.1335] \times [0.2313\ 0.3615] = [0.0253\ 0.0483] \tag{12.32}$$

In a similar way, all the data in the weighted normalized decision-making matrix can be obtained, and the results are presented in Table 12.7.

Table 12.6 The normalized decision-making matrix

Criteria	Wheat-based	Corn-based	Cassava-based
CC	[0.1092 0.1335]	[1.3615 1.6641]	[0.3777 0.4616]
TA	[0.0821 0.1003]	[1.3875 1.6959]	[0.2762 0.3375]
H.Tox	[0.0823 0.1005]	[1.3874 1.6957]	[0.2769 0.3384]
PMF	[0.0695 0.0849]	[1.3972 1.7077]	[0.2262 0.2765]
LCC	[0.7448 0.9103]	[0.7875 0.9625]	[0.9129 1.1157]
SB	[0.7871 0.9620]	[0.7871 0.9620]	[0.8771 1.0720]
CED	[0.6679 0.8163]	[0.8348 1.0203]	[0.9302 1.1369]
FS	[0.0360 0.0440]	[0.1802 0.2202]	[1.4052 1.7174]

Table 12.7 The weighted normalized decision-making matrix

Criteria	Wheat-based	Corn-based	Cassava-based
CC	[0.0253 0.0483]	[0.3149 0.6016]	[0.0874 0.1669]
TA	[0.0118 0.0213]	[0.1991 0.3609]	[0.0396 0.0718]
H.Tox	[0.0056 0.0094]	[0.0941 0.1584]	[0.0188 0.0316]
PMF	[0.0040 0.0067]	[0.0796 0.1341]	[0.0129 0.0217]
LCC	[0.1368 0.1956]	[0.1447 0.2068]	[0.1677 0.2398]
SB	[0.0287 0.0599]	[0.0287 0.0599]	[0.0319 0.0668]
CED	[0.0163 0.0280]	[0.0204 0.0350]	[0.0227 0.0390]
FS	[0.0027 0.0054]	[0.0136 0.0268]	[0.1064 0.2094]

Table 12.8 The reference series

Criteria	Reference series
CC	[0.3149 0.6016]
TA	[0.1991 0.3609]
H.Tox	[0.0941 0.1584]
PMF	[0.0796 0.1341]
LCC	[0.1677 0.2398]
SB	[0.0319 0.0668]
CED	[0.0227 0.0390]
FS	[0.1064 0.2094]

Table 12.9 The correlation degree of the series of each alternative to the reference series

	Wheat-based	Corn-based	Cassava-based
Correlation degree	5.4678	7.4517	6.2592
Ranking	3	1	2

According to Eqs. (12.21)–(12.23), the reference series can be obtained, and the results are presented in Table 12.8.

Finally, the correlation degree of the series of each alternative to the reference series can be determined by Eqs. (12.24)–(12.25), and the results are presented in Table 12.9.

Therefore the corn-based pathway for bioethanol production was recognized as the most sustainable, followed by cassava- and wheat-based pathways in the descending order.

4 Discussion

In order to investigate the effects of the weights of the criteria on the sustainability ranking of the three pathways for biofuel production, sensitivity analysis was carried out by changing the weights of the criteria for sustainability assessment of biofuel production pathways, and the following nine cases have been studied:

Case 0: Equal weights—an equal weight (0.1250) was assigned to the eight criteria for sustainability assessment of biofuel production pathways.

Case 1–8: A dominant weight (0.4400) was assigned to each of the eight criteria in each case and all the other nine criteria were assigned an equal weight (0.0800).

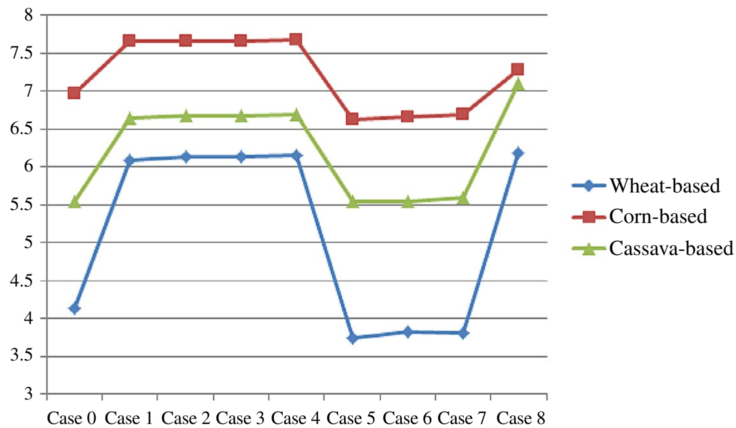

Fig. 12.2 The results of sensitivity analysis.

The results of sensitivity analysis are presented in Fig. 12.2. It is apparent that the results are robust; the sustainability sequence from the most sustainable to the least is corn-, cassava-, and wheat-based pathways for bioethanol production in all the nine cases.

5 Conclusion

This study developed an interval multicriteria decision making method for sustainability ranking of alternative biofuel production pathways, the interval AHP was employed to determine the weights (relative importance) of the criteria for sustainability assessment of biofuel production pathways, and the interval GRA method was employed to determine the sustainability sequence of the alternative biofuel production pathways. The developed interval multicriteria decision making method for sustainability ranking of alternative biofuel production pathways has the following two advantages:

(1) The use of interval AHP method can effectively address the ambiguity, hesitation, and vagueness existing in human's judgments when comparing the relative preference of a criterion over another.

(2) The use of interval GRA can achieve multicriteria decision making under uncertainties, because the data in the decision-making matrix are interval numbers rather than the traditional crisp numbers.

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CHAPTER 13

A multicriteria intuitionistic fuzzy group decision-making method for sustainability ranking of biofuel production pathways

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1 Introduction

In order to select the most sustainable biofuel production pathway among multiple options, and the investigation of the sustainability of different pathways for biofuel production, there are various studies focusing on sustainability assessment and measurement of biofuel production pathways. Liew et al. (2014) have carried out a comprehensive literature review of technologies and assessment methods on economic performance, safety, health and environment, and social impacts, and the typical methods for sustainability assessment of biofuel production were also evaluated. Mata et al. (2013) developed a sustainability evaluation methodology by using five indicators including life cycle energy efficiency, fossil energy ratio, contribution to global warming, land use intensity, and carbon stock change emissions for sustainability analysis of biofuels through the supply chain. Azapagic and

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Stichnothe (2011) reviewed several sustainability categories of biofuels and show how to determine the life cycle economic sustainability and environmental impacts of biofuels. All these methods can effectively be used for sustainability assessment of biofuel production pathways, but there are several problems needing to be resolved in future:

(1) It is difficult to collect some data of biofuel production pathways with respect to the criteria for sustainability assessment.

(2) The ambiguity and hesitations existing in human judgments cannot be addressed in these methods.

(3) The selection of the most sustainable biofuel production pathway usually involves multiple groups of different decision-makers/stakeholders with different preferences and opinions.

To address the earlier issues, group multicriteria decision-making method is introduced in this chapter.

With the increasing complexity and uncertainty of objectives and fast growth of the knowledge and information, it is difficult for a single decision-maker to effectively resolve the problems due to the limited knowledge and experience. It is needed to gather multiple decision-makers with different knowledge structures and experience to conduct a group decision-making (Kilgour and Eden, 2010). Thus group decision-making can be regarded as the process in which multiple decision-makers participate in decision-making analysis, gather individual judgments into group judgments, and then make decisions according to the group information. The concept of group decision-making was first proposed by Black in 1958. He divided group decision-making into two categories according to the decision-makers' code of conduct: collective decision-making and game problem (Black, 1958). Moving from single decision-maker to a multiple decision-maker setting introduces a great deal of complexity into the analysis. Hwang and Lin (1987) further defined the group decision-making into the analysis which is extended to account for the conflicts among different interests groups who have different objective, goals, and so forth. The preference of group members to give alternatives, and then based on a certain rule to become a group compromise or a consistent preference order.

Group multicriteria decision-making is created when different evaluation criteria are involved in group decision-making. Decision-makers judge each alternative based on different criteria and their own preferences. The individual preferences are aggregated into group preferences in order to evaluate, sort, and select the alternatives. Group multicriteria decision-making involves three basic processes (Safarzadeh et al., 2018): (1)

Aggregation of individual information. Summarize and aggregate the importance of each criterion and evaluation of each project from different decision-makers. The judgment information of single decision-maker is collected into the judgment criteria of group decision-makers. (2) Calculation of the criteria weights. Different criteria have different influences on the projects' evaluation. This difference is expressed by the weight of the criteria. (3) Evaluation and selection of the alternatives. Applying the group multicriteria decision-making methods to evaluate different objectives. The optimal project will be selected according to certain judgment principles.

In the group decision-making process, there are ubiquitous uncertainties of the objectives. Managing and modeling of uncertain information are vital for the acquisition of desirable solutions. To overcome this issue, fuzzy sets are introduced in a way to help linguistic variables be expressed appropriately (Zadeh, 1965). The fuzzy set uses the membership degree as a single index which reflects the state of support or opposition attitude of the decision-makers to the different objectives. It extends the eigenfunctions of the membership degree from integer 0 and 1 to the closed interval [0, 1]. The fuzzy set breaks through the logic shackles in conventional analysis methods and opens up a new field for decision-makers to deal with fuzzy information. However, with the development of decision-making theory and fuzzy set method, it is found that it will be difficult to accurately describe the uncertainty of objectives by simply relying on fuzzy sets. This is because the fuzzy set only involves the membership degree, but neglects the hesitation and the indeterminacy often involved in decision-making. To fully reflect the characteristics of affirmation, negation, and hesitation of human cognitive performance, the intuitionistic fuzzy set (IFS) was proposed by Atanassov (1986) and Atanassov and Gargov (1989). IFS is characterized by a membership function, a nonmembership function, and a hesitancy (indeterminacy) function. Compared with conventional fuzzy set, IFS has a better ability to accurately describe the natural attributes of objectives. Thus this method has gradually become the hotspot in the research area of fuzzy mathematics and decision-making for recent 30 years (Mardani et al., 2015), and has been widely used to describe the imprecise, vague, or uncertain preferences of the decision-makers in decision-making process (Xu and Zhao, 2016).

In the following research, the theory of IFS has been continuously improved. The development of IFS accelerates the application in solving the real-word problems. Recent applications of intuitionistic multicriteria decision-making methods are mainly used for management, evaluation, and prediction. For the application in management, it includes the water resources management (Hernandez and Uddameri, 2010), sustainable energy management (Boran et al., 2012), human resources management (Zhang and Liu, 2011), and so on. For the application in evaluation, it includes the supplier selection and evaluation (Boran et al., 2009), location selection (Devi and Yadav, 2013), strategy selection (Liu et al., 2012), web quality evaluation (Wang, 2009), network security evaluation (Li et al., 2011), performance evaluation (Zhang, 2012), and so on. And for the application in prediction, it includes air quality prediction (Yue et al., 2009), sentiment prediction (Wang et al., 2013), and so on.

Compared with a large number of theoretical studies, the applications of IFS are still at the start-up stage. Most of the advance IFS decision-making methods have not been applied in practice. More theoretical methods should be studied in the application of real words. Researchers mainly apply existing intuitionistic fuzzy decision-making methods to areas such as logistics management and resource management. Applications in other practical areas should also be explored.

In order to fill the gap of industrial application of IFS, this chapter aims at developing a multicriteria intuitionistic fuzzy group decision-making method for sustainability ranking of alternative bioethanol production pathways based on intuitionistic fuzzy theory. The intuitionistic fuzzy set is flexible to manage the vagueness, and it enables the users to reduce the loss of the original information and can effectively assure the credibility of the decision-making results according to the opinions and preferences of the decision-makers.

2 Methods

The multicriteria group decision-making method for sustainability ranking proposed in this chapter is based on the intuitionistic fuzzy set. For better readability, the basic conception of the intuitionistic fuzzy set is first introduced in Section 2.1. The detailed methodology of the multicriteria group decision-making is then introduced in Section 2.2.

2.1 Basics of intuitionistic fuzzy set

Definition 1 Intuitionistic fuzzy set (Atanassov, 1986)
The intuitionistic fuzzy set (IFS) was developed by Atanassov (1986), and the definition of IFS is presented in Eq. (13.1)

$$\alpha = \{(x, \mu_\alpha(x), \upsilon_\alpha(x)) | x \in X\} \tag{13.1}$$

where $\mu_\alpha(x): X \rightarrow [0,1]$, $x \in X \rightarrow \mu_\alpha(x) \in [0,1]$ and it represents the degree of membership of the element $x \in X$ to the set α, and $\upsilon_\alpha(x): X \rightarrow [0,1]$, $x \in X \rightarrow \upsilon_\alpha(x) \in [0,1]$ and it represents the degree of nonmembership of the element $x \in X$ to the set α, and they satisfy the condition of $0 \leq \mu_\alpha(x) + \upsilon_\alpha(x) \leq 1$ for all $x \in X$.

Besides the membership and nonmembership, the degree of indeterminacy of $x \in X$ to the set α can be determined by Eq. (13.2)

$$\pi_\alpha(x) = 1 - \mu_\alpha(x) - \upsilon_\alpha(x), x \in X \tag{13.2}$$

It is worth pointing out that $\pi_\alpha(x)$ can be recognized as a measure of the certainty of knowledge about x, and the smaller the value of $\pi_\alpha(x)$, the more certainty the knowledge about x (Boran et al., 2009).

Definition 2 Arithmetic operations
Assuming that there are two intuitionistic fuzzy numbers α denoted by $\alpha = (\mu_\alpha, \upsilon_\alpha, \pi_\alpha)$, where $\mu_\alpha \in [0,1]$, $\upsilon_\alpha \in [0,1]$, $\mu_\alpha(x) + \upsilon_\alpha(x) \leq 1$, and $\pi_\alpha = 1 - \mu_\alpha - \upsilon_\alpha$ and β denoted by $\beta = (\mu_\beta, \upsilon_\beta, \pi_\beta)$, where $\mu_\beta \in [0,1]$, $\upsilon_\beta \in [0,1]$, $\mu_\beta(x) + \upsilon_\beta(x) \leq 1$, and $\pi_\beta = 1 - \mu_\beta - \upsilon_\beta$, the arithmetic operations including addition, multiplication, and scale multiplication between α and β are presented in Table 13.1.

Definition 3 Score, accuracy, and indeterminancy
The score, accuracy, and indeterminancy of the intuitionistic fuzzy number $\alpha = (\mu_\alpha, \upsilon_\alpha, \pi_\alpha)$ can be determined by Eqs. (13.9)–(13.11), respectively.

$$S(\alpha) = \mu_\alpha - \upsilon_\alpha \tag{13.9}$$

$$A(\alpha) = \mu_\alpha + \upsilon_\alpha \tag{13.10}$$

$$I(\alpha) = \mu_\alpha - \upsilon_\alpha \tag{13.11}$$

where $S(\alpha)$, $A(\alpha)$, and $I(\alpha)$ are the score, accuracy, and indeterminancy of the intuitionistic fuzzy number $\alpha = (\mu_\alpha, \upsilon_\alpha, \pi_\alpha)$.

Definition 4 Comparisons of intuitionistic fuzzy numbers (Beliakov et al., 2011)
As for the two intuitionistic fuzzy numbers $\alpha = (\mu_\alpha, \upsilon_\alpha, \pi_\alpha)$ and $\beta = (\mu_\beta, \upsilon_\beta, \pi_\beta)$, a defuzzied score can be determined for these two intuitionistic fuzzy numbers to identify the preference relationship between them based on the work of Szmidt and Kacprzyk (2009) and that of Liao and Xu (2014), as presented in Eqs. (13.12), (13.13).

Table 13.1 Arithmetic operations between two intuitionistic fuzzy numbers

Operations	Formulas	References
Addition	$\alpha \oplus \beta = (\mu_\alpha, \upsilon_\alpha, \pi_\alpha) \oplus (\mu_\beta, \upsilon_\beta, \pi_\beta) = (\mu_\alpha + \mu_\beta - \mu_\alpha \mu_\beta, \upsilon_\alpha \upsilon_\beta)$ (13.3) $\oplus_{j=1}^{n} \alpha_j = \oplus_{j=1}^{n} \left(\mu_{\alpha_j}, \upsilon_{\alpha_j}, \pi_{\alpha_j}\right) = \left(1 - \prod_{j=1}^{n} \left(1 - \mu_{\alpha_j}\right), \prod_{j=1}^{n} \upsilon_{\alpha_j}\right)$ (13.4) where $\alpha_j (j = 1, 2, \ldots, n)$ are intuitionistic fuzzy numbers	Atanassov (1986) Xu (2007a)
Multiplication	$\alpha \otimes \beta = (\mu_\alpha, \upsilon_\alpha, \pi_\alpha) \otimes (\mu_\beta, \upsilon_\beta, \pi_\beta) = (\mu_\alpha \mu_\beta, \upsilon_\alpha + \upsilon_\beta - \upsilon_\alpha \upsilon_\beta)$ (13.5) $\otimes_{j=1}^{n} \alpha_j = \otimes_{j=1}^{n} \left(\mu_{\alpha_j}, \upsilon_{\alpha_j}, \pi_{\alpha_j}\right) = \left(\prod_{j=1}^{n} \mu_{\alpha_j}, \prod_{j=1}^{n} \left(1 - \upsilon_{\alpha_j}\right)\right)$ (13.6) where $\alpha_j (j = 1, 2, \ldots, n)$ are intuitionistic fuzzy numbers	Atanassov (1986) Xu (2007a)
	$\lambda \alpha = (1 - (1 - \mu_\alpha)^\lambda, (\upsilon_\alpha)^\lambda)$ (13.7) where λ represents any positive real numbers	De et al. (2000)
Exponentiation	$\alpha^n = ((\mu_\alpha)^n, 1 - (1 - \upsilon_\alpha)^n)$ (13.8) where n represents any positive real numbers	De et al. (2000)

$$\rho(\alpha) = \frac{1}{0.5(1+\pi_\alpha)(1-\mu_\alpha)} \tag{13.12}$$

$$\rho(\beta) = \frac{1}{0.5(1+\pi_\beta)(1-\mu_\beta)} \tag{13.13}$$

(1) $\alpha = (\mu_\alpha, \upsilon_\alpha, \pi_\alpha)$ is superior to $\beta = (\mu_\beta, \upsilon_\beta, \pi_\beta)$ when $\rho(\alpha) > \rho(\beta)$.
(2) $\alpha = (\mu_\alpha, \upsilon_\alpha, \pi_\alpha)$ is equivalent to $\beta = (\mu_\beta, \upsilon_\beta, \pi_\beta)$ when $\rho(\alpha) = \rho(\beta)$.
(3) $\alpha = (\mu_\alpha, \upsilon_\alpha, \pi_\alpha)$ is inferior to $\beta = (\mu_\beta, \upsilon_\beta, \pi_\beta)$ when $\rho(\alpha) < \rho(\beta)$.

Definition 5 Distance between two intuitionistic fuzzy numbers (Xu and Yager, 2008)
Suppose $\alpha = (\mu_\alpha, \upsilon_\alpha, \pi_\alpha)$ and $\beta = (\mu_\beta, \upsilon_\beta, \pi_\beta)$, the distance between α and β can be determined by Eq. (13.14).

$$d(\alpha, \beta) = \frac{1}{2}\left(\left|\mu_\alpha - \mu_\beta\right| + \left|\upsilon_\alpha - \upsilon_\beta\right| + \left|\pi_\alpha - \pi_\beta\right|\right) \tag{13.14}$$

Definition 6 Similarity between two intuitionistic fuzzy sets (Xu, 2007b)
Assume that $X = \{\alpha_1, \alpha_2, \ldots, \alpha_n\}$ be a universe of discourse, the similarity between two intuitionistic fuzzy sets $A = \{\langle \alpha_j, \mu_A(\alpha_j), \upsilon_A(\alpha_j)\rangle \mid \alpha_j \in X\}$ and $B = \{\langle \beta_j, \mu_A(\beta_j), \upsilon_A(\beta_j)\rangle \mid \beta_j \in X\}$ can be determined by Eq. (13.15).

$$S(A, B) = 1 - \left[\frac{\sum_{j=1}^{n}\left(\left|\mu_A(\alpha_j) - \mu_B(\beta_j)\right|^2 + \left|\upsilon_A(\alpha_j) - \upsilon_B(\beta_j)\right|^2 + \left|\pi_A(\alpha_j) - \pi_B(\beta_j)\right|^2\right)}{\sum_{j=1}^{n}\left(\left|\mu_A(\alpha_j) + \mu_B(\beta_j)\right|^2 + \left|\upsilon_A(\alpha_j) + \upsilon_B(\beta_j)\right|^2 + \left|\pi_A(\alpha_j) + \pi_B(\beta_j)\right|^2\right)}\right]^{\frac{1}{2}} \tag{13.15}$$

2.2 Similarity measure-based multicriteria decision-making method

The multicriteria intuitionistic fuzzy group decision-making (MCIFGDM) method for sustainability ranking of biofuel production pathways was developed in this section, and the MCIFGDM method was based on the similarity measure. Let $\{A_1, A_2, \ldots, A_m\}$ be a discrete set of alternative biofuel production pathways, $\{C_1, C_2, \ldots, C_n\}$ be a set of criteria for sustainability assessment of biofuel production pathways, the developed MCIFGDM method was presented in the following steps:

Step 1: Determine the intuitionistic fuzzy decision-making matrix according to the opinions of different stakeholders/decision-makers (Ren and Liang, 2017).

Assume that a total of K groups of decision-makers/stakeholders participate in assessing the m biofuel production pathways $\{A_1, A_2, \ldots, A_m\}$ by using the n criteria $\{C_1, C_2, \ldots, C_n\}$, they were first asked to use extreme poor (EP), very poor (VP), poor (P), medium poor (MP), fair (F), medium good (MG), good (G), very good (VG), and extreme good (EG) (as presented in Table 13.1) to rate the m alternative biofuel production pathways with respect to each of the n criteria for sustainability, and these natural words can be transformed into intuitionistic fuzzy numbers according to Table 13.2.The decision-making matrix by the kth decision-maker is presented in Eq. (13.16).

$$X^k = \begin{array}{c} \\ A_1 \\ A_2 \\ \vdots \\ A_m \end{array} \begin{array}{cccc} C_1 & C_2 & & C_n \\ \left(\mu_{11}^{x,k}, \upsilon_{11}^{x,k}, \pi_{11}^{x,k}\right) & \left(\mu_{12}^{x,k}, \upsilon_{12}^{x,k}, \pi_{12}^{x,k}\right) & \cdots & \left(\mu_{1n}^{x,k}, \upsilon_{1n}^{x,k}, \pi_{1n}^{x,k}\right) \\ \left(\mu_{21}^{x,k}, \upsilon_{21}^{x,k}, \pi_{21}^{x,k}\right) & \left(\mu_{22}^{x,k}, \upsilon_{22}^{x,k}, \pi_{22}^{x,k}\right) & \cdots & \left(\mu_{2n}^{x,k}, \upsilon_{2n}^{x,k}, \pi_{2n}^{x,k}\right) \\ \vdots & \vdots & \ddots & \vdots \\ \left(\mu_{m1}^{x,k}, \upsilon_{m1}^{x,k}, \pi_{m1}^{x,k}\right) & \left(\mu_{m2}^{x,k}, \upsilon_{m2}^{x,k}, \pi_{m2}^{x,k}\right) & \cdots & \left(\mu_{mn}^{x,k}, \upsilon_{mn}^{x,k}, \pi_{mn}^{x,k}\right) \end{array} \tag{13.16}$$

where X^k represents the decision-making matrix determined by the kth decision-makers, and $(\mu_{ij}^{x,\ k}, \upsilon_{ij}^{x,\ k}, \pi_{ij}^{x,\ k})$ represents the performance of the ith biofuel production pathway with respect to the jth criterion by the kth decision-maker.

Step 2: Determine the weights of the criteria for sustainability assessment according to the opinions of different stakeholders/decision-makers.

Table 13.2 The linguistic variables and corresponding intuitionistic fuzzy numbers (Zhang and Liu, 2011)

Linguistic terms	Abbreviation	Intuitionistic fuzzy numbers
Extreme poor (extreme low)	EP (EL)	(0.05,0.95,0.00)
Very poor (very low)	VP (VL)	(0.15,0.80,0.05)
Poor (low)	P (L)	(0.25,0.65,0.10)
Medium poor (medium low)	MP (ML)	(0.35,0.55,0.10)
Fair (medium)	F (M)	(0.50,0.40,0.10)
Medium good (medium high)	MG (MH)	(0.65,0.25,0.10)
Good (high)	G (H)	(0.75,0.15,0.10)
Very good (very high)	VG (VH)	(0.85,0.10,0.05)
Extreme good (extreme high)	EG (EH)	(0.95,0.05,0.00)

The decision-makers were asked to use the linguistic terms including extreme low (low), very low (VL), low (L), medium low (ML), medium (M), medium high (MH), high (H), very high (VH), and extreme high (EH) (as presented in Table 13.2) to evaluate the relative importance of the n criteria for sustainability, and these linguistic terms can be transformed into intuitionistic fuzzy numbers. The weights of the n criteria determined by the kth decision-maker are presented in Eq. (13.17).

$$W^k = \left[\left(\mu_1^{W,k}, \mu_1^{W,k}, \pi_1^{W,k}\right) \left(\mu_2^{W,k}, \mu_2^{W,k}, \pi_2^{W,k}\right) \cdots \left(\mu_1^{W,k}, \mu_1^{W,k}, \pi_1^{W,k}\right)\right] \tag{13.17}$$

where W^k represents the weight vector determined by the kth decision-maker, and $(\mu_j^{W,k}, \mu_j^{W,k}, \pi_j^{W,k})$ represents the weight of the jth criterion determined by the kth decision-maker.

Step 3: Determine the weighted intuitionistic fuzzy decision-making matrix determined by each decision-maker. The weighted intuitionistic fuzzy decision-making matrix determined by the kth decision-makers can be determined by Eqs. (13.18), (13.19)

$$Y^k = \begin{array}{c} \\ A_1 \\ A_2 \\ \vdots \\ A_m \end{array} \begin{array}{cccc} C_1 & C_2 & & C_n \\ \left(\mu_{11}^{y,k}, \upsilon_{11}^{y,k}, \pi_{11}^{y,k}\right) & \left(\mu_{12}^{y,k}, \upsilon_{12}^{y,k}, \pi_{12}^{y,k}\right) & \cdots & \left(\mu_{1n}^{y,k}, \upsilon_{1n}^{y,k}, \pi_{1n}^{y,k}\right) \\ \left(\mu_{21}^{y,k}, \upsilon_{21}^{y,k}, \pi_{21}^{y,k}\right) & \left(\mu_{22}^{y,k}, \upsilon_{22}^{y,k}, \pi_{22}^{y,k}\right) & \cdots & \left(\mu_{2n}^{y,k}, \upsilon_{2n}^{y,k}, \pi_{2n}^{y,k}\right) \\ \vdots & \vdots & \ddots & \vdots \\ \left(\mu_{m1}^{y,k}, \upsilon_{m1}^{y,k}, \pi_{m1}^{y,k}\right) & \left(\mu_{m2}^{y,k}, \upsilon_{m2}^{y,k}, \pi_{m2}^{y,k}\right) & \cdots & \left(\mu_{mn}^{y,k}, \upsilon_{mn}^{y,k}, \pi_{mn}^{y,k}\right) \end{array} \tag{13.18}$$

$$\begin{aligned} \left(\mu_{ij}^{y,k}, \upsilon_{ij}^{y,k}, \pi_{ij}^{y,k}\right) &= \left(\mu_j^{W,k}, \upsilon_j^{W,k}, \pi_j^{W,k}\right) \otimes \left(\mu_{ij}^{x,k}, \upsilon_{ij}^{x,k}, \pi_{ij}^{x,k}\right) \\ &= \left(\mu_j^{W,k}\mu_{ij}^{x,k}, \upsilon_j^{W,k} + \upsilon_{ij}^{x,k} - \upsilon_j^{W,k}\upsilon_{ij}^{x,k}, 1 + \upsilon_j^{W,k}\upsilon_{ij}^{x,k} - \mu_j^{W,k}\mu_{ij}^{x,k} - \upsilon_j^{W,k} - \upsilon_{ij}^{x,k}\right) \end{aligned} \tag{13.19}$$

where Y^k represents the weighted decision-making matrix determined by the kth decision-makers, and $(\mu_{ij}^{y,k}, \upsilon_{ij}^{y,k}, \pi_{ij}^{y,k})$ represents the data in cell (i, j) in the weighted decision-making matrix.

Step 4: Determine the aggregated decision-making matrix.

The relative importance of the decision-makers can be rated according to the linguistic terms including very important (VI), important (I), medium (M), unimportant (U), and very unimportant (VU), and these linguistic variables can be transformed into intuitionistic fuzzy numbers according to Table 13.3.

Table 13.3 Linguistic variables for rating the role of the decision-makers (Zhang and Liu, 2011)

Linguistic terms	Abbreviation	IFN
Very important	VI	(0.90,0.05,0.05)
Important	I	(0.75,0.20,0.05)
Medium	M	(0.50,0.40,0.10)
Unimportant	U	(0.25,0.60,0.15)
Very unimportant	VU	(0.10,0.80,0.10)

Assume that $R_k = (\mu_k^R, \upsilon_k^R, \pi_k^R)$ is the role importance of the kth decision-maker described by the intuitionistic fuzzy numbers, and the role importance of each decision-maker in crisp numbers can be determined by Eq. (13.20) (Boran et al., 2009).

$$\lambda_k = \frac{\left(\mu_k^R + \pi_k^R\left(\frac{\mu_k^R}{\mu_k^R + \upsilon_k^R}\right)\right)}{\sum_{k=1}^{K}\left(\mu_k^R + \pi_k^R\left(\frac{\mu_k^R}{\mu_k^R + \upsilon_k^R}\right)\right)} \tag{13.20}$$

where λ_k represents the role weight of the kth decision-maker, and $\lambda_1 + \lambda_2 + \cdots + \lambda_K = 1$.

The decision-making matrices presented in Eq. (13.18) can be aggregated into a unique decision-making matrix by using the intuitionistic fuzzy weighted averaging (IFWA) operator (Büyüközkan and Göçer, 2017), see Eqs. (13.21), (13.22).

$$\begin{aligned}\left(\mu_{ij}^{y}, \upsilon_{ij}^{y}, \pi_{ij}^{y}\right) &= IFWA_{\lambda}\left(\left(\mu_{ij}^{y,1}, \upsilon_{ij}^{y,1}, \pi_{ij}^{y,1}\right), \left(\mu_{ij}^{y,2}, \upsilon_{ij}^{y,2}, \pi_{ij}^{y,2}\right), \ldots, \left(\mu_{ij}^{y,K}, \upsilon_{ij}^{y,K}, \pi_{ij}^{y,K}\right)\right) \\ &= \left(1 - \prod_{k=1}^{K}\left(1-\mu_{ij}^{y,k}\right)^{\lambda_k}, \prod_{k=1}^{K}\left(\upsilon_{ij}^{y,k}\right)^{\lambda_k}, \prod_{k=1}^{K}\left(1-\mu_{ij}^{y,k}\right)^{\lambda_k} - \prod_{k=1}^{K}\left(\upsilon_{ij}^{y,k}\right)^{\lambda_k}\right)\end{aligned} \tag{13.21}$$

$$Y = \begin{array}{c|cccc} & C_1 & C_2 & & C_n \\ A_1 & (\mu_{11}^{y}, \upsilon_{11}^{y}, \pi_{11}^{y}) & (\mu_{12}^{y}, \upsilon_{12}^{y}, \pi_{12}^{y}) & \cdots & (\mu_{1n}^{y}, \upsilon_{1n}^{y}, \pi_{1n}^{y}) \\ A_2 & (\mu_{21}^{y}, \upsilon_{21}^{y}, \pi_{21}^{y}) & (\mu_{22}^{y}, \upsilon_{22}^{y}, \pi_{22}^{y}) & \cdots & (\mu_{2n}^{y}, \upsilon_{2n}^{y}, \pi_{2n}^{y}) \\ \vdots & \vdots & \vdots & \ddots & \vdots \\ A_m & (\mu_{m1}^{y}, \upsilon_{m1}^{y}, \pi_{m1}^{y}) & (\mu_{m2}^{y}, \upsilon_{m2}^{y}, \pi_{m2}^{y}) & \cdots & (\mu_{mn}^{y}, \upsilon_{mn}^{y}, \pi_{mn}^{y}) \end{array} \tag{13.22}$$

where Y represents the aggregated decision-making matrix, and $(\mu_{ij}^{y}, \upsilon_{ij}^{y}, \pi_{ij}^{y})$ represents the aggregated score of the ith biofuel production pathway with respect to the jth criterion.

Step 5: Determine the ideal and antiideal intuitionistic fuzzy solutions. The ideal and antiideal intuitionistic fuzzy solutions after determining the aggregated decision-making matrix according to Xu (2007b), as presented in Eqs. (13.23)–(13.26).

$$A^{+} = \left\{a_1^{+}, a_2^{+}, \ldots, a_n^{+}\right\} \tag{13.23}$$

$$a_j^{+} = \left(\max_{i=1}^{m} \mu_{i1}^{\gamma},\ \min_{i=1}^{m} \upsilon_{i1}^{\gamma},\ 1 - \max_{i=1}^{m} \mu_{i1}^{\gamma} - \min_{i=1}^{m} \upsilon_{i1}^{\gamma}\right) \tag{13.24}$$

$$A^{-} = \left\{a_1^{-}, a_2^{-}, \ldots, a_n^{-}\right\} \tag{13.25}$$

$$a_j^{-} = \left(\min_{i=1}^{m} \mu_{i1}^{\gamma},\ \max_{i=1}^{m} \upsilon_{i1}^{\gamma},\ 1 - \min_{i=1}^{m} \mu_{i1}^{\gamma} - \max_{i=1}^{m} \upsilon_{i1}^{\gamma}\right) \tag{13.26}$$

Step 6: Determine the degree of the similarity of each alternative and the ideal intuitionistic fuzzy solution and that of each alternative and the anti-ideal intuitionistic fuzzy solution.

The degree of the similarity of each alternative and the ideal intuitionistic fuzzy solution and that of each alternative and the antiideal intuitionistic fuzzy solution can be determined by Eqs. (13.27), (13.28).

$$S(A_i, A^{+}) = 1 - \left[\frac{\sum_{j=1}^{n}\left(\left|\mu_{ij}^{\gamma} - \max_{i=1}^{m}\mu_{ij}^{\gamma}\right|^2 + \left|\upsilon_{ij}^{\gamma} - \min_{i=1}^{m}\upsilon_{ij}^{\gamma}\right|^2 + \left|\pi_{ij}^{\gamma} - \left(1 - \max_{i=1}^{m}\mu_{i1}^{\gamma} - \min_{i=1}^{m}\upsilon_{i1}^{\gamma}\right)\right|^2\right)}{\sum_{j=1}^{n}\left(\left|\mu_{ij}^{\gamma} + \max_{i=1}^{m}\mu_{ij}^{\gamma}\right|^2 + \left|\upsilon_{ij}^{\gamma} + \min_{i=1}^{m}\upsilon_{ij}^{\gamma}\right|^2 + \left|\pi_{ij}^{\gamma} + \left(1 - \max_{i=1}^{m}\mu_{i1}^{\gamma} - \min_{i=1}^{m}\upsilon_{i1}^{\gamma}\right)\right|^2\right)}\right]^{\frac{1}{2}} \tag{13.27}$$

$$S(A_i, A^{-}) = 1 - \left[\frac{\sum_{j=1}^{n}\left(\left|\mu_{ij}^{\gamma} - \min_{i=1}^{m}\mu_{ij}^{\gamma}\right|^2 + \left|\upsilon_{ij}^{\gamma} - \max_{i=1}^{m}\upsilon_{ij}^{\gamma}\right|^2 + \left|\pi_{ij}^{\gamma} - \left(1 - \min_{i=1}^{m}\mu_{i1}^{\gamma} - \max_{i=1}^{m}\upsilon_{i1}^{\gamma}\right)\right|^2\right)}{\sum_{j=1}^{n}\left(\left|\mu_{ij}^{\gamma} + \min_{i=1}^{m}\mu_{ij}^{\gamma}\right|^2 + \left|\upsilon_{ij}^{\gamma} + \max_{i=1}^{m}\upsilon_{ij}^{\gamma}\right|^2 + \left|\pi_{ij}^{\gamma} + \left(1 - \min_{i=1}^{m}\mu_{i1}^{\gamma} - \max_{i=1}^{m}\upsilon_{i1}^{\gamma}\right)\right|^2\right)}\right]^{\frac{1}{2}} \tag{13.28}$$

where $S(A_i, A^{+})$ and $S(A_i, A^{-})$ represent the similarity of the *j*th alternative and the ideal intuitionistic fuzzy solution and that of the *j*th alternative and the antiideal intuitionistic fuzzy solution.

Step 7: Determine the relative similarity measure of each alternative and rank the alternatives.

The relative similarity measure of the *i*th alternative can be determined by Eq. (13.29) (Xu, 2007b).

$$d_i = \frac{S(A_i, A^+)}{S(A_i, A^+) + S(A_i, A^-)} \tag{13.29}$$

where d_i represents the relative similarity measure of the ith alternative.

After determining the relative similarity measures of all the alternatives, they can be ranked, and it is obvious that the bigger the relative similarity measure, the more superior the alternative.

The framework of the multicriteria intuitionistic fuzzy group decision-making method for sustainability ranking is shown in Fig. 13.1.

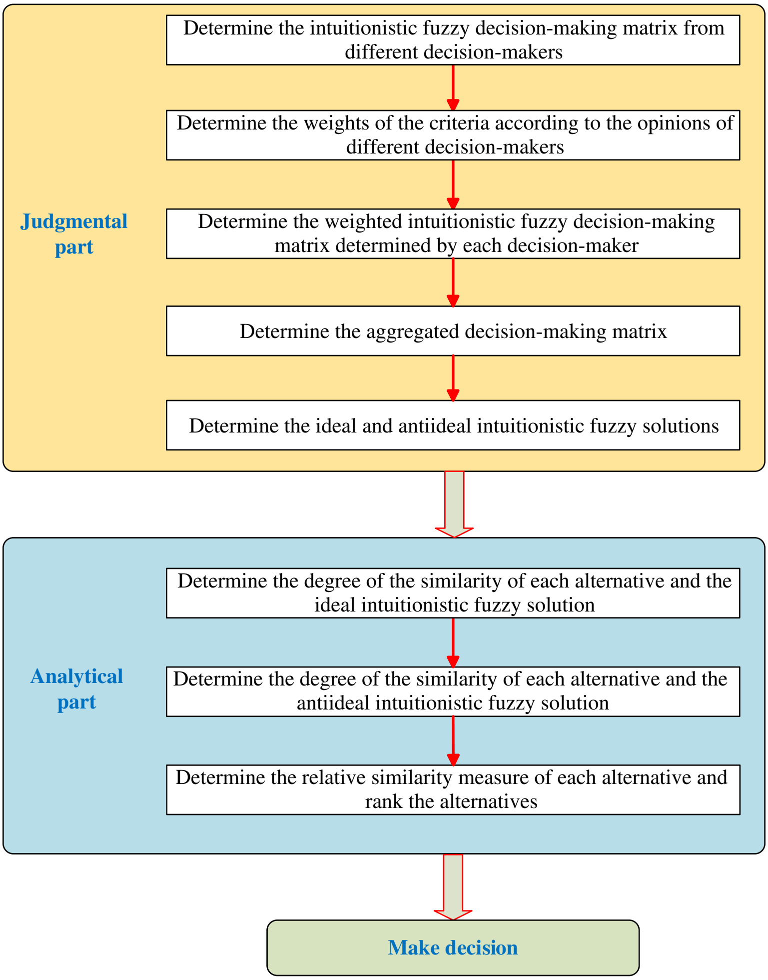

Fig. 13.1 The framework of the multicriteria intuitionistic fuzzy group decision-making method.

3 Case study

In order to illustrate the developed multicriteria intuitionistic fuzzy group decision-making method for sustainability ranking of biofuel production pathway, three scenarios for bioethanol were investigated by the developed multicriteria intuitionistic fuzzy group decision-making method, and they are corn-, wheat-, and cassava-based technologies for bioethanol production. The criteria in three categories including economic, environmental, technological, and social-political aspects were used to rank these three biofuel production pathways. Life cycle cost (LCC) is the only criterion in economic aspect to measure economic performance. Four criteria including climate change (CC), terrestrial acidification (TA), human toxicity (H. Tox), and particulate matter formation (PMF) were employed to measure environmental performances. Technology maturity (TM) is used to measure technology advance. Social benefits (SB), contribution to economic development (CED), and food security (FS) were used in social-political category.

Three groups of decision-makers/stakeholders were invited to participate in the decision-making process, and they are investor group (DM#1), engineer group (DM#2), and user group (DM#3). The representative stakeholder in each group was asked to use the linguistic terms presented in Table 13.2 to rate the three alternative pathways with respect to each criterion and determine the relative importance of these nine criteria for sustainability assessment of biofuel production pathways, and the results are presented in Tables (13.4)–(13.6).

According to Table 13.2, all the linguistic terms presented in Tables 13.4–13.6 can be transformed into intuitionistic fuzzy numbers, and the results are presented in Tables 13.7–13.9.

According to Eqs. (13.18), (13.19), the three weighted decision-making matrices can be determined according to the preferences and opinions of each group. Taking the data of cell (1,1) in the weighted decision-making matrix determined by DM#1 as an example:

$$\begin{aligned}&(0.35, 0.55, 0.10)\otimes(0.95, 0.05, 0)\\&=(0.35\times0.95, 0.55+0.05-0.55\times0.05, 1+0.55\times0.05\\&-0.35\times0.95-0.55-0.05)=(0.3325, 0.5725, 0.0950)\end{aligned}$$

In a similar way, all the three weighted decision-making matrices can be determined. The role importance of the three decision-maker groups including investor group (DM#1), engineer group (DM#2), and user group (DM#3) is recognized as very important (VI), important (I), and

Table 13.4 The relative performances of the three pathways for bioethanol production with respect to each criterion and the relative weights using linguistic terms by DM#1

	Wheat-based	Corn-based	Cassava-based	Relative importance
LCC	MP	F	G	EH
CC	MP	EG	G	VH
TA	P	VG	F	H
H.Tox	MP	EG	G	M
PMF	EP	EG	F	M
TM	G	G	F	VH
SB	F	F	G	L
CED	F	G	VG	M
FS	VP	MP	VG	H

Table 13.5 The relative performances of the three pathways for bioethanol production with respect to each criterion and the relative weights using linguistic terms by DM#2

	Wheat-based	Corn-based	Cassava-based	Relative importance
LCC	P	F	MG	H
CC	MP	VG	G	VH
TA	MP	G	F	VH
H.Tox	P	VG	G	VH
PMF	VP	VG	MP	VH
TM	F	F	MP	M
SB	MG	MG	G	M
CED	MG	G	VG	M
FS	P	MP	G	M

Table 13.6 The relative performances of the three pathways for bioethanol production with respect to each criterion and the relative weights using linguistic terms by DM#3

	Wheat-based	Corn-based	Cassava-based	Relative importance
LCC	MP	MP	F	H
CC	MP	VG	F	H
TA	MP	G	F	H
H. Tox	P	VG	G	H
PMF	P	VG	G	M
TM	MG	MG	F	VH
SB	MG	MG	VG	VH
CED	F	MG	G	VH
FS	EP	P	MG	VH

Table 13.7 The relative performances of the three pathways for bioethanol production with respect to each criterion and the relative weights using intuitionistic fuzzy numbers by DM#1

	Wheat-based	Corn-based	Cassava-based	Relative importance
LCC	(0.35,0.55,0.10)	(0.50,0.40,0.10)	(0.75,0.15,0.10)	(0.95,0.05,0.00)
CC	(0.35,0.55,0.10)	(0.95,0.05,0.00)	(0.75,0.15,0.10)	(0.85,0.10,0.05)
TA	(0.25,0.65,0.10)	(0.85,0.10,0.05)	(0.50,0.40,0.10)	(0.75,0.15,0.10)
H.Tox	(0.35,0.55,0.10)	(0.95,0.05,0.00)	(0.75,0.15,0.10)	(0.50,0.40,0.10)
PMF	(0.05,0.95,0.00)	(0.95,0.05,0.00)	(0.50,0.40,0.10)	(0.50,0.40,0.10)
TM	(0.75,0.15,0.10)	(0.75,0.15,0.10)	(0.50,0.40,0.10)	(0.85,0.10,0.05)
SB	(0.50,0.40,0.10)	(0.50,0.40,0.10)	(0.75,0.15,0.10)	(0.25,0.65,0.10)
CED	(0.50,0.40,0.10)	(0.75,0.15,0.10)	(0.85,0.10,0.05)	(0.50,0.40,0.10)
FS	(0.15,0.80,0.05)	(0.35,0.55,0.10)	(0.85,0.10,0.05)	(0.75,0.15,0.10)

Table 13.8 The relative performances of the three pathways for bioethanol production with respect to each criterion and the relative weights using intuitionistic fuzzy numbers by DM#2

	Wheat-based	Corn-based	Cassava-based	Relative importance
LCC	0.25,0.65,0.10	0.50,0.40,0.10	0.65,0.25,0.10	0.75,0.15,0.10
CC	0.35,0.55,0.10	0.85,0.10,0.05	0.75,0.15,0.10	0.85,0.10,0.05
TA	0.35,0.55,0.10	0.75,0.15,0.10	0.50,0.40,0.10	0.85,0.10,0.05
H.Tox	0.25,0.65,0.10	0.85,0.10,0.05	0.75,0.15,0.10	0.85,0.10,0.05
PMF	0.15,0.80,0.05	0.85,0.10,0.05	0.35,0.55,0.10	0.85,0.10,0.05
TM	0.50,0.40,0.10	0.50,0.40,0.10	0.35,0.55,0.10	0.50,0.40,0.10
SB	0.65,0.25,0.10	0.65,0.25,0.10	0.75,0.15,0.10	0.50,0.40,0.10
CED	0.65,0.25,0.10	0.75,0.15,0.10	0.85,0.10,0.05	0.50,0.40,0.10
FS	0.25,0.65,0.10	0.35,0.55,0.10	0.75,0.15,0.10	0.50,0.40,0.10

Table 13.9 The relative performances of the three pathways for bioethanol production with respect to each criterion and the relative weights using intuitionistic fuzzy numbers by DM#3

	Wheat-based	Corn-based	Cassava-based	Relative importance
LCC	0.35,0.55,0.10	0.35,0.55,0.10	0.50,0.40,0.10	0.75,0.15,0.10
CC	0.35,0.55,0.10	0.85,0.10,0.05	0.50,0.40,0.10	0.75,0.15,0.10
TA	0.35,0.55,0.10	0.75,0.15,0.10	0.50,0.40,0.10	0.75,0.15,0.10
H.Tox	0.25,0.65,0.10	0.85,0.10,0.05	0.75,0.15,0.10	0.75,0.15,0.10
PMF	0.25,0.65,0.10	0.85,0.10,0.05	0.75,0.15,0.10	0.50,0.40,0.10
TM	0.65,0.25,0.10	0.65,0.25,0.10	0.50,0.40,0.10	0.85,0.10,0.05
SB	0.65,0.25,0.10	0.65,0.25,0.10	0.85,0.10,0.05	0.85,0.10,0.05
CED	0.50,0.40,0.10	0.65,0.25,0.10	0.75,0.15,0.10	0.85,0.10,0.05
FS	0.05,0.95,0.00	0.25,0.65,0.10	0.65,0.25,0.10	0.85,0.10,0.05

medium (M) corresponding to (0.90,0.05,0.05), (0.75,0.20,0.05), and (0.50,0.40,0.10), respectively. According to Eq. (13.20), the crisp role weights of these three groups are 0.4133, 0.3444, and 0.2423, respectively. After this, the aggregated decision-making matrix can be obtained by aggregating the three weighted decision-making matrices according to Eqs. (13.21), (13.22), and the results are presented in Table 13.10.

The ideal and antiideal intuitionistic fuzzy solutions can be determined by Eqs. (13.23)–(13.26), and the results are presented in Table 13.11.

Then, the degree of the similarity of each alternative and the ideal intuitionistic fuzzy solution and that of each alternative and the antiideal

Table 13.10 The aggregated decision-making matrix

	Wheat-based	Corn-based	Cassava-based
LCC	(0.2683,0.6257,0.1061)	(0.3946,0.4910,0.1143)	(0.5765,0.3002,0.1233)
CC	(0.2975,0.5950,0.1075)	(0.7614,0.1699,0.0687)	(0.5946,0.2765,0.1288)
TA	(0.2540,0.6373,0.1088)	(0.6375,0.2350,0.1275)	(0.4048,0.4722,0.1230)
H.Tox	(0.1972,0.7033,0.0995)	(0.6388.0.2663,0.0949)	(0.5460,0.3184,0.1356)
PMF	(0.1089,0.8415,0.0496)	(0.6388,0.2663,0.0949)	(0.3852,0.4896,0.1252)
TM	(0.4586,0.4078,0.1336)	(0.4586,0.4078,0.1336)	(0.3056,0.5842,0.1102)
SB	(0.2486,0.6388,0.1126)	(0.2486,0.6388,0.1126)	(0.3174,0.5600,0.1226)
CED	(0.2767,0.6075,0.1158)	(0.3632,0.5039,0.1329)	(0.4133,0.4671,0.1196)
FS	(0.0965,0.8474,0.0561)	(0.2010,0.6944,0.1046)	(0.4916,0.3719,0.1365)

Table 13.11 The ideal and antiideal intuitionistic fuzzy solutions

	Ideal solutions	Antiideal solutions
LCC	(0.5765,0.3002,0.1233)	(0.2683,0.6275,0.1061)
CC	(0.7614,0.1699,0.0687)	(0.2975,0.5950,0.1075)
TA	(0.6375,0.2350,0.1275)	(0.2540,0.6373,0.1088)
H.Tox	(0.6388,0.2663,0.0949)	(0.1972,0.7033,0.0995)
PMF	(0.6388,0.2663,0.0949)	(0.1089,0.8415,0.0496)
TM	(0.4586,0.4078,0.1336)	(0.3056,0.5842,0.1102)
SB	(0.3174,0.5600,0.1226)	(0.2486,0.6388,0.1126)
CED	(0.4133,0.4671,0.1196)	(0.2767,0.5075,0.1158)
FS	(0.4916,0.3719,0.1365)	(0.0965,0.8474,0.0561)

Table 13.12 The ideal and antiideal intuitionistic fuzzy solutions

	Wheat-based	Corn-based	Cassava-based
Degree of the similarity to ideal solutions	0.6098	0.8714	0.8527
Degree of the similarity to antiideal solutions	0.9462	0.6633	0.7071
Relative similarity measure of each alternative	0.3919	0.5678	0.5467
Ranking	3	1	2

intuitionistic fuzzy solution can be determined by Eqs. (13.27), (13.28), and the results are presented in Table 13.12.

Finally, the relative similarity measure of each alternative bioethanol production pathway can also be determined by Eq. (13.29), and the results are also presented in Table 13.12. It is apparent that the corn-based pathway for bioethanol production is most sustainable, followed by cassava- and wheat-based pathways in the descending order.

4 Conclusions

This study aims at developing a multicriteria intuitionistic fuzzy group decision-making method for sustainability ranking of alternative bioethanol production pathways based on intuitionistic fuzzy theory which allows multiple groups of decision-makers/stakeholders to participate in the decision-making and allows the decision-makers to use intuitionistic fuzzy numbers to rate the alternative with respect to the evaluation criteria and determine the relative weights of the evaluation criteria, and the relative importance of the roles of the decision-makers/stakeholders was also recognized as different. All in all the developed method for sustainability ranking of alternative bioethanol production pathways has the following advantages:

(1) The ambiguity and hesitations existing in human's judgments can be appropriately addressed.

(2) Multiple groups of decision-makers are allowed to participate in the decision-making process.

(3) This is an object-oriented method which can determine the sustainability sequence of the alternative bioethanol production pathways according to the opinions and the preferences of the decision-makers/stakeholders.

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CHAPTER 14

An aggregated life cycle sustainability index for ranking biofuel production pathways

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1 Introduction

Biofuels (bioethanol and biodiesel) which can be synthesized via various bioprocesses can have significant contributions on promoting the development of sustainable and renewable energy in future (Peng et al., 2018). Biofuels can be produced from various different feedstocks, that is, sweet sorghum, cassava, sweet potato, biomass, sewage sludge, *Jatropha curcas* L., and *Pistacia chinensis* (Ren et al., 2015a). In addition, the development of biofuel industry can not only enhance the energy security, but also mitigate the negative environmental impacts. For instance, sewage sludge can be converted into energy via various biochemical processes (Ren et al., 2017). However, different pathways for the production of biofuels have different production costs, different environmental impacts, and different social contributions, thus, the sustainability performances of different pathways for biofuel production are different. Therefore sustainability assessment of different pathways for biofuel production is beneficial to select the most sustainable biofuel production pathway among multiple options.

Life cycle sustainability assessment (LCSA) comprised of life cycle assessment (LCA), an economic life cycle analysis (ELCA, also called "life cycle

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costing"), and social life cycle assessment (SLCA) has been recognized as a promising methodology for sustainability assessment and measurement of biofuel production pathways from life cycle perspective (Berriel et al., 2018). LCA can be used to study the environmental impacts, ELCA can be used to measure the economic performances, and SLCA can be employed to measure social influences (Ren and Toniolo, 2018; Ren et al., 2018). However, sometime it is still difficult for the decision-makers to determine the sustainability sequence of the alternative biofuel production pathways, because one biofuel production pathway may perform better on some criteria, but it may also perform worse on some other criteria, and the decision-makers/stakeholders are puzzled when facing various conflicting criteria for sustainability assessment. Accordingly, multicriteria decision analysis (MCDA) was usually combined with LCSA for sustainability prioritization of different alternatives. For instance, Ekener et al. (2018) combined LCSA and MCDA to analyze the sustainability of biofuels and fossil fuels for transportation. Ren et al. (2015b) combined LCSA and VIKOR (VlseKriterijumska Optimizacija I Kompromisno Resenje) which is a typical MCDA method for sustainability ranking of different bioethanol production pathways. Martín-Gamboa et al. (2017) combined life cycle thinking and DEA (Data Envelopment Analysis) to a sustainability-oriented MADA for ranking and benchmarking energy systems. All the methods can help the decision-makers/stakeholders to select the most sustainable biofuel production pathway. However, the decision-makers/stakeholders still do not know the life cycle sustainability performance of each biofuel production quantitatively.

To address this issue, composite sustainability index and aggregated sustainability index are introduced with considerable investigations. The investigations involve all the agricultural, construction, and industrial sectors with the research scale from local to global. von Wirén-Lehr (2001) assessed the sustainability in agriculture by aggregating seven goal-oriented concepts. Kamali et al. (2018) proposed a life cycle sustainability framework using aggregated sustainability indices for residential modular buildings. Ren (2018) developed a life cycle aggregated sustainability index method for the prioritization of industrial systems under data uncertainties. Roth et al. (2009) outlined the approach to the evaluation of sustainability of current and future electricity supply options by multicriteria decision analysis. Clerici et al., 2004assessed the sustainability at the province scale by aggregated environmental performance indicators. Tilman and Clark (2014) analyzed the sustainability linked to the environment and human health under effect of the global diets. In summary, the composite sustainability index and aggregated sustainability index are innovative approaches to aggregate multidimensional indexes into the one or some comparable indexes to evaluate sustainability (Gan et al., 2017). Computing aggregate values is a common method used for constructing indices. An index can be either simple or weighted depending on its purpose (Singh et al., 2012). Composite index needs to be designed within a coherent framework to ensure the specific parameters involved in the assessment process can change over time based on the interests of the particular stakeholders.

However, despite the plenty outcomes for the application of composite and aggregated sustainability index methods, there are two key gaps for life cycle aggregated sustainability method (Ren et al., 2015b): (1) how to address the qualitative criteria that are difficult to quantify in the social aspect and (2) how to use the multidimensional LCSA results for decision making.

Ren (2018) developed a powerful method which can quantify the life cycle aggregated sustainability of industrial systems by aggregating the data of each alternative with respect to the evaluation criteria and the weights of the criteria into a single sustainable index, and this method was employed in this chapter to determine the sustainability sequences of three pathways for bioethanol production (wheat-, corn-, and cassava-based bioethanol production pathways) by using the life cycle aggregated sustainability index method.

2 Life cycle aggregated sustainability index of three biofuel production pathways

The method developed by Ren (2018) was employed to study the life sustainability performances of three pathways for bioethanol production including wheat-, corn-, and cassava-based options in Chinese conditions (Ren et al., 2015b). The framework of multicriteria decision making for life cycle sustainability assessment is shown in Fig. 14.1. The function unit is 1 ton bioethanol. The life cycle boundary of bioethanol systems is shown in Fig. 14.2. The distance from the field to the plant for bioethanol production and that from the plant to the market are assumed to be 300 and 500 km, respectively. The life cycle sustainability performances of these three scenarios for bioethanol production are presented in Table 14.1 based on the work of Ren et al. (2015b). It is apparent that all the data in the life cycle sustainability performance matrix are crisp numbers, and they can be written in the format of interval numbers, as presented in Table 14.2.

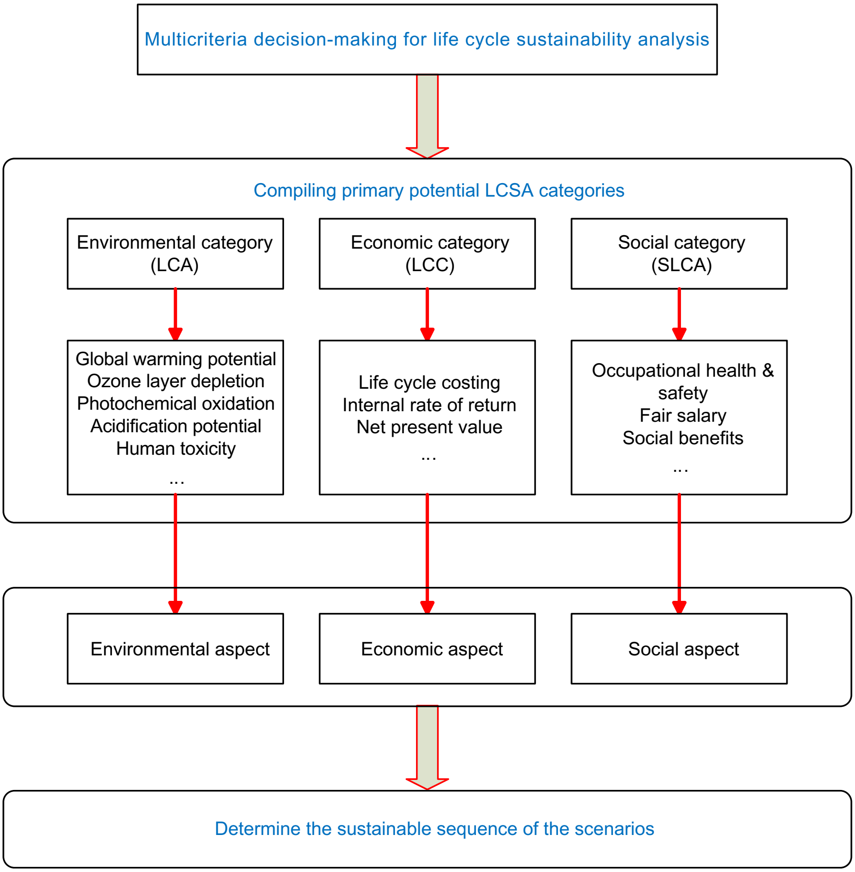

Fig. 14.1 The framework of multicriteria decision making for life cycle sustainability assessment.

After determining the interval life cycle sustainability performance matrix (see Table 14.2), there are three criteria in social pillar including SB, CED, and FS are benefit-type criteria, and the other five criteria are cost-type criteria.

As for the data with respect to the benefit-type criteria, they can be normalized. Taking the data with respect to FS as an example:

$$\left[\frac{0.25-0.25}{9.75-0.25} \quad \frac{0.25-0.25}{9.75-0.25}\right] = [0 \quad 0] \tag{14.1}$$

$$\left[\frac{1.25-0.25}{9.75-0.25} \quad \frac{1.25-0.25}{9.75-0.25}\right] = [0.1053 \quad 0.1053] \tag{14.2}$$

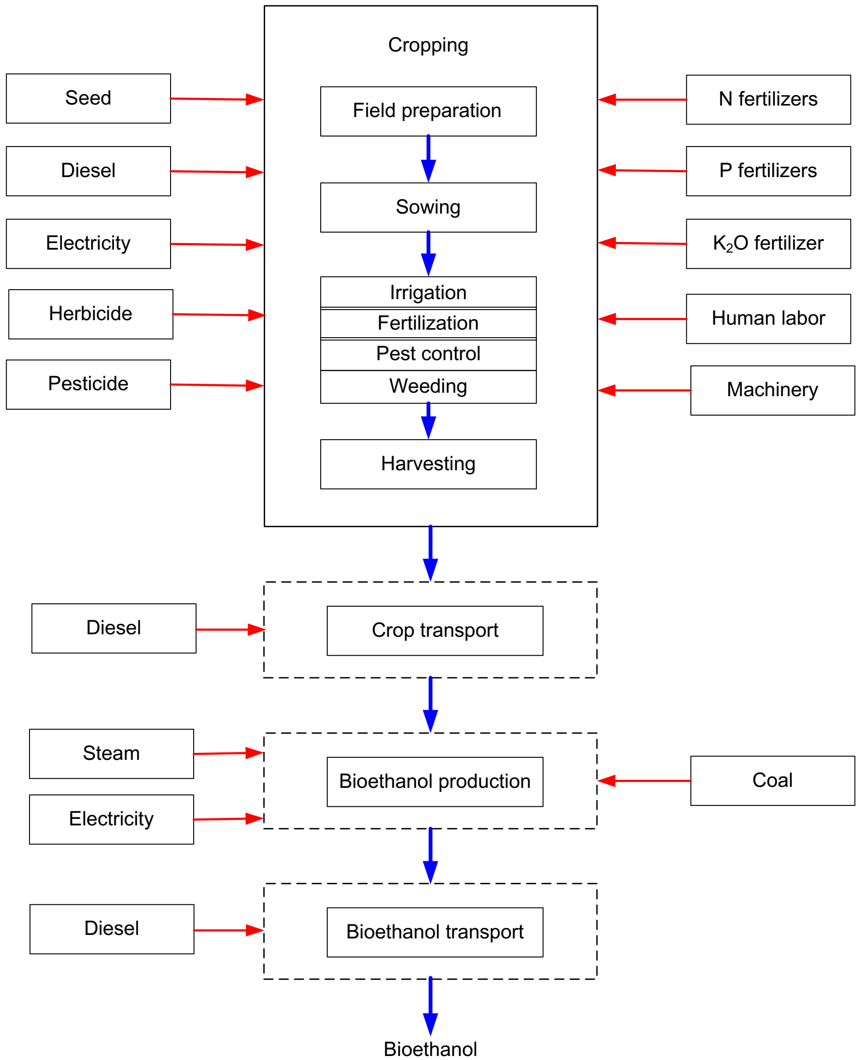

Fig. 14.2 The life cycle boundary of bioethanol systems (Ren et al., 2015b).

$$\left[\frac{9.75-0.25}{9.75-0.25} \quad \frac{9.75-0.25}{9.75-0.25}\right] = [1.0000 \quad 1.0000] \tag{14.3}$$

As for the data with respect to the cost-type criteria, they can also be normalized. Taking the data with respect to CC as an example:

$$\left[\frac{5.746-5.746}{5.746-0.461} \quad \frac{5.746-5.746}{5.746-0.461}\right] = [0 \quad 0] \tag{14.4}$$

Table 14.1 The life cycle sustainability performances of the three pathways for bioethanol production

			Wheat-based	**Corn-based**	**Cassava-based**
	Climate change (CC)	kg CO_2 eq	5.746	0.461	1.662
Environmental (EN)	Terrestrial acidification (TA)	kg SO_2 eq	2.806	0.166	0.834
	Human toxicity (H.Tox)	kg 1,4-DB eq	1.619	0.096	0.481
	Particulate matter formation (PMF)	kg PM_{10} eq	0.342	0.017	0.105
Economic (EC)	Life cycle cost (LCC)	RMB Yuan	5220	4937	4259
	Social benefits (SB)	–	8.75	8.75	9.75
Social (S)	Contribution to economic development (CED)	–	7	8.75	9.95
	Food security (FS)	–	0.25	1.25	9.75

Data from Ren, J., Manzardo, A., Mazzi, A., Zuliani, F., Scipioni, A., 2015b. 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.

Table 14.2 The life cycle sustainability performances of the three pathways for bioethanol production in the format of interval numbers

			Wheat-based	Corn-based	Cassava-based
EN	CC	kg CO_2 eq	[5.746 5.746]	[0.461 0.461]	[1.662 1.662]
	TA	kg SO_2 eq	[2.806 2.806]	[0.166 0.166]	[0.834 0.834]
	H.Tox	kg 1,4-DB eq	[1.619 1.619]	[0.096 0.096]	[0.481 0.481]
	PMF	kg PM_{10} eq	[0.342 0.342]	[0.017 0.017]	[0.105 0.105]
EC	LCC	RMB Yuan	[5220 5220]	[4937 4937]	[4259 4259]
S	SB	–	[8.75 8.75]	[8.75 8.75]	[9.75 9.75]
	CED	–	[7 7]	[8.75 8.75]	[9.95 9.95]
	FS	–	[0.25 0.25]	[1.25 1.25]	[9.75 9.75]

Table 14.3 The normalized life cycle sustainability performance matrix

		Wheat-based	Corn-based	Cassava-based
EN	CC	[0.0000 0.0000]	[1.0000 1.0000]	[0.7728 0.7728]
	TA	[0.0000 0.0000]	[1.0000 1.0000]	[0.7470 0.7470]
	H.Tox	[0.0000 0.0000]	[1.0000 1.0000]	[0.7472 0.7472]
	PMF	[0.0000 0.0000]	[1.0000 1.0000]	[0.7292 0.7292]
EC	LCC	[0.0000 0.0000]	[0.2945 0.2945]	[1.0000 1.0000]
S	SB	[0.0000 0.0000]	[0.0000 0.0000]	[1.0000 1.0000]
	CED	[0.0000 0.0000]	[0.1053 0.1053]	[1.0000 1.0000]
	FS	[0.0000 0.0000]	[0.1053 0.1053]	[1.0000 1.0000]

$$\left[\frac{5.746-0.461}{5.746-0.461} \quad \frac{5.746-0.461}{5.746-0.461}\right]=[1.0000 \quad 1.0000] \tag{14.5}$$

$$\left[\frac{5.746-1.662}{5.746-0.461} \quad \frac{5.746-1.662}{5.746-0.461}\right]=[0.7728 \quad 0.7728] \tag{14.6}$$

In a similar way, all the data can be normalized, and the normalized life cycle sustainability performance matrix can be determined, as presented in Table 14.3.

The interval preference relation-based goal programming model (Zhang, 2016) was first employed to determine the weights of the three pillars of sustainability (environmental, economic, and social aspects) and the local weights of the criteria in each dimension. In order to investigate the relative weights of the three pillars of sustainability and that of the criteria in each dimension, a focus group meeting in which two professors whose research focused on sustainability engineering and renewable energy, two senior chemical engineers, two environmentalists, two senior managers from renewable energy companies, and two residents who live near the plants for bioenergy production were invited to participate in was held in Chongqing, China on 7th August 2017. The participants discussed for establishing the comparison matrices for determining the weights of three pillars of sustainability and the local weights of the criteria in each pillar. Taking the weights of the three pillars as an example:

Step 1: Determining the interval comparison matrix of the three pillars by using the multiplicative preference relation. The interval comparison matrix is presented in Eq. (14.7).

$$\begin{array}{cccc} & \text{Environmental} & \text{Economic} & \text{Social} \\ \text{Environmental} & [1\ \ 1] & [2\ \ 4] & [5\ \ 7] \\ \text{Economic} & \left[\frac{1}{4}\ \ \frac{1}{2}\right] & [1\ \ 1] & [2\ \ 3] \\ \text{Social} & \left[\frac{1}{7}\ \ \frac{1}{5}\right] & \left[\frac{1}{3}\ \ \frac{1}{2}\right] & [1\ \ 1] \end{array} \tag{14.7}$$

Step 2: Transforming the multiplicative preference relation presented in Eq. (14.7) into fuzzy preference relation. For instance, the relative preference of environmental aspect to economic is $[2\ \ 4]$, and this multiplicative preference relation can be transformed into

$$\left[\frac{2}{1+2}\ \ \frac{4}{1+4}\right] = \left[\frac{2}{3}\ \ \frac{4}{5}\right] \tag{14.8}$$

In a similar way, all the elements in the matrix presented in Eq. (14.7) can be transformed into fuzzy preference relations, and the fuzzy preference relation matrix is presented in Eq. (14.9).

$$\begin{array}{cccc} & \text{Environmental} & \text{Economic} & \text{Social} \\ \text{Environmental} & \left[\frac{1}{2}\ \ \frac{1}{2}\right] & \left[\frac{2}{3}\ \ \frac{4}{5}\right] & \left[\frac{5}{6}\ \ \frac{7}{8}\right] \\ \text{Economic} & \left[\frac{1}{5}\ \ \frac{1}{3}\right] & \left[\frac{1}{2}\ \ \frac{1}{2}\right] & \left[\frac{2}{3}\ \ \frac{3}{4}\right] \\ \text{Social} & \left[\frac{1}{8}\ \ \frac{1}{6}\right] & \left[\frac{1}{4}\ \ \frac{1}{3}\right] & \left[\frac{1}{2}\ \ \frac{1}{2}\right] \end{array} \tag{14.9}$$

Step 3: Establishing the goal programming for determining the interval weights of the three pillars of sustainability. The goal programming (14.10) is established to determine the weights of environmental, economic, and social pillars:

$$
\begin{aligned}
&\text{Min} = c_{\mathrm{EN}}^{-} + c_{\mathrm{EN}}^{+} + d_{\mathrm{EN}}^{-} + d_{\mathrm{EN}}^{+} + c_{\mathrm{EC}}^{-} + c_{\mathrm{EC}}^{+} + d_{\mathrm{EC}}^{-} + d_{\mathrm{EC}}^{+} + c_{\mathrm{S}}^{-} + c_{\mathrm{S}}^{+} + d_{\mathrm{S}}^{-} + d_{\mathrm{S}}^{+}\\
&\text{s.t.}\\
&\omega_{\mathrm{EN}}^{-} + \omega_{\mathrm{EC}}^{+} + \omega_{\mathrm{S}}^{+} \geq 1\\
&\omega_{\mathrm{EC}}^{-} + \omega_{\mathrm{EN}}^{+} + \omega_{\mathrm{S}}^{+} \geq 1\\
&\omega_{\mathrm{S}}^{-} + \omega_{\mathrm{EN}}^{+} + \omega_{\mathrm{EC}}^{+} \geq 1\\
&\omega_{\mathrm{EN}}^{+} + \omega_{\mathrm{EC}}^{-} + \omega_{\mathrm{S}}^{-} \leq 1\\
&\omega_{\mathrm{EC}}^{+} + \omega_{\mathrm{EN}}^{-} + \omega_{\mathrm{S}}^{-} \leq 1\\
&\omega_{\mathrm{S}}^{+} + \omega_{\mathrm{EN}}^{-} + \omega_{\mathrm{EC}}^{-} \leq 1\\
&0 \leq \omega_{\mathrm{EN}}^{-} \leq \omega_{\mathrm{EN}}^{+} \leq 1\\
&0 \leq \omega_{\mathrm{EC}}^{-} \leq \omega_{\mathrm{EC}}^{+} \leq 1\\
&0 \leq \omega_{\mathrm{S}}^{-} \leq \omega_{\mathrm{S}}^{+} \leq 1\\
&(1/2 + 2/3 + 5/6 - 3 + 0.5)\omega_{\mathrm{EN}}^{-} + (2/3)\omega_{\mathrm{EC}}^{+} + (5/6)\omega_{\mathrm{S}}^{+} - c_{\mathrm{EN}}^{-} + c_{\mathrm{EN}}^{+} = 0\\
&(1/5 + 1/2 + 2/3 - 3 + 0.5)\omega_{\mathrm{EC}}^{-} + (1/5)\omega_{\mathrm{EN}}^{+} + (2/3)\omega_{\mathrm{S}}^{+} - c_{\mathrm{EC}}^{-} + c_{\mathrm{EC}}^{+} = 0\\
&(1/8 + 1/4 + 1/2 - 3 + 0.5)\omega_{\mathrm{S}}^{-} + (1/8)\omega_{\mathrm{EN}}^{+} + (1/4)\omega_{\mathrm{EC}}^{+} - c_{\mathrm{S}}^{-} + c_{\mathrm{S}}^{+} = 0\\
&(1/2 + 4/5 + 7/8 - 3 + 0.5)\omega_{\mathrm{EN}}^{+} + (4/5)\omega_{\mathrm{EC}}^{-} + (7/8)\omega_{\mathrm{S}}^{-} - d_{\mathrm{EN}}^{-} + d_{\mathrm{EN}}^{+} = 0\\
&(1/3 + 1/2 + 3/4 - 3 + 0.5)\omega_{\mathrm{EC}}^{+} + (1/3)\omega_{\mathrm{EN}}^{-} + (3/4)\omega_{\mathrm{S}}^{-} - d_{\mathrm{EC}}^{-} + d_{\mathrm{EC}}^{+} = 0\\
&(1/6 + 1/3 + 1/2 - 3 + 0.5)\omega_{\mathrm{S}}^{+} + (1/6)\omega_{\mathrm{EN}}^{-} + (1/3)\omega_{\mathrm{EC}}^{-} - d_{\mathrm{S}}^{-} + d_{\mathrm{S}}^{+} = 0
\end{aligned}
\tag{14.10}
$$

After solving programming (Eq. 14.10) by Lingo 11.0, the weights of environmental, economic, and social pillars can be determined, and they are $[0.5860 \quad 0.7329]$, $[0.1653 \quad 0.3122]$, and $[0.0947 \quad 0.1018]$, respectively.

In a similar way, the local weights of the criteria in each dimension can also be determined, and the results are summarized in Tables 14.4–14.5.. It is worth pointing out that there is only one criterion in economic aspect, and its local weight is 1. Then, the global weights of the eight criteria can be determined, and the results are presented in Table 14.6.

After determining the normalized life cycle sustainability performance matrix and the global weights of the eight criteria, the weighted normalized life cycle sustainability performance matrix can be determined, and the results are presented in Table 14.7. The ideal solutions can also be determined (see Table 14.7).

The projections of these three pathways for bioethanol production can be determined and the results are presented in Fig. 14.3.

The probability matrix by comparing the projections of each pair of bioethanol production systems can be determined. Taking the probability of the projection of corn-based bioethanol production system on the ideal

Table 14.4 The comparison matrix for determining the weights of the four criteria in environmental aspect

	CC	TA	H.Tox	PMF
Multiplicative preference relation				
CC	[1 1]	[3 5]	[2 3]	[3 4]
TA	[1/5 1/3]	[1 1]	[1/3 1]	[1/2 1]
H.Tox	[1/3 1/2]	[1 3]	[1 1]	[1 2]
PMF	[1/4 1/3]	[1 2]	[1/2 1]	[1 1]
Fuzzy preference relation				
CC	[1/2 1/2]	[3/4 5/6]	[2/3 3/4]	[3/4 4/5]
TA	[1/6 1/4]	[1/2 1/2]	[1/4 1/2]	[1/3 1/2]
H.Tox	[1/4 1/3]	[1/2 3/4]	[1/2 1/2]	[1/2 2/3]
PMF	[1/5 1/4]	[1/2 2/3]	[1/3 1/2]	[1/2 1/2]
Weights	[0.2100 0.4089]	[0 0.1989]	[0.1627 0.3616]	[0.0306 0.2295]

Table 14.5 The comparison matrix for determining the weights of the four criteria in environmental aspect

	SB	CED	FS
Multiplicative preference relation			
SB	[1 1]	[1/3 1/2]	[1/6 1/4]
CED	[2 3]	[1 1]	[1/4 1/2]
FS	[4 6]	[2 4]	[1 1]
Fuzzy preference relation			
SB	[1/2 1/2]	[1/4 1/3]	[1/7 1/5]
CED	[2/3 3/4]	[1/2 1/2]	[1/5 1/3]
FS	[4/5 6/7]	[2/3 4/5]	[1/2 1/2]
Weights	[0.1084 0.1232]	[0.1953 0.2988]	[0.5780 0.6963]

Table 14.6 The global weights of the eight criteria

Pillars	Criteria	Local weights	Global weights
EN [0.5860 0.7329]	CC	[0.2100 0.4089]	[0.1231 0.2997]
	TA	[0 0.1989]	[0 0.1458]
	H.Tox	[0.1627 0.3616]	[0.0953 0.2650]
	PMF	[0.0306 0.2295]	[0.0179 0.1682]
EC [0.1653 0.3122]	LCC	[1.0000 1.0000]	[0.1653 0.3122]
S [0.0947 0.1018]	SB	[0.1084 0.1232]	[0.0103 0.0125]
	CED	[0.1953 0.2988]	[0.0185 0.0304]
	FS	[0.5780 0.6963]	[0.0547 0.0709]

Table 14.7 The weighted normalized life cycle sustainability performance matrix

		Wheat-based	Corn-based	Cassava-based	Ideal solutions
EN	CC	[0.0000 0.0000]	[0.1231 0.2997]	[0.0951 0.2316]	0.2997
	TA	[0.0000 0.0000]	[0 0.1458]	[0 0.1089]	0.1458
	H.Tox	[0.0000 0.0000]	[0.0953 0.2650]	[0.0712 0.1980]	0.2650
	PMF	[0.0000 0.0000]	[0.0179 0.1682]	[0.0131 0.1227]	0.1682
EC	LCC	[0.0000 0.0000]	[0.0487 0.0919]	[0.1653 0.3122]	0.3122
	SB	[0.0000 0.0000]	[0.0000 0.0000]	[0.0103 0.0125]	0.0125
S	CED	[0.0000 0.0000]	[0.0110 0.0180]	[0.0185 0.0304]	0.0304
	FS	[0.0000 0.0000]	[0.0058 0.0075]	[0.0547 0.0709]	0.0709

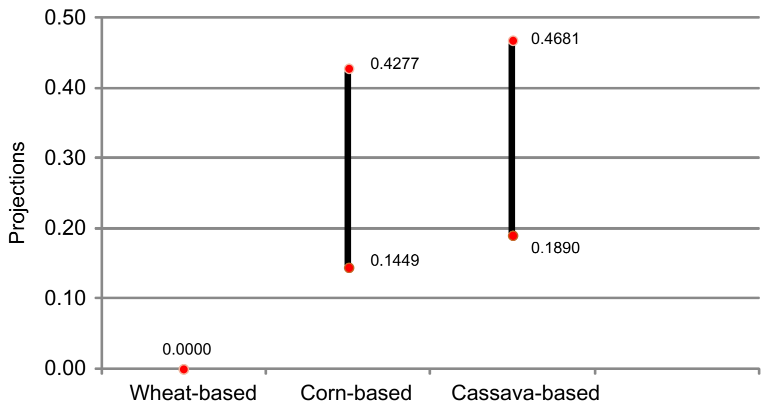

Fig. 14.3 The projections of the three bioethanol production systems on the ideal solution.

solutions be greater than that of cassava-based bioethanol production system as an example:

$$\max\left\{1-\frac{1}{2}\max\left(\frac{(0.4277+0.1449)-(0.4681+0.1890)}{0.4277-0.1449+0.4681-0.1890}+1,0\right),0\right\}$$
$$=0.4249 \tag{14.11}$$

In a similar way, all the elements in the probability matrix can be determined and the results are presented in Eq. (14.12).

$$\begin{array}{cccc} & \text{Wheat} & \text{Corn} & \text{Cassava} \\ \text{Wheat} & 0.5000 & 0 & 0 \\ \text{Corn} & 1.0000 & 0.5000 & 0.4249 \\ \text{Cassava} & 1.0000 & 0.5751 & 0.5000 \end{array} \tag{14.12}$$

After that, the sustainability indices of the three pathways for bioethanol production can be determined and they are as follows:

$$V_1 = \frac{(0.5000 + 0 + 0) + \frac{3}{2} - 1}{3(3-1)} = 0.1667 \tag{14.13}$$

$$V_2 = \frac{(1.0000 + 0.5000 + 0.4249) + \frac{3}{2} - 1}{3(3-1)} = 0.4042 \tag{14.14}$$

$$V_3 = \frac{(1.0000 + 0.5751 + 0.5000) + \frac{3}{2} - 1}{3(3-1)} = 0.4292 \tag{14.15}$$

Therefore the sustainability indices of the three pathways for bioethanol production are 0.1667, 0.4042, and 0.4292. Accordingly, the cassava-based bioethanol production pathway with the sustainability index 0.4292 is the most sustainable, followed by corn-based and wheat-based with the sustainability indices 0.4042 and 0.1667, respectively. The sustainability order of the three bioethanol production scenarios determined by the developed life cycle aggregated sustainability index is consistent with the results determined by the combination of AHP and VIKOR in the work of Ren et al. (2015b). In addition, the life cycle aggregated sustainability indices with respect to cassava-based and corn-based bioethanol production systems are much greater than that of wheat-based bioethanol production systems, and this conclusion was also consistent to the results in the work of Ren et al. (2015b), and they argued that the recognition of cassava-based bioethanol production system as the most sustainable is robust, and corn-based bioethanol production system could be the most sustainable in some cases when changing the weights of the criteria for life cycle sustainability assessment of bioethanol production systems.

3 Conclusions

This chapter shows the feasibility of using life cycle aggregated sustainability index method developed by Ren (2018) to prioritize the alternative biofuel production pathways by aggregating all the criteria into a single sustainability index, three biofuel production pathways including wheat-, corn-, and cassava-based pathways for bioethanol production were studied to illustrate how to determine the life cycle aggregated sustainability index, and the results are consistent to that determined by the combination of LCSA and MCDA method. To some extent, it reveals that the life cycle aggregated sustainability index method is feasible for prioritizing alternative biofuel production pathways.

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Biofuels for a More Sustainable Future

Life Cycle Sustainability Assessment and Multi-Criteria Decision Making

Edited by **Jingzheng Ren, Antonio Scipioni, Alessandro Manzardo and Hanwei Liang**

Provides the life cycle sustainability analysis, multi-criteria decision-making tools, industry insights, and policy implications for promoting the sustainable development of biofuels

Biofuels for a More Sustainable Future: Life Cycle Sustainability Assessment and Multi-Criteria Decision-Making provides a comprehensive sustainability analysis of biofuels based on life cycle thinking and develops various multi-dimensional decision-making techniques for prioritizing the biofuel production technologies. Taking a transversal approach, the book combines life cycle sustainability assessment, life cycle assessment, life cycle costing analysis, social life cycle assessment, sustainability metrics, triple bottom line, operations research methods, and supply chain design for investigating the critical factors and key enablers that influence the sustainable development of biofuel industry; summarizing the biofuels technologies, processes, standards and regulations; employing the life cycle thinking for investigating the sustainability of biofuels; defining the broad concept of sustainability and sustainability assessment; developing the decision-support tools for sustainability prioritization of biofuel production pathways; and reviewing the key issues, challenges, and models for sustainable biofuel supply chain design.

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