Green Energy and Technology

Riccardo Basosi · Maurizio Cellura
Sonia Longo · Maria Laura Parisi *Editors*

Life Cycle Assessment of Energy Systems and Sustainable Energy Technologies

The Italian Experience

Springer

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Preface

Energy is one of the key elements included in the 17 Sustainable Development Goals (SDGs) of the United Nations' 2030 Agenda for Sustainable Development. Universal access to energy, a higher share of renewable energy and massive improvements in energy efficiency are part of the top global priorities for sustainable development in the years to come. In detail, ensuring the access to affordable, reliable, sustainable, and modern energy is the main objective of SDG 7. Strong interactions among energy and most of the topics of the other SDGs can be found that will be briefly discussed below. Enabling poor communities to use local clean and renewable energy resources relies on SDG 1 "No poverty" and SDG 10 "Reduced inequalities". In addition, the reduction of energy-related resource and water depletion, greenhouse gas emissions, air–water–soil pollution can contribute to the achievement of SDG 3 "Good health and well-being", SDG 6 "Clean water and sanitation", SDG 12 "Responsible production and consumption", and SDG 13 "Climate action". Furthermore, renewable energy and energy efficiency can support the SDG 2 "Zero hunger" and SDG 11 "Sustainable cities and communities".

Thus, energy is an enabling and strategic factor toward sustainable development and for the transition toward a low-carbon economy.

However, taking sustainability-related decisions in the energy field would require science-based approaches focusing on the whole energy system, including energy generation, distribution, use, and end-of-life.

The Life Cycle Assessment (LCA) methodology is pivotal for this purpose. It is useful for assessing energy-related resource depletion and environmental impacts, for avoiding burden transfer from one life-cycle step to another and from an impact category to the others. Furthermore, it helps in supporting the identification of priority actions in policy making, the selection of the best low-carbon solutions for energy supply and use, the identification of the hot-spots for reducing the carbon intensity of energy systems and the management of the end-of-life of energy systems.

In this context, the aims of this book are to share some Italian experiences on LCA applied to the energy systems and technologies and provide an overview of the most recent outcomes of the research in the field of energy, with a specific focus on renewables, bio-energy, and sustainable solutions. The book consists of two parts: the first one describes case studies and review studies of LCA applied to different energy sources and energy systems (geothermal, photovoltaic, biomass, electricity production, energy-related systems as batteries and smart grids). The second part of the book focuses on LCA applied to bio-energies and bio-energy systems.

The book is the outcome of the activity of several members of the working group “Sustainable energy and technologies” of the “Italian LCA Network” association.

We would like to acknowledge all Authors for submitting their valuable work and the members of the “Italian LCA Network” association for supporting this work. We highlight that the views expressed in this book are those of the Authors and do not necessarily represent the views of the Italian LCA network association.

We hope that LCA practitioners, researchers, and students will consider this book as an opportunity to learn more the applications of the LCA methodology and to understand the environmental impact of energy systems and sustainable energy technologies, through the analysis of their life cycles.

The Italian LCA Network and the Working Group “Energy and Sustainable Technologies”

The Italian LCA Network was created in 2006 with the aim to have a network for exchanging information, methodologies, and good practices on LCA in Italy. In 2012, the Italian LCA Network became an association, founded by the Italian National Agency for New Technologies, Energy and Sustainable Economic Development (ENEA); the Politecnico of Milano; the Universities of Bari, Chieti-Pescara, Padova and Palermo; and the National Interuniversity Consortium for Chemical Reactivity and Catalysis (CIRCC).

The main objectives of the “Italian LCA Network” association are the following: promoting the adoption of the life cycle thinking approach for achieving a sustainable development; promoting the dissemination of the LCA methodology at national level and the exchange of information and best practices on the LCA in Italy; encouraging networking processes among different stakeholders for the realization of national and international projects.

The working group “Energy and sustainable technologies” of the “Italian LCA Network” aims at assessing the energy and environmental performances of energy generation, transformation and use systems; at promoting the eco-efficiency on any level in the energy sector, at following the approach from resource to waste; at analyzing the state of the art of LCA studies on energy and sustainable technologies; at exchanging experiences regarding LCA applied to energy and sustainable technologies.

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LCA Applied to the Energy Sector: State of the Art and Case Studies

Chapter 1
Life Cycle Assessment of Electricity Generation Scenarios in Italy

Maurizio Cellura, Maria Anna Cusenza, Francesco Guarino, Sonia Longo and Marina Mistretta

Abstract Hindering global warming and achieving a more competitive, secure and sustainable energy sector are some of the most relevant goals of the 2030 Framework for climate and energy of the European Union. European countries have to identify and implement strategies for contributing to these ambitious goals. In this context, the authors carried out a scenario analysis on the Sicilian electricity mix in order to estimate the life cycle energy and environmental benefits of the increase of the use of renewable energy technologies for electricity production, and the potential contribution of Sicily in the achievement of the European energy and environmental targets. In detail, the authors identified two electricity generation scenarios for 2030 starting from the Sicilian electricity mix in 2014, performed assumptions on the forecasted electricity demand and assessed the potential of renewable energy sources exploitation and the technical, political, social, and environmental constraints. Then, they applied the Life Cycle Assessment methodology to assess the eco-profiles of the identified electricity generation mixes and compared them with the eco-profile of electricity produced in 2014. The results of the comparison showed a reduction of most of the 16 examined environmental impact categories, except for those related to human toxicity, particulate matter, ionizing radiation and resource depletion.

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1.1 Introduction

Greenhouse gases (GHG) from human activities are the most significant drivers of observed climate change since the mid-twentieth century (US—EPA 2017). From 1970 to 2012, the GHG emissions increased steadily from 24.3 to 46.4 Gton CO_{2eq}/year (Janssens-Maenhout et al. 2017).

In the past years, the awareness of the impacts of human activities on climate has led to the definition of various environmental policies aimed at reducing GHG emissions (e.g. Kyoto Protocol, Paris climate conference—COP21, etc.).

Among the human activities responsible for GHG, the energy sector (including power generation and energy consuming sectors, i.e. buildings, industry, transport and agriculture) represents by far the largest source of emissions. In detail, it accounts for two-thirds of global GHG and 80% of CO_2 emissions (IEA 2014a, 2016, 2017). Therefore, effective actions in the energy sector are essential to tackling the climate change problem (Beccali et al. 2007). Mitigation scenario studies carried out by IPCC indicate that, within the energy system, the electricity sector can play an important role in deep GHG emissions cut, as the decarbonization of electricity generation can be achieved at a much higher pace than in the rest of the energy system (IPCC 2014). A variety of mitigation options exists in the electricity sector, including renewable, nuclear power plants or fossil fuel power stations equipped with carbon capture and storage (CCS) technologies (Bruckner et al. 2014). Many climate change mitigation policies are focused on replacing fossil fuel with renewable energy sources (RES) (Dandres et al. 2012).

In this framework, the European Union (EU) has set ambitious targets for 2030 in the field of GHG emissions and renewable energy generation (European Commission 2011). In detail, the 2030 EU energy and climate objectives aim at cutting GHG emissions by 40% if compared to 1990 levels, and at increasing the share of renewables to at least 27% of EU energy consumption (European Commission 2014). In order to match such objectives, all Member States should contribute to the attainment of these common objectives and targets to different extents (European Commission 2016). In this context, the authors focused their attention on the electricity sector, and carried out a scenario analysis for its generation in Sicily in 2030, considering a high exploitation of RES. The authors followed a life cycle approach in order to assess the potential contribution of Sicily in the achievement of European energy and climate targets. Moreover, as replacing fossil fuels with RES could cause negative impacts, e.g., in terms of resources depletion, they carried out an environmental evaluation of the scenarios, including the assessments of a wide range of environmental impacts.

1.2 Scenario Analysis

In the following steps, the methodology employed in the definition of the electricity generation scenarios in 2030 is briefly described:

- Step 1—Electricity production in Sicily: analysis of electricity production in Sicily from 2009 to 2014 in order to characterize the electricity mix and the RES penetration in the system.
- Step 2—Identification of the renewable energy technologies that can change their capacity production (increase or decrease) in response to a change in the demand for electricity (renewable energy marginal technologies).
- Step 3—Electricity generation scenarios: definition of two electricity scenarios considering a decrease (−0.2%/year) of the electricity demand by 2030 in the first one and an increase (+0.6%/year) in the second one. Both forecasted scenarios are characterized by an increase in electricity produced by RES.

1.2.1 Electricity Production in Sicily

In Sicily, electricity is generated by thermal power plants, hydroelectric plants, wind turbines, and photovoltaic (PV) systems (Region of Sicily 2015). Figure 1.1 shows the evolutions of the Sicilian electricity mix per type of plant and energy source from 2009 to 2014 (TERNA 2017; GSE 2017).

2014 was assumed as reference scenario (RS14) due to the highest availability of data: the annual total electricity production was equal to 22,536 GWh: 75.3% was generated by fossil fuel thermal power plants and 24.7% by RES (TERNA 2017). Concerning RES only 2.1% was generated by hydroelectric plants (1.4% hydropower at reservoir; 0.6% hydropower run of river), 8.4% by PV systems, 13.0% by wind turbines, and 1.2% by bioenergy.

The amount of electricity generated by RES in 2014 was assumed as a reference for the estimation of RES exploitation in the 2030—scenarios, as described in Sect. 1.2.3.

1.2.2 Identification of the Renewable Energy Marginal Technologies

In order to contribute to the EU energy and climate goals, the future electricity production sector should be characterized by an increase of the RES. Then, the potential capacity production of each renewable energy technology should be identified. A technology that can change its capacity production in response to a change in demand in an energy system (increase or decrease) is defined as a marginal technology (Weidema et al. 1999). It is an unconstrained technology, i.e. its capacity can be adjusted in response to a change in demand without being subjected to natural

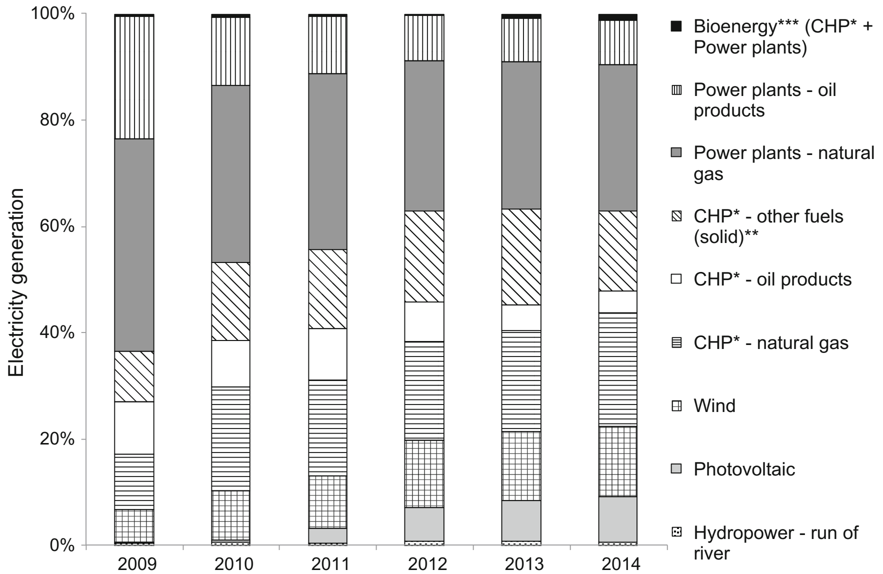

Fig. 1.1 Share of electricity production per type of plant and of energy source in Sicily from 2009 to 2014 (*CHP: combined heat and power plant; **other fuel (solid): coke, brown coal briquettes, etc.; ***bioenergy: biogas, bioliquid and biomass) (own elaboration from TERNA and GSE data) [the production of electricity in pumped storage units from water previously pumped uphill is not included, as indicated in (European Commission 2009)]

capacity constraints (e.g. the amount of water available in a specific region), political constraints (e.g. emission limits), etc. (Weidema et al. 1999).

The authors identified the renewable energy marginal technologies for the Sicilian electricity sector considering the following factors:

1. RES penetration in the Sicilian electricity mix in 2014;
2. the technical potential[1] for the exploitation of RES in Sicily;
3. the European energy policies and the Sicilian energy strategies (Region of Sicily 2008, 2014, 2015).

As forecasts for 2030 were not available, the technical potential for RES exploitation in 2030 was assessed starting from the estimations of RES development in 2020, reported in national and regional studies (Benini et al. 2010; Alterach et al. 2011; Region of Sicily 2008, 2014, 2015).

In detail, considering that the RES exploitation in Sicily in 2016 (TERNA 2017) was still far from the forecasted potentials to be installed within 2020, the authors hypothesized that these potentials for 2020 will be installed in the medium—long period (from now to 2030).

[1]The achievable energy generation of a particular technology given the system performance, topographic limitations, environmental and land-use constraints.

The only exception was represented by the wind-based systems for which estimations for 2030 were available (ANEV—Associazione Nazionale Energia del Vento 2017).

In the following, the procedure for the identification of the renewable energy marginal technologies is described.

Hydropower

The main constraints of hydropower are low social acceptance, high initial investment costs and the need to consider other water-using sectors (e.g. irrigation in agriculture, domestic uses, other industrial uses, etc.) when planning the hydropower development (IEA 2011). In the Sicilian "Energy Master Plan" (Region of Sicily 2008), which is the official document describing the current energy sector and the future energy plans in Sicily, no significant hydropower use increase is taken into account as almost all the available potential is considered exploited and the potential of small hydro is very limited. Then, hydropower technologies cannot be considered marginal in the Sicilian energy system and the future increase of electricity generation from these plants, due to small hydro (Region of Sicily 2008; Cattini et al. 2011) can be assumed as negligible if compared to RS14.

Wind

Barriers to the diffusion of wind power generation include capital costs, uncertainty regarding policy support, impacts of its intermittent generation on the power systems, and low social acceptance due to the visual impact (IEA 2008).

An installed power equal to 2000 MW is forecasted for Sicily in 2030 (+14% compared to RS14) (ANEV—Associazione Nazionale Energia del Vento 2017). This estimation is done excluding the areas subject to environmental and technical constraints (e.g. protection of flora and soil orography). Then, considering this potential and the current installed power, wind plants can be considered as a marginal technology for the Sicilian energy system. Starting from the forecasted installable power (2000 MW) and considering an average wind plant productivity equal to 2000 MWh/MW (RSE 2017), a generation of 4000 GWh is estimated for 2030.

Solar

The penetration in the energy system of this technology mainly depends on its cost (and related incentives). Photovoltaics can be considered a marginal technology, as within the renewable technologies, solar power seems to be the most promising, considering the high solar irradiation in Sicily (Šúri et al. 2007; Huld et al. 2012) and the untapped potential (Region of Sicily 2008, 2014).

In detail, the overall installed solar power in Sicily was 1295 MW in RS14 while a PV installable capacity equal to 1812 MW is forecasted in 2020 (Region of Sicily 2014). Considering that the installed power in 2016 was equal to 1344 MW (+2.7% compared to 2015) (TERNA 2017), it is supposed that the estimated potential for 2020 could be installed by 2030. Starting from the forecasted installable power (1812 MW) and considering an average PV power plant productivity equal to 1500 MWh/MW (EC—JRC 2017), a generation of 2718 GWh is estimated for 2030.

Bioenergy

The potential electricity generation from thermal plants fuelled with bioenergy in Sicily in 2020 is estimated as 795 GWh (Benini et al. 2010), considering the limitations due to other potential uses of bioenergy, e.g. for agricultural applications. As in 2016, the electricity generation by bioenergy was 239.9 GWh (TERNA 2017), it is assumed that the estimated production of 795 GWh will be reached in a medium—-long period (from now to 2030). Thus, thermal plants fuelled with bioenergy can be considered as a marginal technology for the Sicilian energy sector.

1.2.3 Electricity Generation Scenarios Definition

The definition of two possible scenarios for electricity demand in 2030 is based on the prediction of future electricity demand (TERNA 2015). The first scenario, named as "Base scenario" (BS30), estimates a decrease of −0.2%/year of the electricity demand for the Italian Islands (Sicily and Sardinia) (TERNA 2015), which results in a value of 21,512 GWh in 2030 (−3.2% if compared to RS14). The second one, named as "Development scenario" (DS30), estimates an increase of +0.6%/year of the electricity demand (TERNA 2015) which results in a demand of 24,443 GWh in 2030 (+10% if compared to RS14).

The exploitation of RES in both the forecasted scenarios is assumed to be equal to the technical potential discussed in Sect. 1.2.2. The main assumptions on the renewable marginal technologies of the BS30 and DS30 scenarios are summarized in Table 1.1.

The assessment of the electricity generation from thermoelectric plants fuelled with fossil fuels in the future scenarios is based on the difference between the renewable energy production and the forecasted energy demand in the same year. It is 14,062 GWh in BS30 and 16,993 GWh in the DS30 scenario. The percentage distribution of each technology in the thermoelectric sector is considered unchanged if compared to RS14. Both scenarios are in compliance with the European energy strategies as they reduce the fossil fuels dependence by increasing RES penetration.

Table 1.1 Main assumptions on the renewable marginal technologies in the forecasted scenarios

Technology	Marginal technology	Production RS14 (GWh)	Potential increase compared to RS14	Potential production BS30 and DS30 (GWh)
Hydropower	No	146	–	146
PV	Yes	1893	+44%	2718
Wind	Yes	2922	+37%	4000
Bioenergy	Yes	259	+207%	795

Table 1.2 Percentage composition of the generation of 1 kWh of electricity per type of plant and of energy source in RS14, BS30 and DS30 scenario (%)

Type of plant	RS14	BS30	DS30
Hydropower—run of river	0.7	0.7	0.6
Photovoltaic	8.5	12.6	11.1
Wind	13.2	18.6	16.4
CHP—natural gas	21.5	18.1	19.3
CHP—oil products	4.1	3.4	3.7
CHP—other fuels (sol.)	14.9	12.6	13.4
Power plants—natural gas	27.5	23.2	24.6
Power plants—oil products	8.4	7.1	7.6
Bioenergy (CHP+ power plants)	1.2	3.7	3.3

Starting from the electricity mix in RS14 and in the two scenarios, the percentage composition of the generation of 1 kWh of electricity per type of plant and energy source was identified (Table 1.2).

1.3 Life Cycle Assessment

1.3.1 Goal and Scope Definition

An LCA approach has been applied to assess the potential energy and environmental benefits/impacts of the forecasted electricity mixes with respect to RS14. The assessment was carried out in compliance with ISO 14040 (ISO 2006a) and ISO 14044 (ISO 2006b). The production of 1 kWh of gross electricity was selected as a functional unit (FU). The system boundaries included the extraction and transport of raw materials and fuels, the plant construction and operation. For renewable energy systems also the end-of-life step was taken into account. For thermal power plants, the end-of-life was not considered due to a lack of reliable secondary data.

The Cumulative Energy Demand (CED) method was used to assess the primary energy consumption (Frischknecht et al. 2007a), while the impact assessment was performed by means of the ILCD 2011 Midpoint method (EC—JRC 2012).

1.3.2 Life Cycle Inventory

Data collection, data quality and assumptions

The data collection process and the assessment of the percentage composition of 1 kWh of electricity per type of plant and energy source are described in Sect. 2.2.

The eco-profiles of electricity generation by each power plant and energy source were taken from Ecoinvent (Frischknecht et al. 2007b) both for 2014 and 2030. The authors assessed the environmental impacts of the future scenarios by using the eco-profile referred to the current technology development for the following reasons:

- The eco-profile changes of the electricity generated by thermoelectric plants (powered by both fossil fuels and bioenergy) occurring over the next decades were assumed negligible due to the use of mature technologies (Stamford and Azapagic 2014) and due to their lifetime (up to 50 years) (IEA 2014b);
- The eco-profiles of wind and PV power plants are expected to improve with respect to the current ones thanks to the employment of more efficient materials and technologies. This will entail a higher electricity output per unit of input (IEA 2008) and, consequently, the reduced impact per kWh of electricity generated. However, due to the uncertainty on the technological development and on the life cycle data for future technologies (Stamford and Azapagic 2014), the authors, assuming a cautious scenario, considered negligible the future wind and PV plants eco-profile improvement.

1.3.3 Life Cycle Impact Assessment and Discussion of Results

No considerable variations are found in both the scenarios if compared with RS14.

In detail, CED per kWh of electricity produced is 8.9 $MJ_{primary}$ in the BS30 scenario and 9.1 $MJ_{primary}$ in the DS30 scenario. The percentage variations compared to RS14 are equal to −5.8% in BS30 and −3.3% in DS30 (Fig. 1.2). Such a decrease is essentially due to the reduced share of the electricity produced from non-renewable technologies in the future electricity generation mixes.

The renewable primary energy consumption (CED renewable) increases in both the scenarios compared to the reference scenario. In detail, CED renewable is 1.7 $MJ_{primary}$ in BS30 and 1.5 $MJ_{primary}$ in the DS30 scenario, while it is equal to 1.0 $MJ_{primary}$ in the RS14 scenario. The environmental impacts are shown in Table 1.3. Both forecasted scenarios involve a reduction of the impacts in almost all the examined environmental categories, except for human toxicity—no cancer effect (HT—nce), ionizing radiation—human health (IR—hh), ionizing radiation—ecosystem (IR—e), particulate matter (PM) and mineral, fossil and renewable resource depletion (MFRRD).

For the HT—nce, IR—hh, IR—e and PM, the increase of the impact is mainly due to the higher penetration of CHP—biomass and PV plants in the forecasted 2030

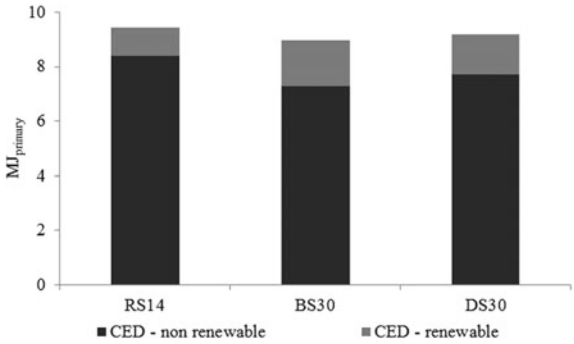

Fig. 1.2 Non-renewable and renewable CED in reference and in forecasted scenarios

Table 1.3 Environmental impact of 1 kWh of electricity in RS14, BS30 and DS30 scenarios

Impact categories	RS14	BS30	DS30
GWP (kg CO_{2eq})	5.75E−01	4.93E−01	5.22E−01
ODP (kg CFC-11 $_{eq}$)	5.58E−08	4.79E−08	5.07E−08
HT—ce (CTUh)	1.39E−08	1.28E−08	1.32E−08
HT—nce (CTUh)	3.91E−08	3.91E−08	3.98E−08
PM (kg PM2.5 $_{eq}$)	2.06E−04	2.88E−04	2.77E−04
IR—hh (kBq U235 $_{eq}$)	8.22E−03	8.89E−03	8.75E−03
IR—e (CTUe)	2.52E−08	2.72E−08	2.68E−08
POFP (kg $NMVOC_{eq}$)	1.53E−03	1.35E−03	1.42E−03
AP (mol H^+_{eq})	2.79E−03	2.42E−03	2.56E−03
T—EU (mol N_{eq})	5.24E−03	4.65E−03	4.89E−03
F—EU (kg P_{eq})	1.21E−04	1.06E−04	1.11E−04
M—EU (kg N_{eq})	4.90E−04	4.33E−04	4.56E−04
F—E (CTUe)	9.44E−01	8.29E−01	8.70E−01
LU (kg $C_{deficit}$)	5.89E−01	5.12E−01	5.40E−01
WRD (m^3 $water_{eq}$)	6.33E−03	5.34E−03	5.69E−03
MFRD (kg Sb_{eq})	2.65E−05	3.92E−05	3.45E−05

energy systems. The increase ranges from +1.8% for HT—nce in DS30 to +39.6% for PM in BS30. In both scenarios, the high penetration of PV in the electricity mix is responsible for a relevant increase in MFRRD impact category (+47.9% in BS30 and +30.2% in DS30).

The reduced share of fossil fuels thermal power plants in the electricity mixes (−12.1% in BS30 and −7.8% in DS30 compared to RS14) involves a reduction of global warming potential (GWP) (−14.4% in BS30, −9.2% in DS30), human toxicity—cancer effects (HT—ce) (−7.6% in BS30, −4.9% in DS30) and photochemical ozone formation potential (POFP) (−12.0% in BS30, −7.3% in DS30), in both sce-

Table 1.4 GWP in the RS14 and in the forecasted scenarios

Scenarios	GWP ($kgCO_{2eq}$/kWh)	Electricity demand (GWh)	Total GWP (tCO_{2eq})
RS14	5.77E−01	22,211	12.78E−07
BS30	4.93E−01	21,512	10.60E−07
DS30	5.22E−01	24,443	12.77E−07

narios. In detail, the reduced impact in these categories is due to the lower production from power plants—natural gas and CHP other fuels, which are the two plants with the highest impact in these environmental categories.

The reduction of the ozone depletion potential (ODP) in both scenarios (−14.0% in BS30 and −9.0% in DS30) is mainly due to the lower generation from thermal power plants fuelled with natural gas. The reduced share of CHP other fuels and power plant—oil in the electricity mixes is mainly responsible for the decrease of terrestrial eutrophication (T-EU), freshwater eutrophication (F-EU), marine eutrophication (M-EU), acidification potential (AP) and freshwater ecotoxicity (F-E). The decrease ranges from −6.6% for T-EU in DS30 to −12.4% for F-EU in BS30.

The lower contribution of power plants powered with natural gas and oil products in both future electricity mix causes a reduction in the impact on land use (LU) (−13.1% in BS30 and −8.4% in DS30), while the reduced contribution from CHP other fuels generation is responsible for water resource depletion decrease (WRD) (−15.7% in BS30 and −10.1% in DS30).

1.3.4 Potential Contribution of Sicily in the Achievement of the 2030 European Climate Target

In order to assess the potential contribution of the Sicilian electricity sector in the achievement of the 2030 European climate target, starting from the impact of 1 kWh of electricity on the GWP impact category, the authors calculated the GHG emissions related to the total electricity demand in the forecasted scenarios (Table 1.4).

The results of Table 1.4 show that the Sicilian electricity sector could contribute to the European climate policy for 2030, reducing GHG emissions of 2.2E−06 tonCO_{2eq} (−17%) in the BS30 scenario, while in the DS30 the variations in the energy generation mix allow to maintain the same level of emissions, with a small decrease equal to −0.1%, even though the electricity demand has increased by 10% with respect to RS14.

1.4 Conclusions

The study presented an integration of the LCA approach and scenario analysis suitable for the evaluation of environmental strategies on a policy level.

In detail, the authors showed the potential contribution of two forecasted electricity scenarios in Sicily to the 2030 Europe climate and energy policy.

The analysis of a wide range of environmental aspects of sustainability through the multi-indicator approach of LCA was carried out. Both the assessed scenarios involve an overall reduction in almost all the environmental impact categories, in comparison with the reference scenario (RS14), confirming that the high penetration of RES could improve the electricity eco-profile significantly. However, with the current state of development of the electricity technologies generation, it is not possible to achieve improvements in the whole set of environmental impacts categories. Then, the integration of the LCA methodology with the scenario analysis could be a useful tool for identifying the potential negative impacts connected to the implemented strategies and could provide a useful support to policymakers in the identification of the more suitable strategies taking into account both the site-specific characteristics of the territory and the most pressing environmental issues.

With reference to the climate target, only the BS30 scenario, characterized by the reduction in the electricity demand and the increase of RES exploitation, could involve a reduction of the GWP. In the DS30 scenario, the benefits due to the increase of the RES are offset by the impacts caused by the electricity demand increase. In order to match the European climate goals strategies aimed at promoting RES, the focus on energy efficiency and on the final consumer's behaviours is mandatory.

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Chapter 2
LCA of Photovoltaic Solutions in the Italian Context

Pierpaolo Girardi and Alessia Gargiulo

Abstract In the present study the main environmental impacts of different solutions for photovoltaic electricity production in the Italian context are discussed. For solution we mean the combination between the cell technology (CdTe, single or multi-crystalline silicon, etc.) and the installation option (roof, ground, etc.). Environmental impacts are analyzed by means of the Life Cycle Assessment approach according to ISO 14040 standard. The life cycle environmental impacts of the different solutions are, also, compared with the impacts of a natural gas combined cycle plant which is, in Italy, the main technology replaced by new photovoltaic installed power. Results show that there isn't a photovoltaic solution which is the best for all the impact categories. All the solutions have several environmental advantages compared to fossil fuel technologies, even compared to natural gas combined cycle. The main negative effect is a relevant land use for ground installations, which represent in Italy almost the 40% of the photovoltaic installed power. Moreover criticalities for what concerns human toxicity impact categories have to be underlined. All the photovoltaic solutions, in the case of non-cancer effect, and five out of eleven, in the case of cancer effects show higher impacts than natural gas combined cycle plant.

Keywords Life Cycle Assessment · Photovoltaic · Environmental impact

2.1 Introduction

The promotion of Renewable Energy Sources (RES) is a pillar of the European strategy for climate and energy (EC—COM/2014/015) and in general of European sustainable development.

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Among RES, photovoltaic is, in Italy, the one with the highest growth in the last decade. Due to strong national promotion policy, only in four years (2008–2011) the average increase of photovoltaic production was around 300%. In 2016 photovoltaic with a production of more than 22 TWh covered almost the 8% of the total national electricity production.

In this framework it is essential to understand if RES, and in particular photovoltaic solutions, whereas contributing to climate change mitigation have the potential to reduce or to increase the contribution of energy system to other environmental impact categories (e.g. air acidification, particulate matter formation potential). A suitable methodology to face this problem is of course the LCA—Life Cycle Assessment—(Sumper et al. 2011). LCA has been widely applied in the field of photovoltaic systems assessment as demonstrated by several literature review studies (Peng et al. 2013). However, the same review studies have underlined the need for further research. In particular there is the need to enlarge the system boundaries because often end of life impacts are not investigated. Moreover there is the need to increase the number of the considered environmental impact categories since, usually, they are only limited to greenhouse effect and to energy payback time (Gerbinet et al. 2014).

In the following paragraphs, after a brief description of the Italian photovoltaic system evolution, we will discuss the LCA of different photovoltaic solutions following the ISO 14040 scheme.

2.2 Evolution of Photovoltaic Production in Italy

The production of electricity from photovoltaic increased in Italy in the last decade (Fig. 2.1). Due to strong national promotion policy, only in four years (between 2008 and 2011) the average increase of photovoltaic production was around 300%. Photovoltaic has covered 18.5% in 2014 and 20% in 2016 of the electricity produced by renewables (GSE 2015), compared to less than 1% in 2009. In 2016 with a production of more than 22 TWh (almost the 8% of the national electricity production) it was the second renewable energy sources, after hydropower.

As regards the size of the plants, recent years show a trend towards smaller plants. During 2014 for example, new installations were essentially residential with an average power of 8.1 kW, considerably lower than the past years. The average plant size, in fact, was three times higher in 2012 and six time higher in 2011. As regard the installation options, almost 40% of the installed power is ground mounted, almost 50% is on buildings (mainly on roof) and 6% is mounted on greenhouses or canopies (such as in car parking). The remaining 4% covers different installation options, sometime really interesting like those on the highway noise barriers (GSE 2015).

Concerning cells technologies and materials, single-crystalline silicon panels have decreased in favour of multi-crystalline panels. In all the Italian regions the multi-crystalline silicon panels cover the majority of the installed power, followed by single-crystalline silicon. Other technologies like thin film cover a little percentage

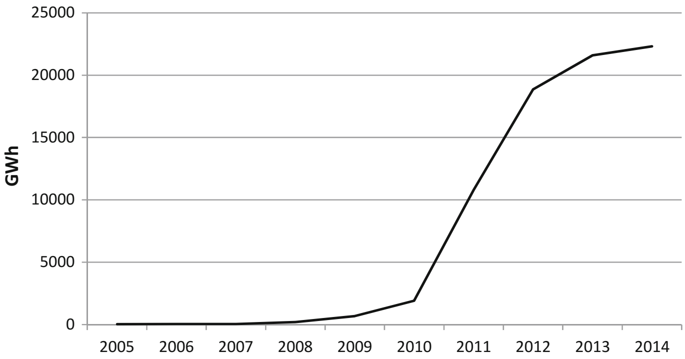

Fig. 2.1 Production from photovoltaic plant. *Source* GSE (2015)

of the installed power. According to GSE (2015), in 2014 more than 72% of the installed power at national level is in multi-crystalline silicon panels, 21% in single-crystalline silicon panels, while remaining technologies accounts for only 7% of the installed power.

From a technology point of view a decisive evolution towards more efficient and low impact solutions is likely in the coming years.

For example the results of APOLLON, a recent EU research project on concentrated photovoltaic outline that this solution could emit per kWh only the 5% of CO_2 eq emitted by a natural gas combined cycle plant (RSE 2014). Similar results come from other studies (Fthenakis and Kim 2013). Also without switching to concentrated photovoltaic, other solutions are available to increase photovoltaic systems performance. For example silicon heterojunction (SHJ) cells offer high efficiencies and several advantages in the production process compared to conventional crystalline silicon solar cells (Louwen et al. 2015). The use of bifacial modules can reduce up to 38% the life cycle CO_2 emission of a single-crystalline system (Gazbour et al. 2016). Shifting from the conventional cell technology to the state-of-the-art PERC (Passivated Emitter and Rear Cell) technology will reduce the energy payback time and greenhouse gas emissions for photovoltaic electricity generation (Luo et al. 2018). Finally, one-axis tracking installations can improve the environmental profile of photovoltaic systems by approximately 10% for most impact categories (Leccisi et al. 2016).

2.3 Goal and Scope

The goal of the present study is to compare, from an environmental point of view, different existing technologies and installation options for electricity production from photovoltaic panels.

To better understand the role of photovoltaic in sustainable development of the national electric system, the different photovoltaic (PV) solutions are also compared with a Natural Gas Combined Cycle (NGCC) power plant. As matter of fact, NGCC is (in terms of efficiency, greenhouse gas and particulate matter emissions) the best fossil fuel technology for electricity production. Moreover it is the main technology pushed out of the market by the new photovoltaic installed power (GME 2014).

The functional unit is 1 kWh delivered to the Italian distribution network. System boundaries include all life cycle phases, including end-of-life.

The considered technologies are: amorphous silicon (a-Si), copper-indium–gallium-selenium thin film (CIS), cadmium telluride thin film (CdTe), single-crystalline silicon (single-Si), multi-crystalline silicon (multi-Si), ribbon-silicon (ribbon-Si).

The considered installation options are: on roof-integrated, on roof-not integrated, ground mounted. The combination between technologies and installation options gives the eleven solutions, illustrated in Table 2.1.

Concerning impact categories, we followed the indications of the working group of European Commission on Product Environmental Footprint Category Rules (PEFCR)[1] for photovoltaic.

In particular, although other documents and updates have followed, we referred to the interim report of July 2014 (Frischknecht and Itten 2014) in which the impact categories are classified by relevance. Impact categories classified as highly relevant have been selected. Among them we excluded the "water scarcity" since it is too much dependent on local conditions for the scope of the present analysis. Moreover the characterization method selected by JRC's guidelines for life cycle impact assessment (EC-JRC 2011) is far from being reliable (it has "to be applied with caution").

Table 2.1 Solutions taken into account in the analysis

Solution	Peak power (kWp)
Slanted-roof installation, a-Si, laminated, integrated	3
Slanted-roof installation, a-Si, panel, mounted	3
Slanted-roof installation, CdTe, laminated, integrated	3
Slanted-roof installation, CIS, panel, mounted	3
Slanted-roof installation, multi-Si, laminated, integrated	3
Slanted-roof installation, multi-Si, panel, mounted	3
Slanted-roof installation, ribbon-Si, laminated, integrated	3
Slanted-roof installation, ribbon-Si, panel, mounted	3
Slanted-roof installation, single-Si, laminated, integrated	3
Slanted-roof installation, single-Si, panel, mounted	3
Open ground installation, multi-Si	570

[1]Product Environmental Footprint Category Rules (PEFCRs) provide specific guidance for calculating and reporting life cycle environmental impacts.

In order to compare the PV system with NGCC we have included two impact categories (acidification potential, photochemical ozone formation potential), that, even though with medium relevance for PV systems, are relevant for conventional fossil fuel power plant. Finally we included the Cumulative Energy Demand (CED) both renewable and non-renewable as it is relevant for estimating the energy payback time of PV solutions.

2.4 Life Cycle Inventory

As regard the inventory analysis, the data used for background systems came from Ecoinvent version 3 (Wernet et al. 2016). When dealing with PV systems, environmental impacts are mainly related to the materials used for module production and installation and to efficiency in electricity production.

Given an installed power (at peak), the production depends on several factors such as solar radiation, modules inclination and cell efficiency. This factors can be expressed by the so-called yield factor (in hours) which is the ratio between the annual production of a PV plant and its installed peak power.

In this study, on the basis of GSE's statistics (GSE 2014) an average yield factor of 1140 h was estimated.

Life time of photovoltaic system has been assumed equal to 30 years.

The PV system includes the panel, the mounting structure and the inverter.

2.5 Life Cycle Impact Assessment

As discussed above the impact assessment has been carried out using eight impact categories (see Table 2.2) and the CED (both renewable and not). Besides being compared to each others, the PV solutions impacts are compared to the electricity production from a NGCC plant. All impacts are referred to the functional unit, that is 1 kWh delivered into the distribution network, with the hypothesis that the PV panels are connected to the distribution network, while the NGCC power plant is connected to the high voltage grid. For this reason the impacts of the kWh delivered by the NGCC include 6.3% of grid losses.

In Tables 2.3 and 2.4 the results of impact assessment are reported for the eleven PV solutions. Values higher than the average are highlighted in red.

As regards the comparison among the different PV solutions, results show that there is not a better or a worse solution for all the impact categories taken into consideration (Fig. 2.2).

On the other hand it can be noticed that the single-Si, slanted roof installation mounted (i.e. not integrated) shows impacts higher than average for six categories and it is the solution with the worst environmental performance for four impact categories out of eight.

Table 2.2 Selected impact categories

Impact cateogories	Method	Unit	References	Relevance (PEFCR)
Climate change	GWP 100 anni	kg CO_2 eq	IPCC (2007)	High
Ecotoxicity, freshwater	USeTox	CTUe	Rosenbaum et al. (2008)	High
Human toxicity, cancer effects	USeTox	CTUh	Rosenbaum et al. (2008)	High
Human toxicity, non-cancer effects	USeTox	CTUh	Rosenbaum et al. (2008)	High
Land use	Soil organic matter lost	kg soil organic carbon	Milà i Canals et al. (2007)	High
Particulate matter/respiratory effects	RiskPoll	kg PM2.5 eq	Greco et al. (2007), Rabl and Spadaro (2004)	High
Acidification	Accumulated exceedance (AE)	Moleq H^+	Seppälä et al. (2006), Posch et al. (2008)	Medium
Photochemical ozone formation	ReCiPE	kg NMVOC eq	Van Zelm et al. (2008) as applied in ReCiPe	Medium
Additional information				
Cumulative energy demand, renewable		MJ	Frischknecht et al. (2007)	High
Cumulative energy demand, non renewable		MJ	Frischknecht et al. (2007)	High

From the point of view of the cells technologies, thin film technologies, CdTe and CIS, show always impact below the average. For climate change they show impacts respectively 50 and 30% lower than single-Si. Among "traditional" technologies the a-Si slanted roof installation integrated has on average lower impact than ribbon-Si, multi-Si and single-Si, with the noticeable exception of the impact categories human toxicity-cancer effect. Among crystalline silicon technologies, multi-Si solutions show lower impacts than single-Si solutions (the differences range from 2 to 16%) for almost all impact categories. One exception is human toxicity (both cancer and non-cancer effects) for which the impacts of this two technologies are almost equal.

Tables 2.5 and 2.6 report the comparison between PV solutions and the NGCC. The results are reported in percentage respect to NGCC impacts.

For climate change, acidification, photochemical ozone formation and freshwater ecotoxicity impact categories, all PV solutions show far better performance than NGCC. As regards climate change, the maximum potential impact among the analyzed PV solutions is only 14% of the NGCC impact. For acidification this ratio raises to 27% while for photochemical ozone formation and freshwater ecotoxicity

Table 2.3 Impact assessment results of PV solutions and NGCC

	Climate change	Human toxicity non-cancer	Human toxicity cancer	Particulate matter
	kg CO_2 eq	CTUh	CTUh	kg PM2.5 eq
slanted-roof installation, a-Si, integrated	4.2.E-02	1.3.E-08	1.5.E-09	5.2.E-05
slanted-roof installation, a-Si, panel, mounted	5.4.E-02	1.6.E-08	3.0.E-09	7.2.E-05
slanted-roof installation, CdTe, integrated	3.4.E-02	1.5.E-08	1.1.E-09	3.4.E-05
slanted-roof installation, CIS, panel, mounted	5.0.E-02	1.4.E-08	1.2.E-09	5.2.E-05
slanted-roof installation, multi-Si, integrated	5.6.E-02	1.8.E-08	1.2.E-09	5.7.E-05
slanted-roof installation, multi-Si, panel, mounted	5.9.E-02	1.9.E-08	1.5.E-09	6.4.E-05
slanted-roof installation, ribbon-Si, integrated	5.0.E-02	1.8.E-08	9.5.E-10	5.5.E-05
slanted-roof installation, ribbon-Si, panel, mounted	5.4.E-02	1.9.E-08	1.3.E-09	6.2.E-05
slanted-roof installation, single-Si, integrated	6.6.E-02	1.8.E-08	1.2.E-09	7.0.E-05
slanted-roof installation, single-Si, panel, mounted	7.0.E-02	1.9.E-08	1.5.E-09	7.6.E-05
open ground installation, multi-Si	6.3.E-02	1.7.E-08	2.1.E-09	6.7.E-05
average of PV solutions	*5.4.E-02*	*1.7.E-08*	*1.5.E-09*	*6.0.E-05*

Functional unit 1 kWh. Impacts higher than the average of the PV different solutions are highlighted in red

Table 2.4 Impact assessment results of PV solutions and NGCC

	Photoch. ozone formation	Acidific.	Freshwater ecotoxicity	Land use
	kg NMVOC eq	molc H+ eq	CTUe	kg C deficit
slanted-roof installation, a-Si, integrated	1.6.E-04	4.4.E-04	4.7.E-02	6.4.E-02
slanted-roof installation, a-Si, panel, mounted	2.2.E-04	5.5.E-04	6.7.E-02	8.5.E-02
slanted-roof installation, CdTe, integrated	1.4.E-04	4.0.E-04	4.8.E-02	5.7.E-02
slanted-roof installation, CIS, panel, mounted	1.8.E-04	4.7.E-04	4.4.E-02	6.8.E-02
slanted-roof installation, multi-Si, integrated	2.4.E-04	5.2.E-04	5.8.E-02	7.4.E-02
slanted-roof installation, multi-Si, panel, mounted	2.6.E-04	5.5.E-04	6.3.E-02	8.0.E-02
slanted-roof installation, ribbon-Si, integrated	2.3.E-04	4.9.E-04	5.6.E-02	7.0.E-02
slanted-roof installation, ribbon-Si, panel, mounted	2.5.E-04	5.3.E-04	6.1.E-02	7.7.E-02
slanted-roof installation, single-Si, integrated	2.7.E-04	6.1.E-04	5.9.E-02	8.1.E-02
slanted-roof installation, single-Si, panel, mounted	2.9.E-04	6.4.E-04	6.4.E-02	8.7.E-02
open ground installation, multi-Si	2.6.E-04	5.4.E-04	7.2.E-02	5.6.E+00
average of PV solutions	*2.3.E-04*	*5.2.E-04*	*5.8.E-02*	*5.7.E-01*

Functional unit 1 kWh. Impacts higher than the average of the PV different solutions are highlighted in red

Table 2.5 Impact assessment results of different PV solutions compared to NGCC (values in percentage)

	Climate change	Human toxicity non-cancer	Human toxicity cancer	Particulate matter
	kg CO_2 eq	CTUh	CTUh	kg PM2.5 eq
slanted-roof installation, a-Si, integrated	8%	126%	112%	45%
slanted-roof installation, a-Si, panel, mounted	11%	155%	219%	62%
slanted-roof installation, CdTe, integrated	7%	139%	83%	30%
slanted-roof installation, CIS, panel, mounted	10%	133%	86%	45%
slanted-roof installation, multi-Si, integrated	11%	170%	86%	50%
slanted-roof installation, multi-Si, panel, mounted	12%	176%	109%	55%
slanted-roof installation, ribbon-Si, integrated	10%	170%	70%	48%
slanted-roof installation, ribbon-Si, panel, mounted	11%	177%	95%	54%
slanted-roof installation, single-Si, integrated	13%	170%	86%	60%
slanted-roof installation, single-Si, panel, mounted	14%	175%	107%	66%
open ground installation, multi-Si	12%	159%	151%	58%
NGCC	100%	100%	100%	100%

Functional unit 1 kWh. PV impacts higher than NGCC impacts are highlighted in red

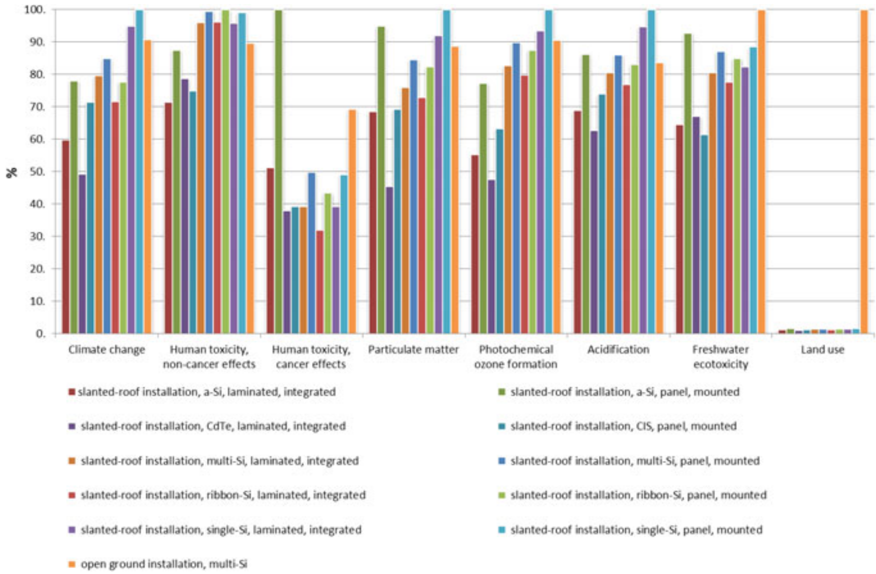

Fig. 2.2 LCIA comparison among PV solutions. Functional unit 1 kWh

it is respectively 36 and 37%. Also for particular matter, PV solutions show better performance than NGCC, but in this case the values are closer and the worst PV solution (single-Si, not integrated) reaches the 66% of the NGCC value. This means that PV solutions have the potential to contribute to this impact category with the same order of magnitude of a NGCC power plant.

On the contrary PV solutions show worse environmental performance for what concern human toxicity impact categories. All the PV solutions generate higher impacts than NGCC power plant in the case of non-cancer effect. For toxicity-cancer effect, five out of eleven solutions entail higher impacts, and the remaining show values very close to those of NGCC. Finally, as foreseeable, for the land use impact category, PV open ground installation is by far the worst solution not only among PV solutions, but also compared to NGCC.

Besides the discussed environmental impact categories, dealing with photovoltaic it is important also to look at the so called CED. CED is in fact linked to the Energy Pay Back Time, which is one of the most used indicator to evaluate PV solutions (Gerbinet et al. 2014). Differences among PV solutions are little and related only to non-renewable energy use. The worst performer under this point of view is again the single-Si, slanted roof installation mounted, while the best one is the roof-integrated installation of CdTe modules (Fig. 2.3).

All the PV solutions have a CED which is almost the half of the NGCC CED value.

Table 2.6 Impact assessment results of different PV solutions compared to NGCC (values in percentage)

	Photoch. ozone formation	Acidific.	Freshwater ecotoxicity	Land use
	kg NMVOC eq	molc H+ eq	CTUe	kg C deficit
slanted-roof installation, a-Si, integrated	20%	19%	24%	16%
slanted-roof installation, a-Si, panel, mounted	27%	23%	34%	21%
slanted-roof installation, CdTe, integrated	17%	17%	25%	14%
slanted-roof installation, CIS, panel, mounted	22%	20%	23%	17%
slanted-roof installation, multi-Si, integrated	29%	22%	30%	18%
slanted-roof installation, multi-Si, panel, mounted	32%	23%	32%	20%
slanted-roof installation, ribbon-Si, integrated	28%	21%	29%	17%
slanted-roof installation, ribbon-Si, panel, mounted	31%	23%	31%	19%
slanted-roof installation, single-Si, integrated	33%	26%	30%	20%
slanted-roof installation, single-Si, panel, mounted	36%	27%	33%	21%
open ground installation, multi-Si	32%	23%	37%	1354%
NGCC	100%	100%	100%	100%

Functional unit 1 kWh. PV impacts higher than NGCC impacts are highlighted in red

2.6 Conclusions

Taking into account the rapid growth of the photovoltaic electricity production in Italy, this study was aimed at comparing the environmental performance of the different PV solutions installed in Italy. Life cycle environmental impacts of the different solutions were, also, compared with the impacts of a natural gas combined cycle plant which is, in Italy, the main technology replaced by new photovoltaic installed power.

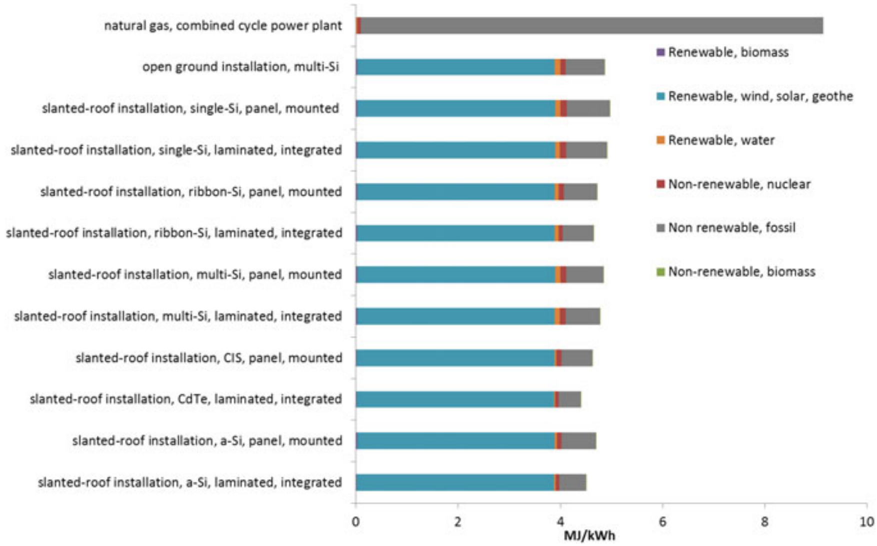

Fig. 2.3 LCIA comparison of PV solutions and NGCC. Impact category: cumulative energy demand. Functional unit 1 kWh

Life cycle environmental performance of a PV power plant depends on both the cell technology (a-si, multi-si, thin film, etc.) and the installation type (on roof integrated or not integrated, open ground mounted).

Results show that there is not a better solution (i.e. combination of cell technology and installation type) for all the analyzed impact categories. Single-Si, slanted roof installation mounted (i.e. not integrated) is the solution that most frequently has the worst environmental performance over the considered impact categories.

On the other hand, photovoltaic solutions lead in general to potential environmental impacts definitely lower than natural gas combined cycle plants. Nevertheless it is important to underline that all PV solutions show criticalities for what concerns human toxicity. As matter of fact all the PV solutions, in the case of non-cancer effect, and five out of eleven, in the case of cancer effects, show higher impacts than a NGCC plant. Concerning particular matter, PV solutions have in general better performance than NGCC, but in this case the values are closer and the worst PV solution reaches the 66% of the NGCC impact. This means that PV solutions can potentially contribute to this impact category with the same order of magnitude of a NGCC power plant. Moreover, open ground solution, which cover in Italy almost the 40% of the photovoltaic installed power, shows a huge amount of land use, when compare to fossil fuel power plants.

The results discussed above refer to the nowadays Italian situation, while the photovoltaic technology is evolving rapidly. Subject of further investigations could be the assessment of technology innovations such as silicon heterojunction, bifacial modules or one-axis tracking installations, especially for impact categories, such as particular matter and human toxicity, for which PV systems and NGCC have similar performance.

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Chapter 3
Geothermal Energy Production in Italy: An LCA Approach for Environmental Performance Optimization

Maria Laura Parisi and Riccardo Basosi

Abstract Geothermal energy is a resource of natural and renewable energy and its exploitation in the Italian Tuscany Region contributes significantly to the regional share of electricity generation from renewable sources, with a value that has grown to about 35% in 2015. The energy produced by geothermal source, such as that produced by other energy sources, generates non-negligible impacts on the environment, closely related to the site-specificity of the source itself. This preliminary study analyzes the operational phase of seven geothermal plants located in the three main Tuscan geothermal areas, with a specific focus on the impacts generated by emissions into the atmosphere. The aim is the assessment of the geothermal power plants environmental performances in relation to the geomorphological characteristics of the sites and the technologies used to exploit the resource.

Keywords Geothermal · Electricity · Power plants · Life cycle assessment
Renewable energy

3.1 Introduction

In 2015, geothermal energy contributed to about 1% of the global electric power generation in the world, through the activity of 613 geothermal power plants for a total installed geothermal power generation capacity of 12,640 MWe (Bertani 2015). In this context, European power plants account for 2,133 MWe of the world installed

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capacity while Italian ones for 916 MWe, basically all located in Tuscany, in the area of Larderello-Travale (795 MWe) and of Monte Amiata (121 MWe).

In this Italian region, where historically geothermal energy started to be used for electricity production, it contributes significantly to the regional electrical and energy needs (Bertani 2015; Bravi and Basosi 2015). In Italy, there are two main geothermal areas at high enthalpy that are currently exploited, both located in southern Tuscany, namely the boraciferous zone in the Southeast of Pisa and West of Siena and the area of Monte Amiata in the Northeast of Grosseto and Southeast of Siena.

Stated that the only clean energy is that one which is not consumed (i.e., saved), geothermal energy can be considered a sustainable and renewable energy only if employed in a suitable environmental manner. In fact, geothermal sources properties are deeply connected to geo-mineralogical phenomena and to highly site-specific factors that have allowed the formation, store, and conservation of the reservoirs.

As any energy source, the use of geothermal energy produces impacts on the environment that are, in turn, very site-specific because of the nature of the resource and its characteristics that change according to the geological age and reservoir depth. Moreover, additional impacts are connected with the particular technology employed for power generation from geothermal energy.

In the context of the environmental impacts generated by anthropic activities, the debate concerning whether or not the geothermal power plants contribute to the global climate change issue has assumed a certain importance in Italy, and Tuscany in particular.

To date, in the analysis of the CO_2 and other greenhouse gases emissions, neither the Kyoto Protocol nor the Intergovernmental Panel on Climate Change (IPCC 2008) have considered the release of greenhouse gases of geothermal origin quantitatively significant as compared to global emissions. This approach has been based on the concept that natural CO_2 emissions from geothermal areas are comparable with those connected with the exploitation of the same area for energy purposes, thus neglecting the temporal variable altogether. Such stance could be considered controversial as the assumption regarding the comparability between greenhouse gases emissions released by a geothermal power plant in all its service life (about 25–30 years) and natural greenhouse gases emissions generated in hundreds of thousands of years might appear in contrast with all the initiatives launched to meet the 2 °C global temperature target (Paris Agreement, COP21 2015 and COP22 2016) in 2030. Indeed, natural CO_2 degassing phenomena have been identified and studied in Italy suggesting that they are strongly affected by the geological and hydrogeological settings of a particular region (Chiodini et al. 1999, 2005). Sometimes, rarely, there are areas without active power plants where CO_2-rich gas emission values are comparable or even higher than those typical of geothermal power generation activity (Chiodini et al. 2007; Frondini et al. 2009).

In addition to the climate change concerns, power generation from geothermal energy can also be significantly responsible for impact on other environmental categories on a local and regional territory scale.

Stated that geothermal energy is definitely a valuable resource with a large potential (it is estimated that only a small part of the available heat is exploited to date) and that, when available, it should always be exploited, it is necessary to implement appropriate technologies that would be able to minimize as much as possible the pressure on the environment of this activity.

Moreover, concerning the use of geothermal energy sources, the economic and energy sustainability could be improved by linking the direct use of cascading heat to electricity production. Indeed, the potential offered by multiple uses, even thermal ones, would allow for a further optimization of the geothermal energy use that nowadays seems to be excessively aimed at the most valuable vector, namely the electric vector.

To address this issue, life cycle assessment offers a powerful methodological approach for the evaluation of the environmental performances of existing geothermal power plants and for the investigation of potential impacts associated with new projects prior their construction in order to define the best strategies and implement suitable methods for environmental emissions mitigation or annihilation.

Given such framework, this study has been developed for the assessment of the environmental performances of selected Italian geothermal power plants for electricity production from an LCA perspective.

3.2 Environmental Impacts of the Geothermal Resource

In the literature, there are several studies dealing with the impacts generated by geothermal power plants (Hagedoorn 2006), some of which approaching this topic in a life cycle perspective (Sullivan et al. 2010; Bayer et al. 2013; Manzella et al. 2018) while others propose simulations for the production of energy in a sustainable way through management models of geothermal sources (Axelsson and Stefansson 2003).

As already discussed, geothermal energy is an energy source that generates impacts in the environment, some of which are highly site-specific, such as land use and effects on biodiversity, subsidence phenomena, heat dispersion in the surrounding environment in relation with the technology used; water consumption in the drilling and operating phases of the plant (which increases in the presence of fluid return systems in the geothermal reservoir), radon emissions; soil emissions related to the contaminants present in the volume of geothermal fluid extracted and the resulting waste, particularly for liquid-dominant systems.

The emissions into the atmosphere connected with geothermal activities are probably the environmental aspect most discussed at present and for this reason, they are the focus of the analysis carried out in this study.

Gases are naturally present in the geothermal fluids, dissolved in the liquid phase or free in the vapor phase depending on the pressure and the temperature of the tank. Gases commonly found in geothermal fluid are carbon dioxide (CO_2), hydrogen sulfide (H_2S), hydrogen (H_2), nitrogen (N_2), methane (CH_4), ammonia (NH_3), argon (Ar), and radon (Rn) (Fridriksson et al. 2016).

These gases are called non-condensable gases (NCGs) since they do not condense in the same conditions as water vapor, usually present in the gaseous phase in higher quantities, but remain in the gaseous phase generating several problems from the point of view of both productivity and efficiency of the plant and from the point of view of the environmental sustainability of energy generation. For this reason, they must be removed from the condensers and heat exchangers.

However, the removal of NCGs has both an economic and an energy impact, as the elimination process requires additional costs and the use of part of the energy produced by the plant itself. Therefore, studying the chemical composition of the tank can be useful during the development of the plant, as it allows to set up the system with technologies suitable for the management of the process depending on the composition of the geothermal fluid.

3.3 Methodological Issues

3.3.1 Goal and Scope of the Study

The aim of the research is the calculation and evaluation of the environmental performances of selected Italian geothermal power plants for electricity production from an LCA perspective in order to assess and recommend solution for the minimization of the environmental impacts in the exploitation of geothermal energy. In particular, the study is focused on the atmospheric emissions of NCGs contained in geothermal fluids produced during the operational phase. The geothermal plants considered in this study are located in the two main Tuscan geothermal areas:

– The boraciferous zone in the Southeast of Pisa and West of Siena which includes the geothermal fields of Larderello and Travale-Radicondoli
– The area of Monte Amiata in the Northeast of Grosseto and Southeast of Siena, where the geothermal fields of Bagnore and Piancastagnaio are located.

A map of the Tuscany geothermal areas showing the geothermal fields exploited for electricity generation is reported in Fig. 3.1.

To date, in Tuscany, there are 34 geothermal power plants managed by Enel Green Power, located in the four territorial areas of Larderello, Radicondoli, Lago and Piancastagnaio belonging to the provinces of Pisa, Siena and Grosseto, with an electricity generation in 2016 equal to about 2% of the national electricity production and 35.6% of the total regional production (TERNA 2016).

The geothermal reservoirs exploited in the Larderello-Travale/Radicondoli and Monte Amiata areas are two: a shallow reservoir contained in the cataclastic levels of carbonate rocks that produce superheated steam, and a deeper and much more diffused reservoir characterized by a crystalline (metamorphic and granitic) rock system located at a depth greater than 2 km.

Fig. 3.1 Location of geothermal areas for electricity production in Tuscany

In the deeper steam-dominated reservoir of the Larderello-Travale/Radicondoli geothermal field, there are values of 20 MPa and 300–350 °C at about 3 km of depth. The percentage of steam phase is between 92 and 98% and the geothermal fluid is characterized by NCGs content ranging between 1 and 20% in weight, with an average value of about 11%. The fluid composition in weight is about 97% of CO_2, 0.3% of H_2S, 0.02% H_2, 1.5% of CH_4, 1.2% of N_2, 0.2% of NH_3, and 0.02% of H_3BO_3.

In the Monte Amiata area, the deeper reservoir is liquid-dominated and there are values of 20 MPa and 300–350 °C between 2.5 and 4 km of depth. In this area, the shallow steam-dominated reservoir is characterized by NCGs content ranging between 1 and 20% in weight (average value 11%), while in the deeper reservoir, the chlorine-alkaline geothermal fluid has a high content of ammonium and boric acid and presents NCGs content ranging between 5 and 10% in weight (average value 8%, della Terra 2008). The relative percentage values that characterize the fluid composition in weight are about 97.3% of CO_2, 0.1% of H_2S, 0.05% H_2, 0.9% of CH_4, 0.1% of N_2, 1.5% of NH_3, and 3.7% of H_3BO_3.

In order to evaluate the potential impact associated with geothermal power plants production of electricity, for this analysis, the system boundaries are set to account only for the operational phase of the geothermal power plants, as shown in Fig. 3.2.

The consumption of resources associated with the drilling, construction, and operation of the wells and the additional materials needed for the construction and operating of geothermal plants have not been included. This is because the impact of plant construction is diluted over the assumed 25 years of plant operation and only accounts for a small amount of total foreground and background emissions (2% of yearly CO_2 emissions, 1% of yearly fossil energy use, 1% of annual matter flows, according to Ulgiati and Brown (Brown and Ulgiati 2002). The functional unit employed is the

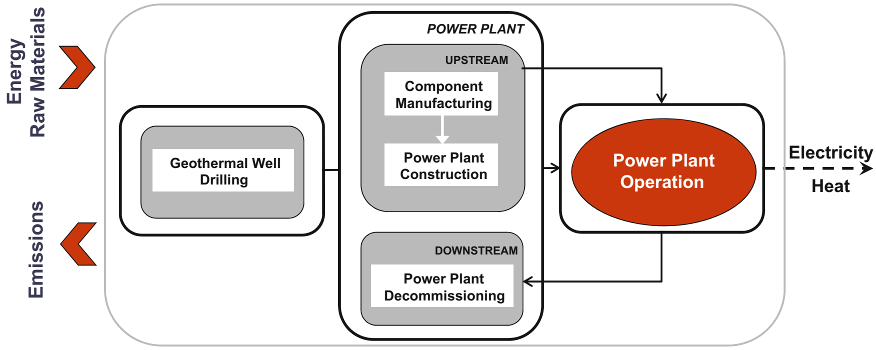

Fig. 3.2 System boundaries for the LCA of geothermal power plants; in this study, only the electric energy production phase has been considered

electricity (1 MWh) produced by the various plants through the conversion of the geothermal energy.

The reference timeframe for the analysis of the plants operational phase is the 2010–2014 historical series.

3.3.2 *Life Cycle Inventory Analysis*

The Tuscan geothermal power plants investigated in this study are seven and are located in the territorial areas of Piancastagnaio, Larderello, and Radicondoli. In details, power plants are:

– Monte Amiata: Bagnore 3, Piancastagnaio 5
– Larderello: Farinello, Sesta 1 and Nuova Larderello
– Travale-Radicondoli: Nuova Radicondoli 1 and Nuova Radicondoli 2

In Table 3.1, general information about the analyzed power plants are reported.

Data concerning the air emissions generated by these power plants have been collected from the geothermal areas monitoring annual reports published by ARPAT (Tuscany Regional Agency for Environmental Protection) during the period 2010–2014 (ARPAT 2010–2014). ARPAT measurement data are based on sampling of the emission materials from the geothermal power plants chimneys in a defined period of the year. We assume that this sample values correspond to an average constant value during the whole year considered.

This study is focused on the potential environmental impacts associated with the emissions of those NCG that are found in greater concentration in the geothermal fluid (CO_2, CH_4, NH_3, H_2S) and that give an appreciable contribution on the selected impact categories, in addition to mercury (Hg) emissions, other metals (lead, arsenic, selenium, chromium, cadmium, nickel, copper, manganese, vanadium), metal compounds (arsenic, antimony and mercury compounds), and boric acid (H_3BO_3).

Table 3.1 Characteristics of the Tuscan geothermal power plants selected for the study: all power plants are equipped with the AMIS (abatement of mercury and hydrogen sulfide) technology in the historical series 2010–2014

Power plant	Province	Start date	Nominal MWe	Effective MWe	Technology
Piancastagnaio 5	Siena	1991	20	13.6	Dry steam
Bagnore 3	Grosseto	1998	20	19.4	Flash
Farinello	Pisa	1995	60	46	Dry steam
Sesta 1	Siena	2002	20	5.5	Dry steam
Nuova Larderello	Pisa	2005	20	14.3	Dry steam
Nuova Radicondoli 1	Siena	2002	40	33.6	Dry steam
Nuova Radicondoli 2	Siena	2010	20	17.5	Dry steam

All data regarding these chemical species have been normalized with respect to the functional unit using the values of power plants inventoried from ARPAT during the tests. Sampling was performed with the following temporal frequency: Piancastagnaio 5 years 2010, 2011, 2013 (no CO_2), 2014; Bagnore 3 years 2010, 2011, 2012, 2013, 2014; Farinello years 2010, 2011, 2012, 2013; Sesta 1 years 2010, 2011, 2013; Nuova Larderello years 2010, 2011, 2012, 2014; Nuova Radicondoli 1 years 2010, 2011, 2012, 2014; Nuova Radicondoli 2 years 2010, 2011, 2012 (no CO_2), 2013, 2014.

Starting from data regarding the composition of the geothermal fluid characteristic for each geothermal field analyzed and from the production capacities based on the MWh effectively generated in the reference period, an estimate of the geothermal fluid mass flow entering each power plants for every year of the historical series was performed. Thus, it has been possible to calculate the average NCGs emission factors for each investigated power plant as the ratio of mass flows (kg/h) on the average load of the power plants measured in MWe/h.

3.3.3 Life Cycle Impact Assessment

The impact categories that have been selected for this study are those associated with the global warming, soil acidification, and human health issues.

For reasons of continuity and comparability of the analysis carried out in this study with previous studies published in the literature (Bravi and Basosi 2014), we chose environmental indicators defined in the impact assessment method CML 2001 V2.05: the Global Warming Potential on a temporal window of 100 years (GWP100, expressed in kilogram of carbon dioxide equivalent, kg CO_2 eq/MWh), the Acidification Potential (ACP, expressed in kilogram of sulfur dioxide equivalent, kg SO_2 eq/MWh), and Human Toxicity Potential (HTP100, expressed in kilogram of 1,4 dichlorobenzene, kg 1,4-DB eq/MWh).

Table 3.2 Potential environmental impacts associated to the electric power generation from coal-fired and natural gas power plants for the climate change, acidification and human toxicity categories

Indicator	Coal-fired	Natural gas
GWP (kg CO_2 eq)	$1.06 \times 10^{+3}$	$6.40 \times 10^{+2}$
ACP (kg SO_2 eq)	505	1.12×10^{-1}
HTP (kg 1,4-DB eq)	$8.71 \times 10^{+1}$	$6.94 \times 10^{+1}$

In order to perform a more significant evaluation, the results of this analysis have been compared with two other power generation systems of comparable power, namely coal, and natural gas.

The environmental impact potential connected with electricity production from these two kinds of fossil fuels has been taken from the Ecoinvent database v. 2.0 (Emmenegger et al. 2007; Roder et al. 2007), where five life cycle phases are considered: before construction, construction, transportation, operation and maintenance, and demolition of power plants.

This discrepancy among the system boundaries defined in the Ecoinvent processes built for power generation plants and the system boundaries chosen for the analysis developed here would not affect the validity of the comparison proposed in the present study.

In fact, for the electricity produced by coal-fired and natural gas power plants, GWP, ACP, and HTP impact categories are predominantly due to direct emissions during the operation of the power plant. In particular, the operation phase accounts for 95% of GWP in coal and 83% in gas plants and 87 and 40% of ACP and 79 and 64% of HPT, respectively (Emmenegger et al. 2007; Roder et al. 2007).

The Ecoinvent data set employed to calculate the coal-fired and natural gas power plants emissions gave values reported in Table 3.2.

To perform the analysis, the software Simapro v 8.0 has been employed.

3.4 Results and Discussion

In general, geothermal wells reaching over 3000 m of depth are realized through earth's crust drilling that increases the permeability of both geothermal fluids and NCGs. The amount of gases and metals contained in geothermal fluids and releases to the environment depends on several factors: depth and location of the geothermal reservoir; characteristics of the electricity generation (flash, binary, or combined cycle) and the abatement systems.

The outputs of the analysis performed in this study are reported in Fig. 3.3 for the three selected impact categories. Looking at the indicators' trend along the reference historical series, it is evident that the environmental impacts connected with the air emissions associated to the NCGs releases are quite significant, at least, for two

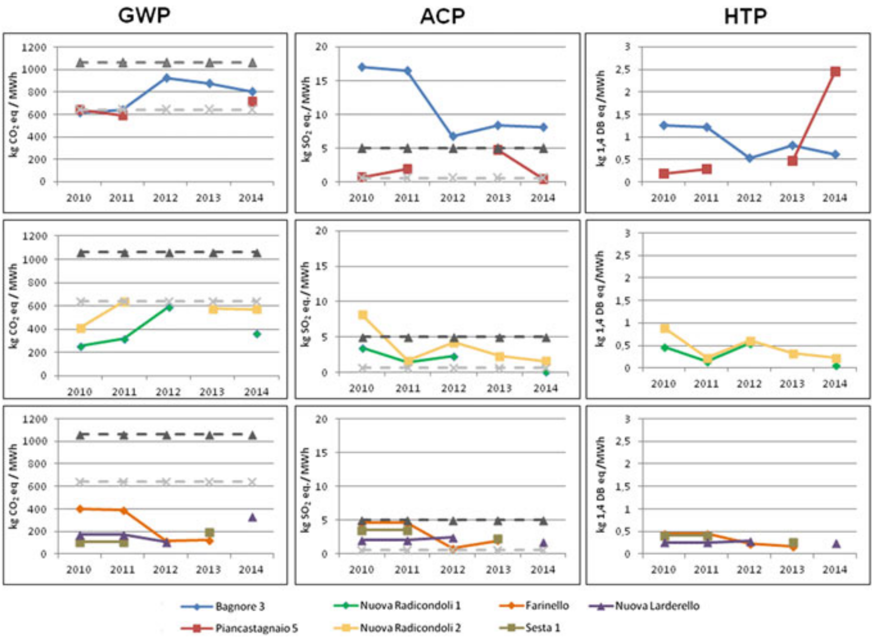

Fig. 3.3 Environmental impacts for GWP, ACP and HTP indicators of the geothermal power plants analyzed in the historical series 2010–2014 (average values for coal-fired and natural gas power plants are in dark gray and light gray, respectively)

impact categories when compared to the potential impact connected with electric energy generation through coal-fired and natural gas power plants.

The GWP computed values depend on the high quantities of CO_2 contained in the fluid that characterize all the analyzed geothermal areas, due to the presence of carbonate rocks in the reservoirs that release gaseous CO_2 in the fluids at high temperatures.

In the Monte Amiata area, the average value of GWP for Bagnore 3 is 771 kg CO_2 eq/MWh, while for Piancastagnaio 5, it has been calculated an average value of 646 kg CO_2 eq/MWh. Such dissimilarity is due to the different conversion technology employed, but most of all it depends on the different geochemistry of the reservoir from which the geothermal fluid is extracted: the elevated values for Bagnore 3 are due to the deeper reservoir exploited by the power plant that extends into metamorphic rock system with consequent presence of high concentrations of CH_4 in the geothermal fluid. In this context, even if CH_4 occurs at lower concentration than CO_2, it has an emission factor 34 times higher than CO_2 and thus it gives a significant contribution to the GWP value. Results for Piancastagnaio 5 reflect the difference in the geomorphological characteristics of the reservoir from which the geothermal fluid is extracted. In fact, the Piancastagnaio reservoir is less deep and it extends into a carbonate rocks system that is characterized by a considerable CO_2 concentration but a minor CH_4 presence compared to the Bagnore reservoir.

In the Travale-Radicondoli area, differences between the two geothermal power plants are mostly due to the installed capacity as both exploit the same geothermal reservoir. The average values are 383 kg CO_2 eq/MWh for Radicondoli 1 and 550 kg CO_2 eq/MWh for Radicondoli 2.

In the Larderello area, lower GWP values were obtained with differences that depend on the effective working capacities of the power plants. The average values are 259 kg CO_2 eq/MWh for Farinello, 196 kg CO_2 eq/MWh for Nuova Larderello and 109 kg CO_2 eq/MWh for Sesta 1.

Concerning the ACP values, for Monte Amiata area, we calculated notable impacts with average values equal to 11.35 kg SO_2 eq/MWh for Bagnore 3 and to 1.94 kg SO_2 eq/MWh for Piancastagnaio 5. In the former case, the higher acidification values depend on the anomalous content of NH_3 in the geothermal fluid extracted from the Bagnore reservoir.

In the Travale-Radicondoli and Larderello areas, we found lower values on the average compared to those computed for the Monte Amiata area that depend on the lower concentration of NH_3. The average values are 1.84 kg SO_2 eq/MWh for Nuova Radicondoli 1, 3.2 kg SO_2 eq/MWh for Nuova Radicondoli 2, 2.96 kg SO_2 eq/MWh for Farinello, 3.11 kg SO_2 eq/MWh for Sesta 1 and 2.07 kg SO_2 eq/MWh for Nuova Larderello. Nevertheless, these ACP values are remarkable when compared to the effects connected with electric energy generation from the two selected fossil resources.

On the other hand, regarding the impacts calculated for the HTP indicator, we found values about 15 times lower than those potentially generated by the use of coal and natural gas for power generation, meaning that at least from the human toxicity point of view the geothermal energy production has a much lower impact.

The effects on this impact category are principally due to the presence of NH_3, H_2S, H_3BO_3, and several metals in the geothermal fluid. Also, in this case, the Monte Amiata area is characterized by more significant results for the sizeable presence of Hg, NH_3, H_2S, and H_3BO_3 in the geothermal fluid in both the reservoir that are exploited by the two power plants: for Bagnore 3 the average value is 0.88 kg 1,4-DB eq/MWh and for Piancastagnaio 5 is equal to 0.5 kg 1,4-DB eq/MWh.

In the area of Travale-Radicondoli, the average computed values are 0.7 and 1.1 kg 1,4-DB eq/MWh for Radicondoli 1 and Radicondoli 2, respectively. Even if mercury has not been detected in 2010, the value of Radicondoli 2 is double compared to the other power plants because of the ammonia and hydrogen sulfide contributions.

In the Larderello area, the computed values are even lower for reasons depending on the geothermal fluid composition and the effective power plants capacity (average values in 1,4-DB eq/MWh: Farinello: 0,33; Sesta 1: 0,361; Nuova Larderello: 0.26). Anyhow, the toxicity potential found in this study show no worrying values. Moreover, in support to our findings, it should be mentioned that a careful monitoring by the ARS (Regional Health Agency) has already been underway for several years and despite the fact that it has detected some critical health problems in the Monte Amiata area compared to other Tuscan contexts, it has not yet shown significant correlations with geothermal activity.

3.5 Concluding Remarks

The results of this study show that exploitation of the geothermal resource, although desirable to replace abuse of fossil resources, does not produce a zero impact and, in particular, it cannot be considered carbon-free.

By focusing the attention on emissions into the atmosphere, it can be shown that there are various factors responsible for variations in the composition and mass of NCGs and metals that are released from the cooling towers of the various plants: location and depth of the reservoirs, characteristics of the technology used (flash, dry steam, binary cycle, combined cycle, etc.), and the abatement systems adopted.

For all these reasons it is evident that, from the comparison among various power plants located in different regions or countries, it is not possible to perform wide range analyses or forecasts valid for multiple sites nor to collect universal data concerning the geothermal power production.

The assessment of the impacts associated with the exploitation activities of geothermal systems is highly site-specific, exactly like the resource. Such assessment should be carried out starting from inventory data as accurate and complete as possible in order to propose solutions and interventions aimed at optimizing the performance of the plants, with a view that should privilege the minimization of environmental pressure, even to the detriment of economic aspects, if necessary.

In addition to abatement systems that work properly and in continuous, the total reinjection of geothermal fluids into the same sampling basin in controlled conditions is the best way to go in order to make geothermal energy a cleaner and safe source of energy with higher social acceptability.

Only with the development of more advanced geothermal exploitation technologies set up to minimize the pressure on the environment, it will be possible to pursue the use of this natural and renewable resource.

To date, progresses made with the closed-loop technology (generally based on the organic rankine cycle—ORC), which exploits geothermal fluids only to transfer heat to a working fluid in a circuit system that supplies the plant, allow to hypothesize viable solutions for geothermal systems even with high enthalpy fluids and with a high concentration of NCGs like those present in Tuscany.

However, we are aware of technological limitations due to site-specific resource conditions, which make it difficult to apply alternative and less invasive technologies (e.g., binary cycles or mixed flash-track systems). The complete re-injection of NCGs in contexts similar to the Monte Amiata one (geothermal fields with a high percentage of NCGs, high pressure and temperatures), that also takes into account aspects related to the economic sustainability of such solutions, should be one of the main challenges that geothermal technological research will have to face in the next future.

In principle, a technology that takes into account environmental and social issues, rather than exclusively economic/financial aspects should be developed. In addition to reducing the pressure of this activity on the environment, this would also allow to overcome or mitigate the social and political concerns that have hitherto slowed down development in the use of a particularly valuable resource at national level and especially for the territories where it is located.

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Chapter 4
Application of LCA for the Short-Term Management of Electricity Consumption

Carlo Brondi, Simone Cornago, Dario Piloni, Alessandro Brusaferri and Andrea Ballarino

Abstract The application of LCA in the energy consumption management can address the sustainability of energy systems. The chapter first aims at summarizing general trends in addressing environmental implication of energy use. Second, LCA methodology is briefly introduced in order to clarify its potentialities and general use in the energy field area. In particular, LCA can contribute to select the best technological choices for an energy system. A challenge in the use of LCA is identified in the representation of a complex system in which the energy producers' contribution changes on a temporal basis. Two approaches are proposed for the LCA use in the short-term perspective: attributional LCA and consequential LCA. The proposed approach examines the application of LCA in a short-term perspective. Both approaches can be used to analyze an efficient configuration of the system. However, the more the temporal and geographical area is restricted, the more specific issues have to be adopted to provide a reliable analysis. In particular, consequential and attributional approaches should be used under different hypotheses and with proper adaptation. The proposed approach examines the application of consequential LCA in a short-term perspective, defined as the time span in which the market system has not reacted to a change yet. Moreover, it could claim environmental impact savings in the presence of an accurate model that is able to predict the hourly marginal technology of the near future (one day to 1 week). The future application of the pro-

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posed approach would be a tool that manages to assess the best hourly consumption trajectory in order to minimize environmental impacts.

Keywords Electricity · LCA · CLCA · ALCA

4.1 Introduction

The consideration of energy consumption as a dominant component of an organization's sustainability cost structure has progressively gained interest along years by including more precise and systematic approaches. Nowadays, the use of energy is considered one of the basic aspects of sustainable society (UN 2015). Indeed, for the first time, the United Nations Sustainable Development Goals (SDG) for 2030 include specific energy-related targets for affordable, reliable, and sustainable energy. In particular, the seventh SDG requires to enhance international cooperation to facilitate access to clean energy technology and promote investment in energy infrastructure and clean energy technology.

The sustainable energy concept requires compliance with two sub-objectives: the increase in energy efficiency by the reduction of the energy process intensity and the reduction of the related social and environmental impacts. The first objective can be performed by introducing technological systems that minimize the use of energy in relation to a set of target performances, re-utilizing energy through recovery systems, reducing dispersions and, where necessary, replacing power generation technologies with more efficient options. As far as the second objective is concerned, the minimization of social and environmental impacts requires to properly assess the effects of energy production and then the setup of systematic policy in order to efficiently reduce social and environmental burden. In particular, this second subobjective implies the replacement of more polluting energy sources, the adoption of end-of-pipe solutions to mitigate adverse effects (i.e., re-utilization of residual flows), and improvements of inefficient distribution methods by the use capillary management of the whole energy grid.

At the strategic level, some documents emphasize the importance of introducing systems for "sustainable energy" consumption. With this meaning, the European Community indicates in its strategic plan (EU-Strategic Plan 2016–2020 DG Energy) a production system able to satisfy social demand, supporting the economy, protecting the environment in the long term. This type of energy is pursued through the increase of renewable energy, the improvement of energy efficiency, and energy savings. The 2014–2018 strategic plan of the US Department of Energy (US Department of Energy 2014) further details the objectives for driving down the costs and improving the performance of clean energy technologies. In particular, in the fifth goal, explicit reference is made to smart grids for an optimal integration of clean electricity into intelligent grid, suggesting as main drivers the intra-hour variability and the demand response mechanisms. The same International Energy Agency (IEA) introduced in 2017 in its annual World Energy Outlook a Sustainable Development Scenario. In addition to the first target for universal access to modern energy services, it includes two targets on environmental effects of energy consumption both to mitigate peak in emissions according to Paris agreements and largely limit other energy-related pollutants (IEA 2017).

The need for more precise and refined tools for the assessment of impacts due to energy consumption is also identified by industrial policy agendas that emphasize the importance of smart sensing systems and inventory systemization of industrial aspects that are linked to the energy consumption. In the past, traditional strategic plans frequently aimed to focus on energy efficiencies along the energy life cycle as a prevalent part of environmental policy in the resource and cost perspective (ARC 2009). More recently, industry policy documents are introducing a focus on better calibration of the real environmental effects of energy systems in order to realize a fine-tuning of industrial policies (Siemens 2017). Energy objectives are linked with the UN SDGs and have been focused on environmental effects rather than just on environmental aspects (Siemens 2017). Finally, this objective is also pursued by the use of recent certification and energy audit schemes. In fact, energy management systems need of infrastructure system to collect, analyze, and report data-related energy consumption, and ensure correctness and integrity of that data in order to ensure minimum energy consumption for the current activity (Kahlenborn et al. 2012). Furthermore, the rise in concerns by new consumers about climate change can positively drive changes in demand toward certified green energy as rewarding criteria for producers. Such influence has been partially registered at global level in shifting energy demand toward renewable sources (Deloitte 2017).

4.1.1 The LCA Application in Analyzing Electricity Life Cycle

LCA has been frequently used as a tool to understand implications of energy management options. In particular, the electricity consumption constitutes a wide area of analysis ranging from energy power production to its distribution by national electricity grid. Such widespread use of the methodology can be linked to LCA intrinsic features. In particular, its inclusion of different impact indicators in the final assessment and its ubiquitous assessment of the effects along the whole energy life cycle can represent strength point in comparison to quantitative methods that are focused on site-specific environmental targets. Moreover, LCA allows to include additional mitigation phases, scenario approaches, sensitivity analyses, and consequential modification of the energy system due to assessed options (Fig. 4.1).

LCA applied to energy system is traditionally used to estimate the effect that a certain consumption of energy or an energy generation option produces in environmental terms. A brief qualitative analysis of the SCOPUS search engine emphasizes that scientific papers including LCA in the title and the abstract are increasing. Concurrently, papers including “LCA” and “Energy Systems” in the title and abstract also increase by representing a consistent part of the entire publications. Conversely, articles including “LCA” and “Power grids” represent a minimal percentage of such

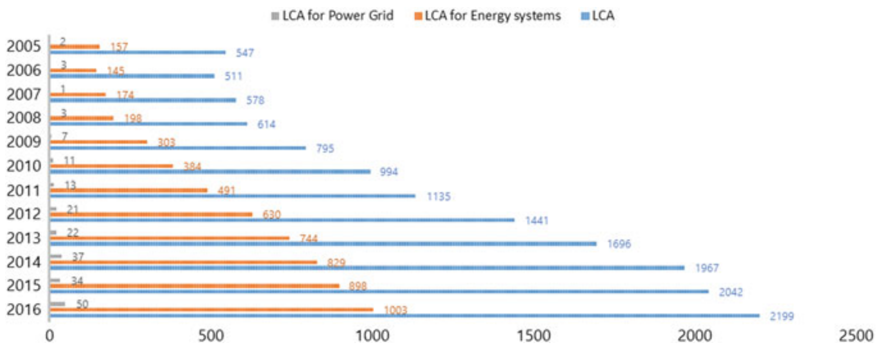

Fig. 4.1 Scopus papers related to LCA and energy (www.scopus.com)

publications. Such discrepancy could partially suggest that current LCA literature is less focused on actual electricity consumption at domestic or industrial level. According to a further in deep analysis, LCA seems to be widely used for the explicit aim to drive innovation in current energy systems by providing alternative solutions to current energy sources in order to identify environmental bottlenecks, to provide comparison and scenarios for technological implementation, and to identify environmental effects on a large scale including substitution of traditional technologies. In general, barriers and uncertainty in LCA study on a real electrical grid depend on the different focuses of the study.

In the Attributional LCA (ALCA), electricity is examined in terms of average impacts both at geographical and at temporal levels. Technological efficiency can be mirrored as average efficiency of a single technology rather than through the efficiency of the modeled plants. Meanwhile, the temporal variance in the environmental impact is assessed according to ex-post provided power in the same reference period. The more an attributional study is focused on a single plant or on a group of plants at regional level, the more LCA datasets may need to be adapted to better depict actual regional power systems and real efficiency of single plant. It must be taken into account though that the maximum detail on the geographical level should contemplate physical and technical limits due to the electricity market.

Main sources of uncertainty can be resumed as follows:

- Inventory methods: Process Chain Analysis (PCA) should be preferred to input–output approaches IOA that are based on monetary data for individual economic sector. In fact, PCA requires a bottom-up approach that uses engineering data and process-specific information preferably obtained directly from the plants. PCA is a time-consuming procedure, but it generally provides results that are more precise (Turconi et al. 2013).
- Efficiency variance: emission factors for the electricity generation are presented for single technology in stationary conditions. However, electricity is usually produced in dynamic conditions, and hence the same technology can change its efficiency according to its operational conditions. Moreover, multi-input and multi-output

systems often occur, i.e., co-combustion power plant is an example of a multi-input system in which a mix of fuels (e.g., coal–straw, waste-coal, etc.) is used as feedstock for the process.

– Change of technology mix: the single contribution of energy production by single technology can change along time according to market and operational constraints. The more the assessment is focused on a limited time span, the more such variability can infer the final assessment of environmental impacts due to the implementation of a certain technology mix. As an example, the contribution of solar and wind power sources can lie on electricity system majorly in certain hours during the day. Such variable contribution is expected to majorly affect grid power production for specific countries in the next decade (Galliani 2016).
– Technology characterization: The operational technology efficiency represents a baseline for the correct assessment of the environmental impact of a power plant. Such assessment tends to vary according to the extension of the assessment. In case the average contribution of technology is included in the assessment, the variance in real efficiency of the same technology has to be aggregated from different power plants. In such case, the age of the power plants assessed should be mirrored in the dataset building. The selection of inappropriate datasets not sufficiently reflecting the real system in focus may clearly result in a significant under or over-estimation of emissions. Conversely, the more the local contribution from single power plants in power supply became relevant, the more their specific features should be analyzed in the technology characterization (i.e., energy recovery efficiency of a plant, the reference year, and the geographical origin of the materials and energy used for the infrastructure).

In the Consequential LCA (CLCA), the marginal effects on existing electricity infrastructures are included in the final assessment as avoided or additional impact from the production of other energy sources. CLCA provides explicit reference on large-scale effects in order to majorly support policymaker and prospective studies (Lund et al. 2010; Olkkonen and Syri 2016). The following potential drawback in CLCA for electricity life cycle can be identified as follows:

– Double counting: CLCA should provide support at operational level to address decisions in a scenario perspective. Unlike an ALCA, a CLCA can overlap with the boundaries of other LCAs, meaning there would be double counting if multiple CLCAs were added together. The reuse of CLCA for different purposes can hardly provide support for large-scale assessment mining (Jones et al. 2017).
– Increased uncertainty: complex relationships (including difficult to model social and economic dynamics) between provided energy and a wider system mean that although a CLCA might be considered more comprehensive there is greater uncertainty in CLCA than in ALCA (Jones et al. 2017). It is therefore valuable to limit the expansion of system boundary to the most relevant processes within the system that are affected by changes in the key variable (i.e., distributed generation uptake) (Mathiesen et al. 2009).
– Approximation of marginal data: CLCA uses marginal data rather than average data to quantify changes within the boundary of the system resulting from the displacement and/or substitution of individual processes. Marginal data implies a

number of assumptions that require a deep knowledge of the electricity system. Such data can be based on perspective or standard impact for a certain technology rather than real substitution at regional or local level.

– Time-related effects: CLCA can include temporal assessments in the form of perspective changes in the electricity life cycle. Such scenarios could fail in the implicit assumption if the temporal scale is related to medium- or short-term changes in the energy supply. In fact, the more the CLCA tends to provide assessments for short-term policies, the more CLCA assume similarities in inventory data with ALCA and need to include bottom-up approaches and coherent substitution at geographic and temporal level (Amor et al. 2014).

Both ALCA and CLCA can be adopted to assess effects in electricity infrastructure in the short term according to a specific technological option. Considering current bottlenecks, ALCA application to electricity life cycle needs to be compliant with application context while CLCA needs to guarantee reliability of assumptions for the estimation of substitution effects. Both prospective analyses can be useful tool for helping decisionmakers to think through future implications of particular technology pathways at a whole system level.

4.2 Short-Term LCA to Address Consumptions Within Electricity Systems

LCA can be used to evaluate the environmental impacts of a technological or operational choice within the electricity life cycle. The adaptation of this methodology to the context of existing electricity systems can support operational decisions in the short term and requires a series of important adaptations:

Temporal adaptation: In the short term, the environmental profile of a kWh supplied to the network becomes a time-varying vector. Such vector depends on the supply conditions of the system at a given instant t. As noted in literature studies, the environmental profile of a unit of energy supplied to the electricity system varies on a temporal basis throughout the day (Soimakallio et al. 2011; Weidema 2003). This aspect can introduce an error if this impact is replaced by the average environmental profile of a kWh on a national scale. Such error can involve remarkable deviations even when the period of analysis of an LCA regards short-term assessments, intended as that period in which it is realistic to hypothesize that the marked system has not yet reacted to possible changes. In fact, the environmental quality of energy can vary along the day so that the specific impact of energy consumption by a single consumer not only depends by the final quantity of consumed energy but also by consumption patterns along the time.

Geographical adaptation: The instant environmental profile is rebuilt considering the local state of the network, the location on the territory of the energy providers. In this sense, the approach of the calculation is more similar to a supply chain LCA in which single contribute from suppliers can change according to operational conditions in order to satisfy a certain demand of energy.

Technological constraints: The environmental profile should take into account the type of technology used and the effective efficiency of the technology within the same technological class. The technological features of the different energy providers should be modeled in order to represent hourly performance based on the different operating conditions. This means that modeling in the short-term perspective requires to introduce both variation in the effective provided power on the electric grid (i.e., efficiency of solar panels depending from solar radiation) and change in operational conditions (i.e., fuel type or efficiency in fuel consumption in a turbogas plant) that can vary the supplier's environmental profiles.

Market constraints: Such aspect needs to be considered in particular in the CLCA since the change in supply involves a previous market bidding phase. In most western markets, the hourly supply of energy is determined through complex mechanisms that include, in general, a programmed hourly power on the electrical grid and a variable ancillary power that depends on the real operating conditions of the electricity grid.

The application of these rules can be detailed from the perspective of the energy consumer. However, the key condition in LCA application depends on the possibility for consumers to be able to alter or not their consumption configuration of provided power on electricity grid by the variation of his demand.

A range of terminology can be used to identify targets for assessments. A number of terms and their definitions as in (Hawkes 2014) are as follows:

- Average Environmental Profile (AEP): The average environmental profile for an average unit of electricity delivered for an electricity system in a certain time span.
- Operating Marginal Environmental Profile (O-MEP): The change in single impact categories to a unit change in electricity demand, where there is no structural change in the electricity system being analyzed (i.e., no power station commissioning or decommissioning, no fuel price changes, etc.).
- Build Margin Environmental Profile (B-MEP): The environmental profile per unit of electricity produced for the next power station included in the market negotiation.
- Marginal Environmental Profile (MEP): The change in environmental profile to a unit change in electricity demand, calculated by weighting the O-MEP and B-MEP to arrive at a "combined" figure.

4.3 Attributional LCA to Analyze Electricity Consumptions in the Short Term

The LCA in an attributional perspective can be applied for ex-post analysis in order to assess the precise impact that a historical consumption of energy has produced in a given time. The application of an attributional approach according to a bottom-up scheme requires that the average national energy profile is replaced by a time-variant environmental profile that derives from the individual contribution of the energy providers and from the contribution of the distribution network. According to this

Table 4.1 Legend of equation system (1) and (2)

OEP(T)	Operating environmental profile of an electricity system that is composed by N supplier at the time $T + \Delta$
[EP_n]	Environmental impact matrix for the supplier N collecting environmental impact categories for single unit (e.g., 1 kWh of provided energy by the supplier n at the time t). Such matrix is time-variant and depends on technology efficiency features of the single provider
EP_{inf}	Additional environmental effects of energy distribution on the electricity grid (e.g., energy dissipation of the distribution network from the single provider to the distribution point and from distribution point to the consumer)
q	Energy in kWh provided by the single supplier at the time t
Q	Total energy provided by the electricity system at the time t
N	Total number of supplier composing the assessed energy system
$T + \Delta$	Time horizon in which energy supply on electricity system is considered

approach, the efficiency of individual providers as well as efficiency features of individual producer can be explicitly counted in the LCA calculation.

The following equation can represent contribution in environmental terms of single supplier from an operational perspective (Table 4.1):

$$\mathbf{OEP}_{\boldsymbol{N},\boldsymbol{T+\Delta}} = \frac{1}{Q}\left(\sum_{n=1}^{N}\mathbf{EP}_{\boldsymbol{n}}(\Delta T) + \mathrm{EP}_{\mathrm{inf}}(\Delta T)\right) \tag{1}$$

$$\mathbf{EP}_{\boldsymbol{n}}(\Delta T) = \int_{t=T}^{T+\Delta}\left[\mathrm{EP}_{e,t}\right] * q_{n,t} \tag{2}$$

The use of the attributional methodology allows a more precise understanding of the impact linked to the consumption of energy in the perspective of the single player. By applying such approach, it becomes possible to identify the qualitative change in the environmental profile on an hourly basis for a certain electricity system. Such figure provides the estimation of the CO_2-eq variation on the Italian electricity system on a specific day for high-voltage production. The main hypothesis in adopting ALCA implies that energy consumption of a single supplier does not alter the configuration of energy production from each producer in a given day. Furthermore, it becomes possible to calculate with more accuracy the measurement of the error compared to the data provided by the commercial databases. In particular, such error results from difference between national AEP and O-MEP as reported in Graph 1. The yellow

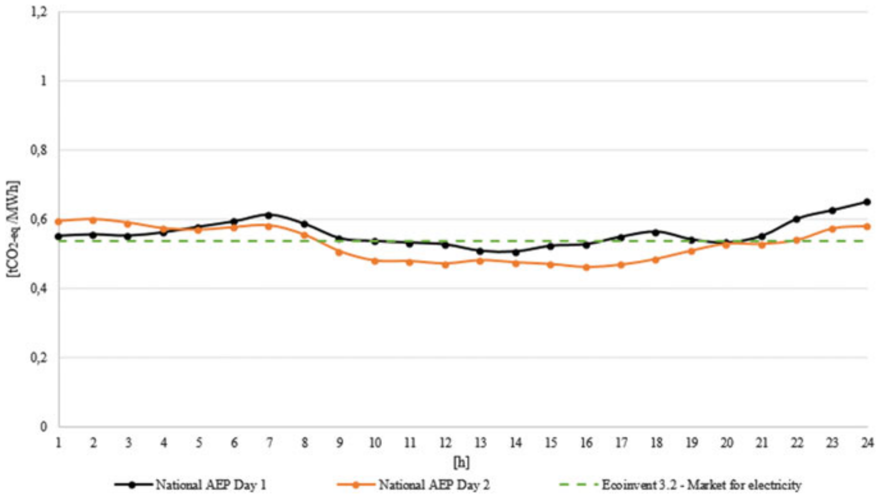

Fig. 4.2 Comparison among AEP CO_2-eq values for two specific days and the AEP provided in an LCA database, with reference to GWP100 values

line represents the value of the impact for the consumption of 1 MWh of electricity at high voltage, as provided by the LCA database (Ecoinvent 2017): the flat line shows that only an average value is provided since the impact is not dependent on the assessed time of the day. Such average data has been compared to hourly national average impacts that are calculated by using data from Italian electricity transmission system operator (Terna 2018) on power production of the represented days and LCA database on specific impacts by technological options for power production. Hence, time dependency introduces variability among different hours of the day. Given that this variability is assessed on data produced after the closure of the day-ahead market, it cannot be exploited in order to shift electricity-intensive productions toward hours characterized with lower than average impacts. Indeed, energy managers should decide energy consumption trajectories before the closure of the market so that the energy provider is able to make the correct bid. Then, once the market is closed, it is possible to use time-dependent impacts only for the already scheduled production, in an ALCA perspective (Fig. 4.2).

The ability to monitor O-MEP depends on the chance to effectively measure the environmental profile of the individual player and then on the related capillarity of the monitoring system. In general, the power and size of the energy production plant can enable a better monitoring at high level. Plants providing power on high-voltage grid and limited variability in power supply present features that facilitate the monitoring. Conversely, local plants for the production of distributed energy in low or medium voltage seem to involve a major complexity in their monitoring. Small-size PV energy productions can represent an example of calculation that can be based on an efficiency estimation rather than on actual measurements.

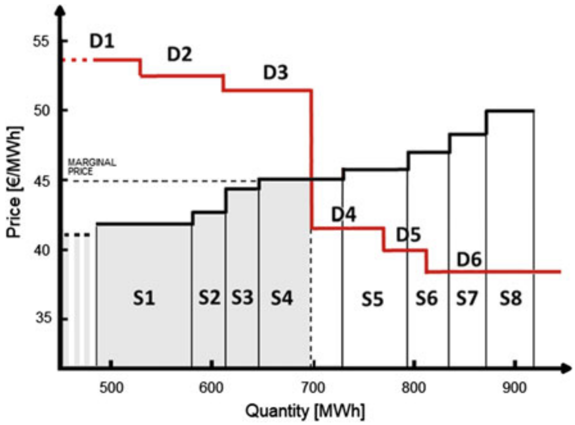

Fig. 4.3 Day-ahead market mechanism to determine energy supply in electric system

4.4 Consequential LCA to Analyze Electricity Consumptions in the Short Term

The consequential LCA appears to be a useful tool to assess the environmental impacts that the modification in electricity consumption produces in the short term. A proper use of the CLCA in the short-term perspective requires first the identification of the instantaneous environmental profile of the electricity system and then the inclusion of the marginal effect linked to energy consumption.

The general implicit hypothesis is that through shifting in time its electricity consumption a consumer can influence the environmental impact of the electrical power supply. In order to assess this shifting properly, the inventory phase has to incorporate market mechanisms: this implies the use of marginal data instead of averages.

- In most advanced countries (US, Canada, Europe, etc.), the prevailing market mechanism for determining both power supply and demand dispatch is the day-ahead market. Such market, divided into zonal sections, defines both the commitment of production units and the consumption profiles of aggregated end users for the following day, on an hourly basis, such as the economic merit is guaranteed. Indeed, the market aggregates both production bids by suppliers, sorting them from minor price to higher price (S bars in Fig. 4.3) and demand bids by users, sorting them in the opposite way (D lines in Fig. 4.3). A production bid is a couple of Q quantity [MWh] and P price [€/MWh] which states the availability of the producer to generate a specific amount Q of energy provided that the awarded price is at least equal to P. On the other hand, a consumption bid (Q', P') sets the availability of an aggregated user to pay at most P' for a specific amount Q' of energy. The interception between these two cumulated profiles addresses three key issues: The marginal price, which is the hourly and zonal value of electricity for both producers and consumers;

- The marginal supplier and the marginal consumer, respectively S4 and D3 in Fig. 4.3;
- The supply and demand power dispatch, defined as follows. Producer bids whose price is lower than the marginal price are accepted. Hence, those producers have the permission to generate the awarded quantity. Conversely, only consumer bids whose price is higher than the marginal allow these users to consume.

Such selection mechanism is performed for different geographical areas determining the power supply dispatch configuration of the electricity grid on an hourly basis. In the framework of the flexible power demand, the use of the CLCA therefore includes the assessment of the combined environmental impacts of the expected energy supply plus the marginal effect on the market.

As for the calculation of the environmental effects from planned supply, it can be calculated through the day-ahead market balance point. O-MEP can be calculated by identifying the technology type and efficiency features of the single supplier. In order to obtain a precise assessment of single impact category, the vector representing the O-MEP should be calculated according to an attributional perspective. This approach is justified by the reduced time margin of the analysis.

As regards the effects of marginal demand, first it is necessary to know the effective shift of the balance point that stems from a change in energy consumption. In general, the demand shift for a certain amount of energy from hour h to hour g produces a precise effect on the perspective of an individual consumer. The demand reduction in the hour h shift to the left the demand curve by excluding from production a certain number of marginal producers that are close to the equilibrium point. Similarly, the increase in demand in the hour g will shift the equilibrium point to the right by including the marginal producers who bid at that time. A greater additional demand for energy in a certain hour produces a greater number of marginal producers that are included in the electricity network configuration. Therefore, the avoided impact coincides with the cumulative B-MEP of the marginal energy producers that do not participate in the energy supply on the day fixed following the change in demand. Similarly, the additional impact coincides with the B-MEP of the marginal suppliers that have to plan an additional energy production for the hour g.

In Fig. 4.4, it is represented the O-MEP for CO_2-eq emissions of two Italian bidding zones, Zone A and Zone B corresponding to the North and to the continental South of Italy. Data on hourly and zonal marginal technology is available on the Italian Power Exchange manager (www.mercatoelettrico.org). As far as the technology-specific impact is concerned, Ecoinvent “At Point Of Substitution” 3.2 should be preferred to the “Consequential long-term” version of the database because of the short-term impact on the system that is needed to be assessed. As Fig. 4.4 shows, the variability of CO_2-eq emissions among different hours of the day is much higher than the one seen for the hourly national average in Fig. 4.2. Moreover, since the marginal technology is not mitigated by any average, the geographical parameter is fundamental: in general, two bidding zone trajectories may differ greatly during the same day. These trajectories were obtained ex-post but in order to be useful in a CLCA perspective, they should be representative of the expected near future.

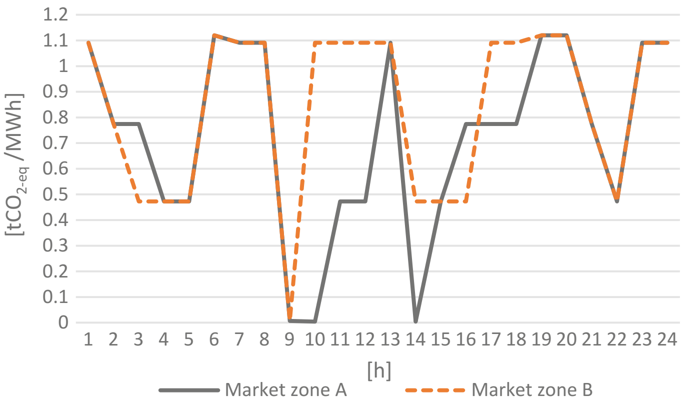

Fig. 4.4 GWP100 impact of two operational marginal environmental profiles representing two Italian bidding zones in the same day

Despite the prospect to accurately estimate the marginal effect that the consumption shift produces in the short term, several applicative and structural drivers can strongly influence the application of this approach.

Consumption-change effects to bid effect. As discussed, the effect of increasing or reducing demand produces a certain shift in the resulting equilibrium point of the day-ahead market. This further knowledge makes it possible to forecast those suppliers, with their environmental impacts, that would be either included or excluded by the variation in energy consumption.

Network monitoring. It is therefore necessary to negotiate energy on the basis of market criteria, knowing the environmental contribution of the different suppliers. As in the case of the OEP, it is therefore necessary to know the environmental profiles and the efficiency of the individual suppliers.

Data availability. The CLCA can be used to estimate the impact produced in the short term. However, the needed data may only be available with a delay of up to 24 h. The lower the time between the negotiation and the actual dispatching of the energy, the more forecast systems tend to mirror real-time systems.

Robustness. Although short-term CLCA requires less assumptions than long-term CLCA, there are three factors that limit its accuracy even in the absence of significant hypotheses. The first factor concerns the actual production of energy by a committed unit. In fact, the CLCA does not consider the ancillary production of energy and the dissipation of the distribution network that depends on the location of the dispatching of the energy in the considered market day. The second factor is that the displaced value of the supplied energy has to be assessed in advance through prediction systems because the shifted demand implicitly requires a modification in the provider bid in the day-ahead market. Indeed, every flexible consumers' bidding strategy has to be taken before knowing the actual market equilibrium. The high variability of MEP between one hour and another, however, means that small inaccuracies in the identification of the supplier can produce consistent shifts in the impact assessment. Finally, since the market sorts suppliers by an economic merit criterion, substituting the marginal supplier does not assure to include producers related to better environmental impacts or to exclude those with worse ones.

4.5 Conclusions

The present work focused on the implementation of the LCA for the analysis of the electric networks in the short-term perspective. In general, two different approaches have been used: the attributional LCA in the case the grid consumer is unable to influence the network configuration and the consequential LCA in the case the consumer can modify the hourly offer curve that is based on the bid market. According to preliminary results, the hourly profile of impact categories is remarkably different depending on the used approaches. More in general, a supplier-led assessment approach is preferred for scenario development where the primary goal of the assessment is to inform the choice between options, such as short-term approach can provide comparable representations on the basis of effective changes in electric networks that are related to consumer choice. The same assessment can be both applied for ex-post analysis in order to identify real decisional outcomes and for ex-ante analysis in the perspective to attain accurate forecasting. The central point for further developments regards the efficiency of data exchange between stakeholders of electric network as required by a number of different research agendas. It is possible to assume that any further improvement will contribute to reduce uncertainty and to improve economic and environmental effectiveness of single stakeholder. As second result, CLCA on the short term can relocate endogenous assumptions by researchers into a set of verifiable data.

A second relevant point regards the need to define further optimization policies in an LCA-based information system. The case presented refers to the monitoring of the produced CO_2-eq by the electric network on the basis of consumption choices of a single player that acts on the electricity system and can influence the day-ahead market. It therefore becomes possible for the consumer to perform policies that balance both the price effects and the environmental effects by managing the consumption of energy in a given time. This approach can produce relevant results in case the single player is an industry that has significant energy consumption and can shift this consumption through the management of hourly production. In this specific case, the player should have the possibility to know in a limited time horizon the precise effect of operational choices that are related to energy consumption. Even in this case, however, it is necessary to define appropriately the company's environmental strategy and the resulting optimization system. For example, the values of environmental impact categories may vary in different ways throughout the day and the introduction of weight systems should translate the relevance that these categories have in the optimization strategy. The concurrent application of LCA-based tool to control the real emissions produced by a company can support energy management tools to control the actual impact of business decisions and compliance with related environmental targets.

Such information framework can prospectively represent a test-case area for Cyber-Physical Systems (CPS) application (Ballarino et al. 2017). These CPSs can in fact exchange real-time information on the energy attributes and real-time demand in a coordinated way. The user company should implement a system of devices that can record energy consumption from electricity and the CPS system should be able to estimate the energy consumed for the work cycle, and the energy required for subsequent scheduling through a time-machine system. Unified to market or to network manager, but also provide distributed data in order to transfer to series of real-time information regarding its expected offer and its efficiency.

Such evolution in electricity management can be inferred by considering the contribution that technologies and the market are producing in the actual configuration of the electricity market system. On the one hand, the opening of the market has produced an increasing growth in the percentage of distributed energy sources and with an increasing variability in the hourly supply. Second, market mechanisms tend to reward power plants that are able to produce large quantities of energy by flexibly respond to the demand. Such selection produces high dynamism in the configuration of the hourly suppliers and in their consequent environmental contribution. A switch to distributed renewables therefore implies a shifting of resource use and environmental impacts both spatially and temporally (e.g., GHG emissions arising "upfront" in the country of product manufacture, rather than during the operational life in the country of deployment), and potential reconfiguration throughout the electricity system. These dynamics pose a challenge for the accounting of real environmental effects from efficiency industry policies in relation to environmental goals when new energy supplier type will replace the current generation.

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Chapter 5
Small-Size Vanadium Redox Flow Batteries: An Environmental Sustainability Analysis via LCA

Pasqua L'Abbate, Michele Dassisti and Abdul G. Olabi

Abstract Electrical energy production from renewable sources has dramatically grown in the recent years in the developed countries, putting the hard problem to be solved of supply discontinuity. How to reach high efficiency and reliability of electrical energy storage system is thus now one of the most challenging goals to be reached: among all, one of the most simple and widespread to use is the electrochemical storage systems. This paper analyzes the sustainability of a small vanadium redox flow battery performed by an LCA approach. This electrical energy storage system was selected for its significant advantages in use, such as the almost infinite lifetime of the vanadium electrolytes, which represent a potentially significant advantage in terms of a sustainable future made of less fossil fuels and more renewable energy. In fact, the LCA analysis performed shows that the production of the battery has a moderate impact, including the effect toxicity while at the end of life, the material and the electrolyte are completely reusable with a small fraction that goes to landfill disposal.

Keywords Vanadium redox flow battery · LCA · Electrical energy storage
Renewable energy sources · Sustainability

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5.1 Introduction

Fossil fuels have dominated over two centuries, because of their large availability, their reduced storage volume, their high electrical energy concentration, and finally their electrical energy transferability (Dassisti et al. 2016). Electricity is a commodity representing about 12% of the total electrical energy transformed all over the world today. This percentage is keen to increase in the future, up to 34% in 2025, due to the present demand trends of global electrical energy consumption and also to the decline in the fossil fuels stock and the increased use of renewable energies in response to the global warming challenges (IEC 2012; Ibrahim et al. 2008). The contribution of electricity generated by renewable sources (wind, tide, sun, etc.) is destined to grow, according to the present trend of reduction of the greenhouse gas emission. Renewable sources are variable in time and space per sé, and this does not couple with the stability required for power supply. The most used means to face this problem is to adopt efficient and reliable electrical energy storage systems (EESS).

There are several electrical energy storage systems used so far, each based on different principles of transformation. Electricity can, in fact, be converted and stored as power potential (hydroelectric pumped, compressed air), kinetic energy, thermal energy, or chemical energy (generally hydrogen, methanol, synthetic natural gas, or electrochemical species) (Dell and Rand 2001). The most convenient means of storing electricity so far for today's applications, and for the new "green" future applications, is the use of electrochemical storage systems because of their limited dimensions and high specific electrical energy storage capacity with respect to other types of storages. There are different types of electrical energy storage systems on the market: this paper focuses on the vanadium redox flow batteries, originally developed by NASA in the early 1970s for long-term space missions (Giner 1976). These batteries are now driving attention for electrical energy storage because of their independence in electrical energy and power rating, fast response, room temperature operation, extremely long life, and potentially low environmental impact.

In particular, we focus on a specific case study of a small-scale vanadium redox flow battery (VRFB) prototype to give the flavor of the environmental sustainability through a life cycle assessment (LCA) analysis. The battery prototype was developed within an industry-funded research project aimed at optimizing VRFB for easy-to-mount civil applications. Environmental sustainability was evaluated to highlight the critical points of the design phase and potential improvements before the market entry. The LCA included also the preparation of the vanadium electrolyte, the core component of the battery. Three different processes were benchmarked: the mixing of suitable vanadium precursors (Electrolyte A), the chemical reduction of V_2O_5 by oxalic acid (Electrolyte B), and the electrochemical reduction of V_2O_5 using a homemade "H-shaped" electrolysis cell (Electrolyte C), The results of this analysis provide a benchmark reference for assessing environmental sustainability of existing storage systems in different applications.

5.1.1 Electrical Energy Storage Systems

Renewable energy sources are discontinuous per sé as affected by different meteorological or environmental conditions. This may represent a strong limitation in use due to the kind of electrical energy quality supply required so far. To partially cope with these limits, it may potentially be possible to balance the electrical energy supply network by increasing the amount of renewable generation plants installed, as well as to spread the installations of renewable generators on a larger area, or even to exploit the complementarity of several renewable sources (IEC 2012). These solutions, however, are possible upon large investments (say the number of plants and/or the improvement of the transmission networks) as well as the existence of international agreements between producers. Considering the cost of extra-renewable generation and the difficulty of building new production/transmission plants, the electrical energy storage systems (EESSs) are a promising alternative solution to this electricity storage problem. Electrical energy storage systems are, in general, devices that can store electrical energy over time before turning it into work. In case of electricity, EESSs are interposed between the place/time where electricity is generated and the place/time where it is consumed: their use is to supply the correct amount of electricity upon variable demand. They serve to recover imbalance between supply and demand, as well as to guarantee the stability and the quality of the power supply itself (voltage and frequency). EESS may also be used to reduce the cost of electricity by storing peak electricity when the price is lower, for use at peak prices at higher prices (Chen et al. 2009). EESSs furthermore support users when power failures occur and can be used in mobile applications within off-grid areas. Finally, for some applications in the transport sector, EESSs contribute to the creation of an ecological transport system by limiting the use of conventional combustion engines and increasing the use of electric vehicles with batteries.

5.1.2 Electrochemical Storage Systems

There are several electrical energy storage systems that convert and store electricity as power potential, kinetic energy, thermal energy, or chemical energy. The most common means of converting and storing electricity is the adoption of electrochemical storage systems. These systems, which are typically named batteries, have been diffused since 1890 with the lead–acid battery (used in mobile and stationary applications) having a life expectancy of 6–15 years, cycle efficiency of 80–90%, and easy recyclability and recharge. This type of batteries is cost-effective, but has low electrical energy density and, due to its hazard risk, its use is forbidden or restricted in different jurisdictions.

In 1915, nickel cadmium and nickel hydride (NiCd, NiMH) batteries were invented, with a higher power density than lead–acid batteries (Liyu et al. 2011), higher number of cycles, and able to operate at low temperatures (−20 °C up to

−40 °C). Due to the toxicity of cadmium, later the NiMHs were built, currently replaced in portable and mobile applications by lithium–ion batteries. Another type is the sodium–sulfur battery, which reaches life cycles of about 4500 cycles and has a discharge time of 6.0–7.2 h: to maintain operating temperatures, a heat source is needed, which uses the accumulated electrical energy, thus partly reducing battery performance. Nickel sodium chloride (NaNiCl) battery, better known as ZEBRA (Zero Emission Battery Research), is a high-temperature battery and has been marketed since 1995, successfully implemented in several electrical vehicle designs (Think City, Smart EV).

Lithium–ion batteries have then become the most important storage technology for portable and mobile applications since the beginning of 2000. Generally speaking, this kind of battery has a very high efficiency, typically in the range of 95–98%, is very flexible and has 5000–6000 duty cycles of lifetime. The main obstacle is the high cost (more than 500 €/kWh): they can only compete with lead–acid batteries for those applications which require short discharge times. Safety is a serious issue in lithium–ion battery technology. Most of the metal oxide electrodes are thermally unstable, and can decompose at elevated temperatures, releasing oxygen which can lead to a thermal runaway (IEC 2012).

Flow batteries are now receiving attention for their electrical energy conservation lasting hours or days with a power up to several MWs (Alotto et al. 2014; Dassisti et al. 2015). Flow batteries are classified as redox batteries and hybrid batteries. In a hybrid fluorescence battery (HFB), one of the active masses is stored internally within the electrochemical cell, while the other remains in the liquid electrolyte and is stored externally in a tank. Hybrid cells thus combine the features of conventional secondary batteries and redox flow batteries. The operating range of these batteries is between 5 and 40 °C; this is due to the solubility limit of V^{3+} in sulfuric acid below 5 °C and to the V^{5+} instability above 40 °C. Recently, researchers at the Department of Energy's Pacific Northwest National Laboratory (USA) found that the addition of hydrochloric acid to sulfuric acid, particularly 2.5 parts of sulfuric acid and 6 parts of acid hydrogen, increases the storage of batteries by increasing their electrical energy capacity by 70% and expanding the operating temperature range, i.e., between −5 and 50 °C (Li et al. 2011).

5.2 VRFB and Their Applications

A redox flow battery (RFB) refers to an electrochemical system that generates a so-called redox system on the surface of the inert electrodes, responsible for the conversion of electrochemical energy (Chuna et al. 2015). Several redox pairs have been studied and tested in RFBs, such as a FeTi system, a Fe–Cr system, and a polyS–Br system.

The first-generation vanadium redox battery (VRB) used sulfuric acid and vanadium species in both semiconductors as electrolytic solution. Vanadium in solution comes from the vanadium pentoxide compound (V_2O_5), which is found in minerals

Table 5.1 Existing types of batteries with vanadium electrolyte

Characteristic	V/V	V/Br	V/Air
Solution	Sulphuric acid	Bromidic acid	Sulphuric acid
Reaction	$VO_2^+ + 2H^+ + e^- \rightleftarrows VO^{2+} + H_2O$ $V^{2+} \rightleftarrows V^{3+} + e^-$	$2VBr_2 + 2Br^- \rightleftarrows 2VBr_3 + 2e^-$ $ClBr_2^- + 2e^- \rightleftarrows 2Br^- + Cl^-$	$4H^+ + O_2 + 4e^- \rightleftarrows 2H_2O$ $4V^{2+} \rightleftarrows 4V^{3+} + 4e^-$
Electrical energy density (Wh/kg)	25	50	41
Standard potential (V)	1.23	1.3	1.49

such as vanadinite and carnotite, present in countries such as Russia, South Africa, and China.

With the vanadium ions, also the bromium can be used, obtaining the pair (Br3−/3Br−). Another type is the vanadium/air battery, which is still in the experimental phase and uses the pair V^{2+}/V^{3+} in a semicircle and the other pair O_2/H_2O. In Table 5.1, the three typologies are given with reactions occurring within the cell and their respective energy density and standard potential (Tang et al. 2012).

The most widespread form of rechargeable vanadium battery uses vanadium redox pairs in both semiconductors. The electrolyte solution is stored in two separate tanks, and simultaneously pumped into the cells where the oxidation reaction occurs, which can lead to battery charge or discharge. The power of the battery depends on the size and number of electrochemical cells, while the capacity of the battery depends on the amount of electrolyte stored in the tanks. Oxidation reactions occurring within the cell are visible in Table 5.1. During the discharge process, a reduction of V^{5+} in V^{4+} occurs, with consequent acquisition of electrons and oxidation of V^{2+} in V^{3+} and the release of electrons. In the charging process, V^{4+} oxidation in V^{5+} occurs with consequent release of electrons and the V^{3+} reduction in V^{2+} with electron capture (Weber et al. 2011). Figure 5.1 shows a schematic diagram of a redox flow battery with electron transport in the circuit, ion transport in the electrolyte and across the membrane, active species crossover, and mass transport in the electrolyte.

Cell stacks and electrolyte tanks can be placed in distinct locations: consequently, storage media can be placed in places where storage containers do not affect the production space (e.g., under a floor or in parking facilities). Vanadium as an electrolyte, compared with other types of electrolytes, based on different redox pairs (iron/chromium, bromine/polysulphide, vanadium/bromine, zinc/bromine) (Bartolozzi 1989), has the unique feature of having the same metal ions in both positive and negative electrodes. In case of mixing of positive and negative electrolytes, the battery capacity does not diminish and does not suffer permanent capacity losses. Vanadium redox batteries can be upgraded at a relatively low incremental cost by increasing the volume of the electrolytes (using large tanks) to have more electrical energy stored. Adding new cell batteries allows to increase power (thus allowing quick supply through the exchange of solutions).

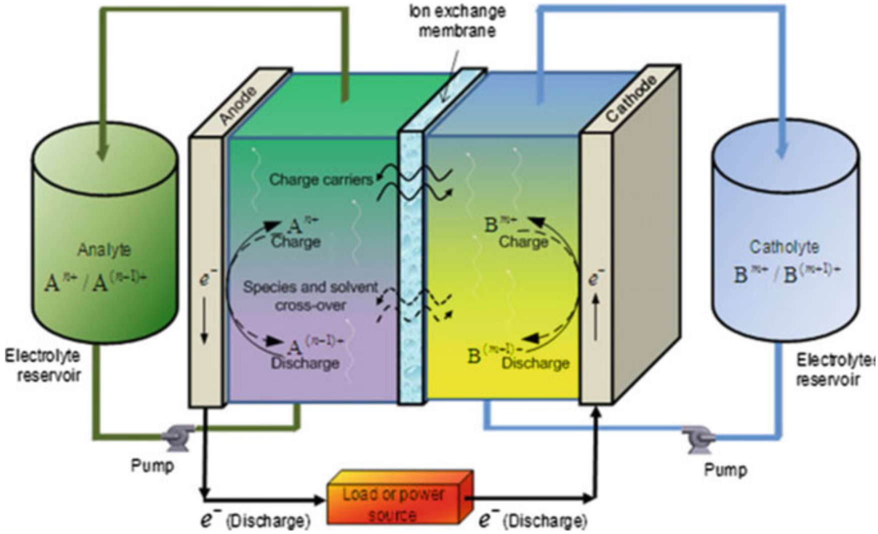

Fig. 5.1 A schematic diagram of a redox flow battery. *Source* Weber et al. (2011)

VRFB technology is most frequently used for renewable energy sources (Nehrir et al. 2011; Beaudin et al. 2010; Kear et al. 2012) as well as for large fixed electrical storage systems, where batteries need to be stored for long periods, with little maintenance while maintaining an almost ready state (Poullikkas 2013). VRFB is also suitable for a wide range of electrical energy storage applications in industry (Chen et al. 2009; Skyllas-Kazacos et al. 2010): say, in the telecommunications industry, as a backup unit in UPS systems, for increased security of supply and stabilization of renewable (Beaudin et al. 2010; Chen et al. 2009; Nehrir et al. 2011), etc. In Japan, for instance, several multi-MWh systems have been installed; one of these systems stores up to 500 kW for 10 h (5 MWh) (Beaudin et al. 2010).

One of the largest VRFB installations has been applied to stabilize a 32 MW wind farm to provide a maximum power of 4 MW/6 MW units in Tomamae Wind Villa in Japan (Chen et al. 2009). The Japan Institute of Energy (2001) and the Rice Research Institute in Denmark (2006) installed a battery to understand the potential of the VRB for everyday wind management. Vanteck (250 kW, 2 h) has also installed the first major commercial VRFB outside Japan to Eskom in South Africa (Nehrir et al. 2011).

5.3 LCA of a VRFB Small-Scale Prototype

This paragraph presents the LCA study for a real small-scale redox flow battery (VRFB) prototype following the ISO 14040 and ISO 14044 standards (ISO 2006a,

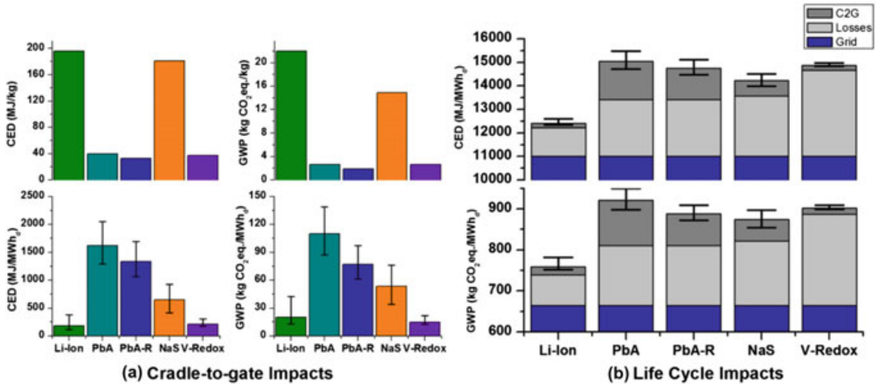

Fig. 5.2 Life cycle battery impacts showing the cradle-to-gate and life cycle stages impacts of batteries. *Source* Hiremath et al. (2015)

b). This battery prototype was developed within an industry-funded research project to optimize small-size VRFBs for several civil applications. Environmental sustainability has been addressed during design and experimentation, to evaluate critical points and support eco-design before entering the market. In Rydh (1999), comparisons are made up of the large vanadium redox (VRFB) stream battery for lead–acid batteries (PbA) for a Swedish scenario. A broader perspective is discussed in Rydh and Sandén (2005) where all the components of a battery photovoltaic system are faced. It is clearly stated that, when it comes to relatively new technologies, there are strong uncertainties about productive efficiency. This in turn can lead to inconsistent system boundaries. This question is clearly discussed in Pehnt (2006) which proposes a dynamic approach to LCA against the LCA state-quo; authors in Hiremath et al. (2015) present a comparative life cycle assessment of cumulative electrical energy demand (CED) and global warming potential (GWP) of four stationary battery technologies: lithium–ion, lead–acid, sodium–sulfur, and vanadium redox flow (see Fig. 5.2). In Arbabzadeh et al. (2015), a life cycle assessment (LCA) model is developed to determine the system configuration needed to achieve a variety of CO_2-eq emission targets and prove that adding VRFB as energy storage could be economically preferable in off-grid configuration only when wind curtailment exceeds 66% for the examined system. In Unterreiner et al. (2016), the ecological impact of recycling and reuse of materials of VRFB were compared with lead–acid, lithium–ion batteries proving that the Li–ion battery has the lowest ecological impact among the three battery technologies provided that for VRF batteries there is still no established recycling process up to date.

The environmental performance of the VRFB batteries was made not only on “cradle-to-gate analyses” but considering also their use stage impact. The proportion of cradle-to-gate impacts in the life cycle impacts of the batteries varies from around 2% (for Li–Ion and V-Redox) to 12% (for PbA) (Hiremath et al. 2015). In Fig. 5.2, it is clear how the manufacturing phase has only a minor impact on the life cycle impacts of the battery compared to its use phase. In the case of Li–Ion and V-Redox batteries, despite both of these are sustainable at the cradle-to-gate stage, the former performs better than the latter when use stage impacts are taken into account. The increasing competitiveness of Li–Ion battery, due to the increasing GHG emissions in the power-grid mix, is mainly due to the effect of its round-trip efficiency. That is, the higher the round-trip efficiency, the better the relative performance of that battery technology at higher environmental loads and vice versa.

5.4 LCA of Small-Scale Vanadium Redox Flow Battery Prototype

Provided there is no source of literature that addresses a small-scale VRFB battery, the aim of the present chapter is to give figures of the environmental sustainability of a small vanadium redox flow (VRFB) battery, to provide a reference benchmark of small VRFB not yet on the market, with a nominal power of 0.15 kW.

As concerns the LCI assumptions, reaction and mixing energy has been considered, expressed in terms of power and storage energy capacity to enable evaluation independently from sizing. In addition, all the raw material extraction and their production were taken into account. As concerns energy, input energy to the hydraulic system and the charge energy per cycle were assumed. For transportation and packaging, all transportations for raw materials were considered. Packaging was not considered, provided its impact was not significant on the overall life cycle. A fixed number of use and disposal were considered too. The hypothesis of a continuous running for 24 h/day over the period of 20 years was made, with an average energy delivery of 1.2 kWh/day for 20 years. Vanadium electrolyte is assumed to have a very long lasting life and its only treatments are filtering before reuse, provided it is self-recovery. The electrolyte with active material is thus assumed to last indefinitely. A deionized water refill was assumed of 100 ml in 20 years. The only consumables are the SPEEK ionic membranes, which were assumed to be replaced every 5 years. As concerns the hydraulic system, a 5-year maintenance is assumed with replacement of seals. At the end-of-life membranes are brought to landfill as well as the pumps. All the other materials have been considered as fully reusable.

System boundaries are given in Fig. 5.3. The functional unit is described in Table 5.2 since the same prototype is taken as reference. The unit processes considered are production of all raw materials of the parts to be assembled (electrodes and cells, membranes, laboratory-prepared vanadium electrolyte, battery cases, hydraulic system), assembly, use, and disposal. The VRFB was assembled and tested in Apulia (Southern Italy). All VRFB components came from southern Italy, except for electrolyte synthesis reagents from northern Italy and Vanadium pentoxide from South Africa. Most of the primary data were obtained by direct measurements in the laboratory where the battery was built and tested for operation and integrated with the literature. Secondary data were obtained with the Ecoinvent v 2.1 database

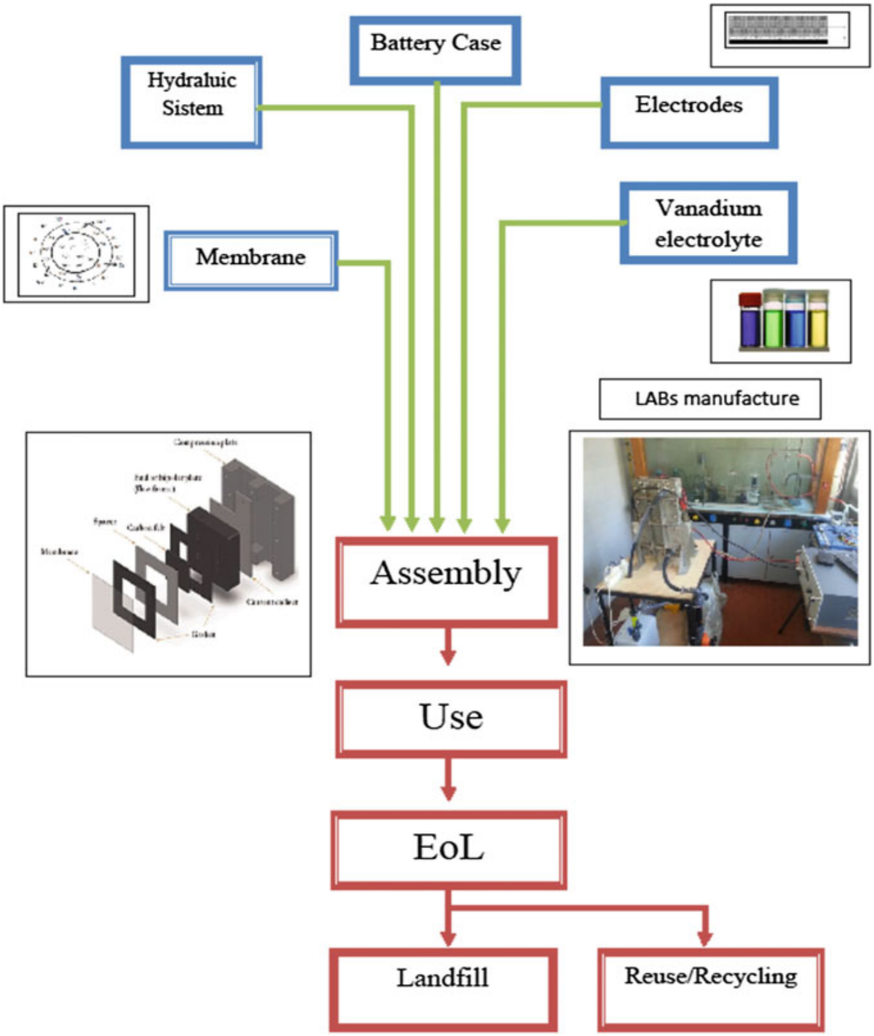

Fig. 5.3 VRFB prototype system boundaries

(Frischknecht et al. 2007; Jungbluth et al. 2008). The software adopted to perform the analysis was CMLCA developed by the Center of Environmental Science at Leiden University (Netherlands). The methods used for impact assessment are USE-tox™ (Rosenbaum et al. 2008), IMPACT2002+ (Jolliet et al. 2003) and ReCiPe 2008 (Goedkoop et al. 2009) for the benchmarking purposes above referred to as the environmental sustainability of EESSs.

The first two were selected considering the toxicity of the vanadium element, to investigate the toxicity of the organism and of humans, the latter being screened

Table 5.2 VRFB prototype analyzed

Characteristics	Value
Number of stack	1
Nominal power	150 W
# Cells/stack	5
Average voltage at end discharge (SoC = 0.2)	6 V
Energy density of electrolyte	36.18 Wh/l
Electrolyte volume	6 l
Overall efficiency	0.85
Average current	25 A
Charge energy	176.47 Wh
Discharge energy	127.5 Wh
Cycle time (charge and discharge)	~3 h

with particular attention to the noncarcinogenic and carcinogenic fraction. With the ReCiPe method, a combined use approach of midpoint impacts and environmental damage assessment (endpoint) is proposed. Subsequently, an LCA comparison was performed considering three different processes of vanadium electrolyte preparation in the laboratory.

The battery prototype considered was made up of polypropylene loaded with graphite, stainless steel plates, steel screws, and brass current collectors. Carbon felt GFD 4.6 EA was used for VRFB electrodes, coated with niobium to avoid hydrogen evolution. The most important property of the electrodes is that they have a large surface area in order to provide high current densities. The membrane adopted was an ion-exchange membrane (commercial sulfonate PEEK), separating the positive and the negative electrolyte solutions. Ion-exchange membrane must allow the ion transfer within the electrolyte while preventing electrons to pass through. Cells were made of high-density PP. Bipolar plates were made of SIGRACET—BPP. Electrolyte is stored in external tanks outside the cell stack. Tanks were made of plastic materials (PE) to resist the low pH environment. Pumps, valves, and piping components were also in plastic (PVC) resistant at low pH environments.

The electrolyte was obtained in the laboratory by a mixing process (method A). Required quantity of precursors V^{2+}/V^{3+} is dissolved in a solution of 1M H_2SO_4 + 2M HCl and mixed with magnetic stirrer for 3 h. Energy inputs include the energy used for the hydraulic system, the magnetic stirrer, the charge energy per cycle, the energy used for the processes of extraction, and production of reagents and materials. All transports of raw materials were considered, while the packaging was not considered as their impact was not significant for the life cycle calculation (0.05% contribution). For the use phase, it was assumed a continuous operation for 24 h/day over a period of 20 years, with an average energy delivery of 1.2 kWh/day. It is assumed that vanadium electrolyte is completely reused, by performing only a mechanical filtration process and adding deionized water (about 100 ml over 20 years). The only consumables are the SPEEK ionic membranes, which are supposed to be replaced every 5 years. Regarding the hydraulic system, it takes 5 years of maintenance to replace the gaskets. In the EoL phase, the membranes and gaskets are brought to the dump, while battery cases, electrodes, and electrolyte solution are reused/recycled.

Impact categories analyzed were (Goedkoop et al. 2009) agricultural land occupation [ALOP ($m^2 \times$ year)], natural land transformation [NLTP (m^2)], marine eutrophication [MEP (kg)], freshwater eutrophication [FEP (kg)], particulate matter formation [PMFP (kg)], marine ecotoxicity [METP100 (kg)], terrestrial acidification [TAP20 (kg)], terrestrial ecotoxicity [TETP100 (kg)], water depletion [WDP(m^3)], metal depletion [MDP (kg)], fossil depletion [FDP (kg)], photochemical oxidant formation [POFP (kg)], climate change [GWP20 (kg)], ionizing radiation [IRP_I (kg)], freshwater ecotoxicity [FETP100 (kg)], urban land occupation [ULOP ($m^2 \times$ year)], human toxicity [HTP100 (kg)], and ozone depletion [ODP inf, x (kg of ODS x and kg CFC-11 equivalents/kg)].

5.5 LCA Results for the VRFB Prototype

The results obtained are grouped into five graphs. The first graph (Fig. 5.4) shows categories of toxicity obtained using the method impact 2002+ and reporting impact categories: Human Health Photochemical Oxidation, Human Health Ionizing Radiation, Human Respiratory Health Effects, and Human Health Human Toxicity (see results in Table 5.3). From Table 5.3 and the graph in Fig. 5.4, it is clear that the total damage on human health is due to the category of human respiratory health effects. The main contribution is given by the use phase. The high values of the use phase originate by the use of fossil fuels for electricity employed at the start of the battery at each cycle. As for the assembly process, the widest contribution is given by the electrolyte production (75%). With the USEtox™ method, it is possible to distinguish between carcinogenic and noncarcinogenic human toxicity. The results in Table 5.4 and the graph in Fig. 5.5 confirm the most impactful from the point of view of toxicity due to the use phase. As concerns, the human toxicity balance is of 50.3% for noncarcinogenic and 49.7% for carcinogenic.

According to the method ReCiPe, Midpoint (I) was used to benchmark the impacts of the three processes abovementioned considering assembly, use, and disposal. The impact categories are reported with the results in Table 5.5 according to the impact categories mentioned above and the related measurement units.

Figure 5.6 shows the results of this analysis. The use phase has the stronger contribution to each category of environmental impact investigated. On the other hand, Fig. 5.7 shows how the production processes of the various components used to assemble the battery have a significant environmental burden. It is clear that the element having the greater effect on the environmental impact of the battery assembly phase is the electrolyte production.

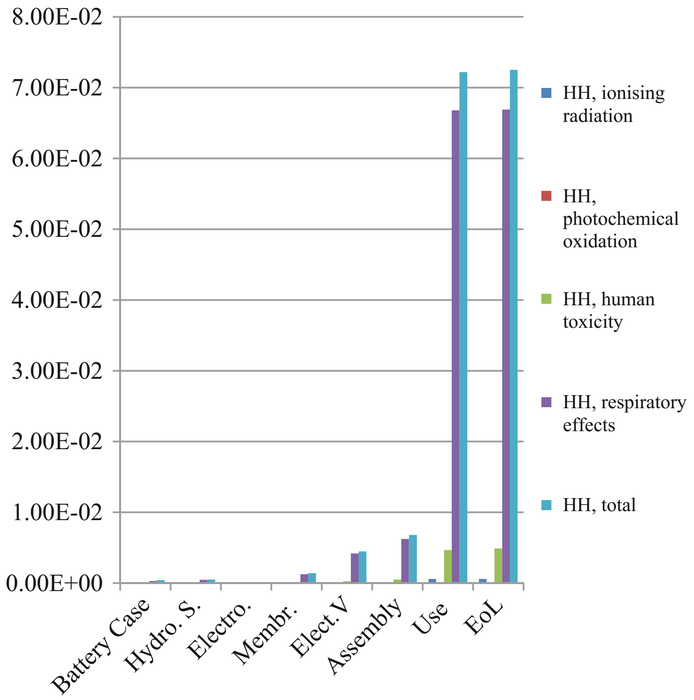

Fig. 5.4 Environmental impact assessment of the VRFB prototype using the impact 2002+, toxicity category in (DALY/kg emission) Jolliet et al. (2003)

Table 5.3 Results of environmental impact assessment of the VRFB prototype/method impact 2002+ (unit ecopoints)

Categories impact	Battery case	Hyd.S.	Electr.	Membrane	Electrolyte	Assemb.	USE	EoL
HH. ion. radiation	6.2E−07	3.1E−07	2.2E−08	1.7E−05	2.3E−05	4.2E−05	0.000593	0.000594
HH. pho.oxidation	2.63E−07	4.04E−06	8.56E−09	1.04E−06	6.08E−06	1.14E−05	6.94E−05	6.95E−05
HH. hum. toxicity	0.000115	3.47E−05	2.89E−07	0.000103	0.000255	0.000508	0.00467	0.0049
HH. resp. effects	0.000305	0.000464	3.70E−06	0.00127	0.00419	0.00624	0.0668	0.0669
HH. total	0.000421	0.000504	4.02E−06	0.0014	0.00448	0.0068	0.0722	0.0725

5.6 Comparison of Synthesis Processes for the Preparation of the Vanadium Electrolyte

From the results obtained, it has been found that the production of the vanadium electrolyte contributes more to the environmental impact of the components used to assemble the VRFB. With a second LCA study, three different syntheses of the

Table 5.4 Environmental impact assessment of VRFB using the USEtox™ (Rosenbaum et al. 2008)

Impact category	Elect.V	Assembly	Use	EoL	Unit
USEtox. human toxicity. carcinogenic	1.75E−06	3.92E−06	2.53E−05	2.67E−05	CTU
USEtox. human toxicity. non-carcinogenic	2.35E−06	6.89E−06	2.67E−05	2.70E−05	CTU
USEtox. human toxicity. total	4.10E−06	1.08E−05	5.40E−05	5.47E−05	CTU

Table 5.5 Results of environmental impact assessment of VRFB, using the ReCiPe midpoint (I), for the processes of assembly, USE, and EoL

IC	Assembly	USE	EoL	Unit
ALOP	0.51	5.51	0.02	m^2a
NLTP	0.02	0.195	0	m^2
MEP	87.7	1.09E+03	0	kg N-Eq
FEP	87.7	1.09E+03	0	kg P-Eq
PMFP	0.133	1.36	0.01	kg PM10-Eq
METP100	0.33	3.41	0.06	kg 1.4-DCB-Eq
TAP20	0.389	4.57	0.01	kg SO_2-Eq
TETP100	0.00877	0.0663	0.0003	kg 1.4-DCB-Eq
WDP	0.215	3.41	0	m^3
MDP	5.38	11.3	0.1	kg Fe-Eq
FDP	32	339	0	kg oil-Eq
POFP	0.331	2.81	0	kg NMVOC
GWP20	87.7	1.09E+03	0	kg CO_2-Eq
IRP_I	9.89	138	0	kg U235-Eq
FETP100	0.418	4.02	0.07	kg 1.4-DCB-Eq
ULOP	0.452	2.81	0.02	m^2a
HTP100	4.26	21.4	0.4	kg 1.4-DCB-Eq
ODPinf	8.89E−06	9.03E−05	0.00E+00	kg CFC-11-Eq

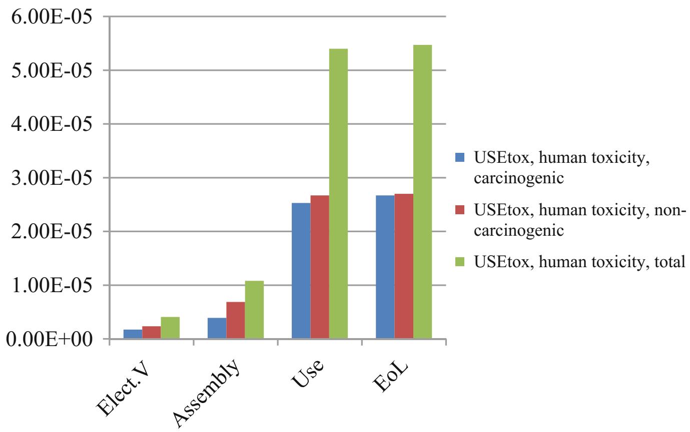

Fig. 5.5 Environmental impact assessment of the VRFB prototype using the USEtox™ Rosenbaum et al. (2008)

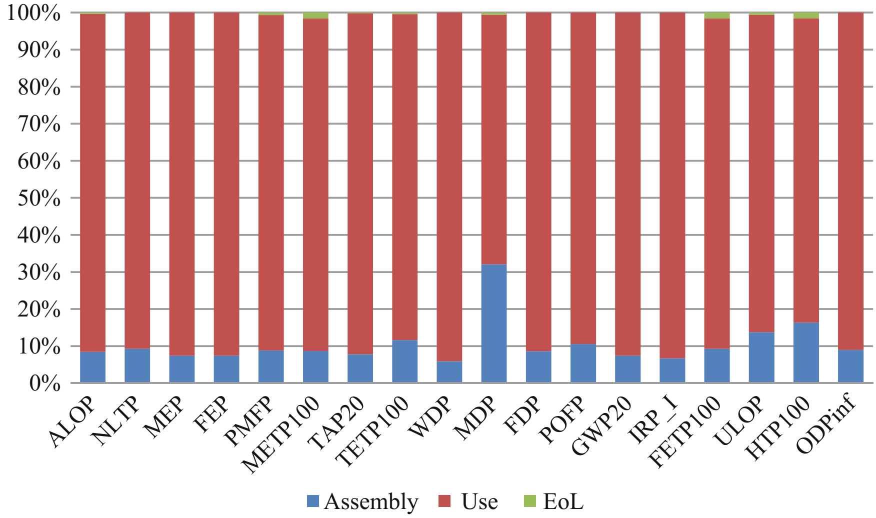

Fig. 5.6 Environmental impact assessment of VRFB, using the ReCiPe midpoint (I), for the processes of assembly, USE, and EoL

electrolyte conducted in the laboratory were compared. The selected functional unit (FU) is 6 L of electrolyte produced, and the system boundaries are from the "cradle to the door." The raw materials considered are the reagents used for the three syntheses, while consumables, laboratory glassware, and equipment used for the three processes (except electrodes and cells) have been excluded from the analysis. The transport of reagents was calculated as the distance from the place of purchase to the electrolyte preparation site. All reagents come from northern Italy with the exception

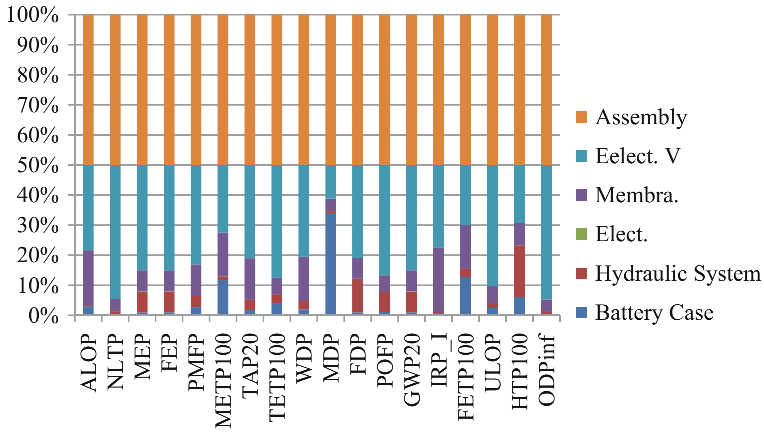

Fig. 5.7 Environmental impact assessment of VRFB using the ReCiPe midpoint (I) of production processes: electrolyte, membrane, hydraulic system, battery case, electrodes, and assembly

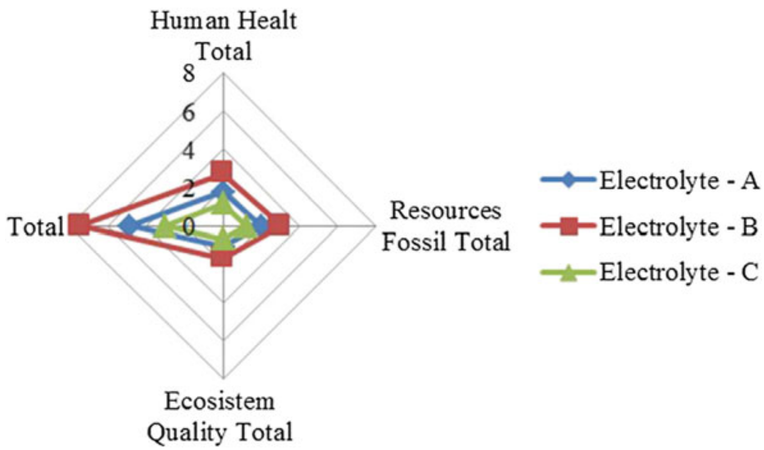

Fig. 5.8 Environmental impact assessment, using the ReCiPe endpoint (I/A), comparison of environmental damage (figures in ecopoints)

of deionized water (domestic production) and V_2O_5 (from South Africa). The energy used for the preparation of each reagent or raw material has been considered, as well as each energy input of the preprocessing phases of the electrolyte. No containers for packaging and storage of the finished product have been considered. ReCiPe 2008 (Goedkoop et al. 2009) is used as a method for measuring environmental impacts. In this case, three indicators of environmental damage were used: damage to human health, damage to ecosystems, and damage to the availability of resources.

Figure 5.8 shows that the results obtained by comparing the three different synthesis procedures of vanadium electrolyte are reported. The synthesis produced by

Electrolyte C is the lowest impact, while the Electrolyte B preparation process has the highest impact (Guinée 2002).

5.7 Conclusions

Renewable energy adoption is one of the viable strategies to respond effectively to the problem of global warming posing, on the other hand, the problem of adequate and reliable electrical energy storage systems. Among the existing ESSs, batteries have an important role: among the various types of batteries, the most interesting from a sustainability point of view are the vanadium redox flow batteries. This kind of battery still requires a large amount of space, while it is an environmentally sustainable battery, easy to regenerate, and recycle many of its components. In fact, the LCA analysis presented shows that the production of the battery has impacts that make this kind of battery viable for mass diffusion, including the effect toxicity, which is usually an important aspect for other types of existing types of batteries. At the end of life being the material and the electrolyte completely reusable, only a small fraction goes to landfill disposal. The improvement of sustainability of this kind of battery should then be concentrated on the use phase. In our case, the highest impact is due to the use of Italian power mix to operate the battery at each cycle over the 20 years lifetime considered. Surely using electrical energy from renewable sources can significantly reduce the resulting impact. The preparation of the vanadium electrolyte is a second critical point; from the comparison of the three sequences of electrolyte preparation processes (A, B, and C), we have identified the electrolyte preparation C as the best methodology in terms of environmental sustainability. Provided the electrolyte vanadium has been produced in the laboratory, in view of mass production, a significant improvement of the environmental sustainability is expected (Rydh and Sandén 2005). These results can be easily extended to other VRFB size units—provided the same technology and materials are adopted—thus allowing an easy benchmarking of the EESS applications.

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Part II
LCA Applied to Bio-energy: State of the Art and Case Studies

Chapter 6
Life Cycle Assessment of Renewable Energy Production from Biomass

Lucia Lijó, Sara González-García, Daniela Lovarelli, Maria Teresa Moreira, Gumersindo Feijoo and Jacopo Bacenetti

Abstract Among the different alternatives to conventional fossil fuels, the production of renewable energy from biomass (i.e. bioenergy) is regarded as an interesting option since it involves the valorisation of waste streams, residues and non-food crop biomass. Although a standardised framework regulates the Life Cycle Assessment (LCA) methodology, its application in practice poses some methodological difficulties. This chapter reviews the main methodological issues that a LCA practitioner has to face when it comes to the environmental assessment of bioenergy systems. Despite its complexity, consequential LCA is considered an interesting approach for informing policy-makers and decision-makers about the indirect effect of a specific strategy. In this sense, indirect environmental burdens such as indirect land use change should be included in the study. Moreover, the selection of the system function and system boundaries are other methodological issues that directly affect the results obtained and, therefore, the comparability of LCA studies, intensified in particular in the case of bioenergy systems due to their complexity. In more detail, some bioenergy systems co-produce multiple products, increasing the variability of

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the functions provided by the system, as well as of the system boundaries chosen to overcome multifunctionality (subdivision, system expansion or allocation). The selection of the appropriate methodology and impact categories, as well as the gaps in characterisation factors, is other methodological drawbacks.

Keywords Biomass · Life cycle assessment · Methodological issues Bioenergy systems

6.1 Introduction

6.1.1 Interest in Renewable Energy Production and Use

Energy is a potential indicator of economic and social development and improved quality of life (Ahiduzzaman and Sadrul Islam 2011). Currently, about 85% of the world's energy requirements are supplied by conventional fossil fuels (Srirangan et al. 2012). However, there are important issues regarding the sustainability of their use, including (i) depletion of fossil reserves, (ii) significant environmental impacts and (iii) large price fluctuations. Society's concerns about environmental and health issues arising from the use of fossil fuels has increased due to the increasing concentration of greenhouse gases (GHGs) such as CO_2, CH_4, CFCs, ozone, N_2O and halons in the atmosphere. The release of these gases derives mainly from human activities, which threaten not only environmental sustainability but also the socio economic situation, favouring global climate change and its related societal consequences (Berners-Lee et al. 2012). In this sense, there is abundant scientific evidence that changes in global climate are caused by anthropogenic activity. The effects of climate change are manifested even on a daily basis through the multiplication of extreme weather events such as heat waves, floods and droughts, the distribution of vector-borne diseases and their impact on disaster risk and malnutrition (Panwar et al. 2011). However, facing global climate change represents a great challenge (Ahiduzzaman and Sadrul Islam 2011). Effective measures to counteract the drivers of ongoing climate change and improve public response to its consequences are essential (Adamo 2015). Moreover, the concept of Green Economy has received increasing support from researchers and policy-makers. In this context, the development of renewable energy can reduce GHG emissions into the atmosphere, while contributing to solve other crucial challenges, such as improving the reliability of energy supply, saving fossil energy sources, securing local energy supply, creating 'green jobs' opportunities and ensuring sustainable development in rural areas (Panwar et al. 2011; Gasparatos et al. 2017).

With this aim, the European Commission published in 1997 'Energy for the future: Renewable sources of energy', a White Paper for a Community Strategy and Action Plan laying the foundations of the European Union (EU) policy on renewable energy (European Commission 1997). This document proposed to increase the share of renewable energy in the European gross energy consumption to 12% by 2010. Thereafter, the EU promoted the production of electricity from renewable energy sources under Directive 2001/77/EC (European Parliament 2001). In 2007, the European Commission proposed an integrated Energy and Climate Change programme, which included the commitment to achieve a reduction of at least 20% of GHG emissions by 2020 compared to 1990 levels. Subsequently, EU Directive 2009/28/EC set the target of achieving a 20% share of renewable energy in gross energy consumption and 10% of renewable energy in transport by 2020 (European Parliament 2009). To this purpose, each Member State has its own target for the share of energy from renewable sources and should have implemented a set of policies to achieve this objective. Therefore, the Member States had to prepare National Renewable Energy Action Plans with detailed roadmaps and measures to reach the 2020 renewable energy targets (Scarlat et al. 2015). Despite these activities, renewable energy currently accounts for a relatively small proportion of global final energy consumption (~19% of global primary energy); however, it has the potential to supply all human energy needs (Edenhofer et al. 2011; Ahiduzzaman and Sadrul Islam 2011). Recently, new targets for 2020–2030 have been introduced through the 2030 Framework for Climate and Energy. The targets are to achieve a 40% cut in GHG emissions compared to 1990 levels, at least 27% of renewable energy consumption and at least 27% energy savings compared to the business-as-usual scenario. The European Commission has therefore proposed specific policies to support the achievement of these targets, mainly by means of trading schemes and indicators for competitiveness and security of the energy system.

6.1.2 Renewable Energy Sources: The Potential of Biomass

According to the literature (Ellabban et al. 2014), renewable energy sources have the potential to supply the total present global energy needs. Among the different alternatives, the use of biomass for bioenergy production is considered one of the most promising sources (Cherubini and Strømman 2011) and its potential adds up to 20 times the current global energy requirements, being superior to hydroelectric, marine and geothermal energy.

The term biomass includes all organic material (containing residues) derived from crops, plants and trees and the biomass-based energy implies its conversion into heat, electricity and biofuels (Ellabban et al. 2014). Therefore, sources for bioenergy production can be very different and may include several different production processes, such as wood for thermal energy production or oilseeds production for oil extraction to produce biofuels. All biomass sources are regarded as an alternative capable of replacing fossil resources by producing different fuels and chemicals due to its carbon content. Biomass is synthesised through the photosynthetic process that converts atmospheric carbon dioxide and water into sugars, which are used by plants to produce complex materials, generically known as biomass. In bioenergy systems, it is important to ensure the supply of renewable, consistent and regular feedstock. In particular, feedstock can be sourced from different parts of plants, distinguishing the biomass-to-energy production in first, second and third generation. First-generation biofuels are those produced from dedicated crops that compete with food and feed production, resulting in multiple ethical, political and environmental concerns (Cherubini 2010). Second-generation biofuels come from raw materials based on waste, residues or biomass from non-food crops. They are considered a sustainable alternative to fossil fuels and to first-generation biofuels as well (Cherubini 2010). Third generation biofuels are derived from algae and microalgae cultivation.

Ensuring the environmental sustainability of biomass production is a crucial issue for the sustainable production of biofuels (Cherubini 2010). The Life Cycle Assessment (LCA) methodology has been widely used to compare the environmental impacts produced by fossil and renewable energy sources. With regard to climate change, the LCA studies available in the literature showed that bioenergy entails, in most cases, a reduction of GHG emissions. However, these benefits are not unambiguously obtained in all the environmental impact categories studied. Biomass production has been identified as an important source of other environmental impacts, affecting impact categories such as land use, acidification, eutrophication and ecotoxicity, among others. Additionally, large-scale cultivation of dedicated biomass (i.e. energy crops) could affect bioenergy potential, global food prices and water scarcity. To ensure the sustainable development of bioenergy, integrated policies for energy, land use and water management are needed, along with international cooperation, regulations, certification mechanisms and sustainability criteria (Popp et al. 2014). In this sense, a strategy has been proposed based on the development of biorefinery and biotransformation technologies to transform biomass feedstock into clean forms of energy (An et al. 2011; Kamm and Kamm 2004; Srirangan et al. 2012; Volsky and Smithhart 2011).

6.1.3 Environmental Aspects Linked to Bioenergy

It is generally believed that the use of renewable energies contributes to mitigating the environmental impacts associated with the use of fossil fuels. When biomass is burned or used after conversion into other biofuels (e.g. biodiesel, ethanol, biogas), its carbon content is released into the atmosphere as CO_2, which had been previously captured by the plant in the photosynthetic process. Therefore, biomass-based energy is considered carbon neutral. In addition, the use of biomass reduces NO_X and SO_X emissions into the environment (compared to the use of conventional fossil fuels) as it contains less nitrogen and sulphur than, for example, coal (Herbert and Krishnan 2016).

However, numerous studies indicate that biomass energy is not entirely clean (Field et al. 2008; Rahman et al. 2013; Herbert and Krishnan 2016). Increased production of biomass for renewable energy has the potential to offset fossil fuels requirements, but negative aspects can also be identified that threaten ecosystem conservation and diminish food/feed security (Field et al. 2008).

The production of biomass-based energy always involves the indirect use of fossil energy for the cultivation, transport or manufacturing phases of the process. Cultivation of bioenergy crops could hypothetically damage the environment due to agricultural practices and land and water degradation (Bindraban et al. 2009; Herbert and Krishnan 2016). Biodiversity loss, water harvesting, reduced soil productivity, introduction of invasive energy crops, use of agrochemicals (e.g. pesticides, herbicides and fertilisers) and their derived effects on the aquatic environment, as well as air emissions associated with NO_X, SO_2, NH_3, N_2O should not be omitted. In this sense, the use of non-food biomass produced on marginal land is considered as a potential sustainable option (Bindraban et al. 2009). Significant amounts of biomass could be produced in the short term without displacing food crops. It is therefore justified that we need to address the environmental impacts associated with biomass production, including background processes.

In addition, other negative aspects are also linked to the production and use of biomass. Managing biomass for energy production requires a large amount of storage space, as well as land and water. Soil erosion reduces soil productivity due to agriculture activities (Pimentel 2001), which contributes to water and nutrients run off and subsequently, eutrophication. In addition, changes in soil carbon content resulting from some agricultural activities that can lead to deforestation problems are well known. In this sense, the type of crop may behave differently. According to the literature (Field et al. 2008), management of agricultural land declassified with perennial grassland can increase carbon content, mainly due to the inputs to the soil, including roots and leaf litter.

Therefore, promoting biomass-based renewable energy requires knowledge of the risks (e.g. food security, soil degradation) and opportunities (increasing energy independence, improving rural economies and offsetting climate change) in the area. The modernisation of biomass conversion technologies, together with more efficient biomass production and conversion routes, are challenges to be undertaken. In addition, it is essential to promote standards, practices and regulations to protect the environment.

6.2 Key Methodological Aspects in LCA of Biomass-Based Energy System

LCA has evolved from its origins in the early 1970s into a complex tool that is now being widely applied in research, industry, policy and standards and continues to expand as it is able to determine the environmental impact of products or systems (McManus and Taylor 2015). Renewable energy policies are increasingly considering LCA as the driving tool for selecting the most adequate bioenergy pathways and guiding decision-makers (European Commission 2014). As part of the EU sustainability framework for biofuels and bioliquids, the EU Renewable Energy Directive (RED) (2009/28/EC) and the Fuel Quality Directive (FQD) (2009/30/EC) contain minimum GHG emission requirements that biofuels must meet on a mandatory basis in order to obtain public funding (Edwards et al. 2017). In more detail, GHGs must be reduced by 35% compared with fossil fuels in installations built before October 2015, while the threshold is raised to 50% for installations working from 2017 (Edwards et al. 2017). In order to standardise the quantification of these environmental impacts, the International Organisation for Standardisation (ISO) reissued a regulatory framework for LCA studies during the period 1997–2000. Updates to these documents were completed in 2006, so that the previous standards were combined in ISO 14040 (2006) and ISO 14044 (2006). However, in some aspects, these standards are rather generic, leading to some difficulties in the practical application of this tool, especially when associated with the assessment of complex processes such as bioenergy systems. Among the studies on bioenergy available in the literature, there are often important differences in results, not only due to different approaches to biomass production, conversion technologies and end-use options, but also due to the definition of system boundaries, functional units, allocation methods, assumptions in the building of life cycle inventory (LCI), the fossil energy reference system, etc. (Cherubini et al. 2009), which makes it difficult to compare the different studies.

6.2.1 Life Cycle Model

Traditionally, the life cycle model of LCA studies included all the processes that are identified to make a significant contribution to the supply chain of the system, known as attributional LCA (aLCA). aLCA describes the potential environmental impacts that can be directly attributed to a process or product throughout its life cycle, assuming that it is embedded into a static technosphere (Wolf et al. 2010).

With regard to biomass-based energy production, the current trend is to move to life cycle assessments in more complex decision-making contexts that describe how environmental impacts could change in response to possible policy decisions. This approach, named consequential LCA (cLCA), integrates the supply chain as theoretically expected from the consequence of the decision taken, including the changes resulting from the interaction between the system and markets (Wolf et al. 2010). Hence, this model does not reflect the actual or estimated supply chain, but rather models a hypothetical generic supply chain through market-mechanisms and potentially includes policy interactions and changes in consumer behaviour.

Compared to conventional aLCA, the consequential approach has proven to be particularly interesting in informing policy and decision-makers about the indirect effects of a specific strategy (Vázquez-Rowe et al. 2014). However, this perspective adds great complexity to the LCA study, as it often means that it should include additional economic aspects such as marginal costs and market effects (McManus and Taylor 2015). Moreover, a consistent approach for cLCA has not yet been established by LCA practitioners. The coexistence of these two different approaches with markedly different perspectives is one of the greatest challenges faced by the LCA community (Zamagni et al. 2012; McManus and Taylor 2015). The potential application of cLCA to the different processes for the production of renewable energy from biomass is particularly interesting, even if biomass waste is used as feedstock, due to

- The multifunctionality of these systems since the production of co-products and by-products (e.g. heat and digestate in the biogas-to-electricity production process; press cake or glycerol in the production of first-generation biofuels) adds complexity to the analysis;
- The presence of a framework of public subsidies for the energy produced, since the environmental consequences due to changes in subsidies could be evaluated;
- The reduction of the carbon storage in soil derived from the removal of biomass residues from agricultural or forestry land to produce bioenergy (Cherubini et al. 2009);
- The expansion of land use due to the creation of a market for biomass residues or by-products (Cherubini et al. 2009).

According to Ahlgren et al. (2015), the choice of the LCA approach (aLCA or cLCA) is closely related to the question of research. Despite the assertions made in the ILCD Handbook (JRC 2010), the choice of aLCA or cLCA is not always straightforward. In particular, careful considerations should be given to every study whether the methodological choices that have been made supply meaningful answers to the research questions.

6.2.2 Function of the System and Functional Unit

In every LCA study, the selection of the function of the system is an important methodological step, as it is directly related to the identification of the functional unit (FU) and to the delimitation of the system boundaries. The FU is a key aspect when comparing biofuels with the appropriate fossil reference, since it should guarantee that both systems provide the same service (Cherubini and Strømman 2011).

The selection of the FU in LCA studies focusing on renewable energy systems is not trivial; it is important to note that it is closely related to the objective of the study. Among the different LCA studies of bioenergy systems, there is a remarkable variability in the FU used, making it difficult to compare the results obtained in these studies. In fact, depending on the objective, more than one FU may be appropriate (Cherubini and Strømman 2011). According to the literature (Ahlgren et al. 2015; Cherubini and Strømman 2011), it can be identified:

- The input-based FU such as 'feedstock use', suitable for determining the best use of land or biomass, which allows comparison between different uses for a given feedstock. This approach is particularly interesting for first-generation crops since it allows the quantification of land use efficiency (Cherubini 2010). For example, González-García et al. (2013) considered 1 tonne of dry biomass for the environmental comparison of the production of three different energy crops (maize, triticale and wheat) cultivated for biogas production in Italy;

- The output-based FU, such as energy production, which identifies the best way to supply a product such as biogas or electricity from different feedstocks. For instance, for the comparison of two different feedstocks for biodiesel production in Italy, Bacenetti et al. (2017) considered 1 GJ of energy contained in each biofuel, while Carneiro et al. (2017) suggest using 1 MJ of energy contained in ethanol;
- The function-based FU, which represents the best choice according to the ISO standard. For example, in biofuels studies, if the aim is to compare different fuels, travelled distance may be a good option, as fuels have different engine conversion efficiencies. A simple comparison with 1 MJ of fuels would not reflect the diversity of the fuels. As an example, González-García et al. (2012) considered a distance of 1 km driven by a mid-size flexi-fuel vehicle that uses an 85% blend of ethanol and gasoline for the environmental comparison of ethanol production from different fast-growing wood crops.

6.2.3 Managing Multifunctionality

The problem becomes more complex as some bioenergy systems have more than one function and provide more than one product or service. In this sense, the one selected to define the FU depends on the goal and scope of the study. As regards the production of biodiesel from oil crops such as soybean, sunflower and rapeseed, the extraction of crude vegetable oil also includes the production of press cake, while during transesterification, in addition to biofuels, glycerol is also produced. In addition, the conversion of wood biomass into electricity by means of Organic Rankine Cycle (ORC) or by gasification and pyrolysis coupled to a Combined Heat and Power (CHP) plant involves the co-production of heat (Wolf et al. 2016). Another example in renewable energy systems is anaerobic digestion of biomass for biogas production. These systems involve the co-production of different products, both from the anaerobic digestion process and the end-use of the biogas produced. Moreover, this is especially relevant when anaerobic digestion is selected as a treatment option for organic waste management, as resource recovery from waste results in cost-effective multifunctional systems (Heimersson et al. 2017). In more detail, in these systems the main function may be the treatment of organic waste, while other secondary functions can be identified, such as: (i) the production of biogas, (ii) the production of electricity, heat and/or biomethane from biogas and (iii) the production of digestate to be used as organic fertiliser. Therefore, the way these multiple products are considered in LCA studies is becoming increasingly important (Heimersson et al. 2017). Different approaches to solving multifunctionality have been proposed. The choice of the most appropriate one depends, among others, on the goal of the study, available data and information, and the characteristics of the multifunctional process or product (Wolf et al. 2010). In detail:

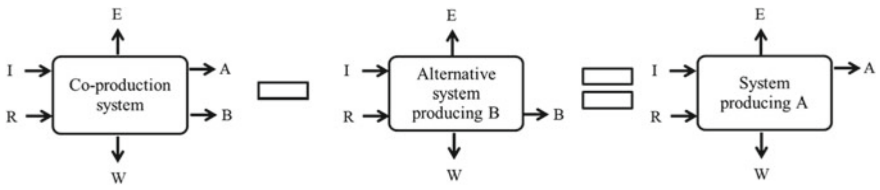

Fig. 6.1 System expansion. Acronyms: I—inputs, R—resources, E—emissions, W—wastes, A—product A, B—product B

- Subdivision refers to the collection of data individually for several mono-functional processes that are components of the multifunctional process and give rise to the production of the product under study;
- System expansion includes two options: (i) adding another function to make the system comparable (i.e. system expansion in the stricter sense) or (ii) extracting the non-required function of the system by subtracting the processes that provide an equivalent function (i.e. substitution by system expansion). System expansion should only be applied if a direct substitution effect can be robustly modelled. The Renewable Energy Directive (European Commission 2009) requires allocation by partitioning, based on the lower heating value (LHV) of the products, with the exception of excess electricity, which is addressed by system expansion;
- Allocation solves multifunctionality by partitioning the flows of individual inputs and outputs between the co-products according to certain criteria. According to ISO 14044, allocation should be avoided whenever possible by applying subdivision or system expansion (ISO 14044 2006). When unavoidable, the inputs and outputs of the system should be partitioned between its different products or functions in such a way that reflects a relationship between them, either physical, economic, energetic or exergetic.

In these complex systems, subdivision cannot be conducted since it is not possible to inventory the system in such detail that it allows each flow to be linked to each product (Heimersson et al. 2017). The use of substitution to avoid allocation is consistent with the recommendations of ISO 14044 and the International Reference Life Cycle Data (ILCD) Handbook (ISO 14044 2006; Wolf et al. 2010). This can be done by giving to the system a credit for secondary functions, awarding the system with the avoided negative impacts of the avoided product or service that the secondary functions replace, as shown in Fig. 6.1.

This perspective was followed, for example, in the study performed by Lijó et al. (2017) to manage the multifunctionality of a biogas system. In this case, anaerobic digestion is a co-production system in which biogas (product A) and digestate (product B) are co-produced. As shown in Fig. 6.1, given that biogas was identified as the main product, the burdens of an alternative system providing the same function as the digestate (i.e. fertilisation of agricultural soil) were subtracted. However, this alternative adds uncertainty to the results obtained due to the lack of primary data for the replaced processes. Moreover, this approach should be considered carefully

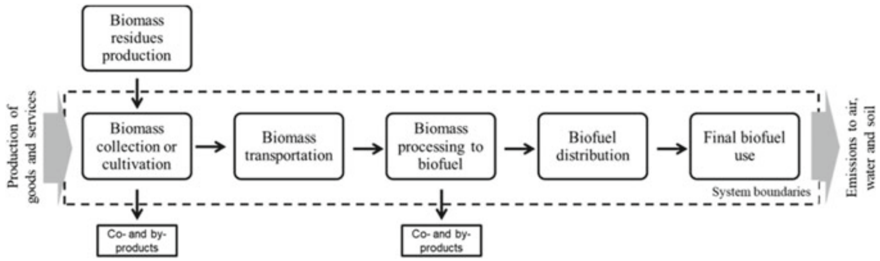

Fig. 6.2 Life cycle chain and system boundaries of bioenergy systems

since biogas systems using agricultural feedstock (e.g., cereal silages), digestate can be used as an organic fertiliser for the production of the energy crop; therefore, it does not leave the system boundaries. Moreover, if biogas is produced from animal manure, the digestate may play the same role as animal waste if it has not been diverted to anaerobic digestion. Fernández-Tirado et al. (2016) first performed a system expansion to address multifunctionality due to biodiesel and press cake co-production; however, subsequently, economic allocation between oils (for biodiesel) and meals co-produced was also considered. Similarly, Bacenetti et al. (2017) performed an economic allocation between crude vegetable oil and press cake, as well as between biodiesel and glycerol in the LCA of two biodiesel systems. Moreover, Wardenaar et al. (2012) conducted an analysis considering different allocation methods (economic, physical and substitution).

6.2.4 System Boundaries

The system boundaries define which unit processes belong to the analysed system, since they are necessary to provide the function to the system. Therefore, the system boundaries separate the analysed system from the rest of the technosphere and nature, defining where the system exchanges elementary flows with nature, and, therefore, produces the environmental impacts (Wolf et al. 2010).

Figure 6.2 depicts the processes included within the system boundaries when the overall life cycle chain of bioenergy system is under study. However, depending on the goal of each LCA study, the system boundaries of the bioenergy chain may change. For example, for the comparison of different feedstock for biodiesel production, a cradle-to-gate approach may be considered, i.e. considering the background and foreground processes up to the gate of the biodiesel factory (Fernández-Tirado et al. 2016; Bacenetti et al. 2017).

By maintaining the example of biogas production, the definition of the system boundaries of some systems for the anaerobic digestion of organic waste streams may be conflicting. In more detail, the distribution of the burdens related to waste production between the production system and the treatment system may be problematic (Doka 2007). The question is whether waste can be regarded as a valuable product, as it can produce biogas or a waste material that needs to be managed. Organic waste is considered as a zero-value product because biogas plants do not usually have to pay for it. In these cases, it is common practice in LCA studies to consider that the production of organic waste such as manure, food and industrial wastes are excluded from the system boundaries of the biogas system, as they are considered to be waste streams from other production systems (i.e. livestock and food sectors).

Broadly speaking, although its use in biogas plants is an option for its management, the production of this waste would not be influenced by a change in the biogas management scheme. A similar analysis can be applied to the digestate fraction from anaerobic digestion. Farmers who use the produced digestate as an organic fertiliser do not usually have to pay for it; therefore, it can be considered as a waste and the environmental impacts of its handling should be allocated to the anaerobic digestion process. This poses a major problem when applying digestate to agricultural land; while emissions of mineral fertilisers or animal manure are fully attributed to the agricultural production, emissions from the digestate would be allocated to the biogas system. This would lead to questionable conclusions from the LCA studies. According to Doka (2007), there are good reasons to include the application of digestate as a waste in LCA of biogas systems, there are also equally justified reasons to set the cut-off limit that includes the application of digestate in agriculture as a recycled material.

Unlike animal manure in the biogas process, for the wood-to-energy production chain, the use of lignocellulosic matrices as feedstock for energy purposes cannot be managed with the zero-value approach. When biomass such as pruning residues, leaves or branches from forestry utilisation are used as feedstock for energy plants, their production is included in the system boundary and, usually, allocation is performed (Muench and Guenther 2013; Patel et al. 2016).

In order to quantify the environmental gains of bioenergy production, a fossil-based reference system is required. Therefore, its definition can also play a key role in the outcomes obtained for a specific study. The reference system should reflect the most representative conventional way of providing the same function as the bioenergy system under study for a specific geographical location. When analysing the production of electricity from biogas, the reference fossil-based electricity system can be produced from coal, oil or natural gas, which entails different potential environmental burdens used as reference (Cherubini and Strømman 2011). For example, Van Stappen et al. (2016) displaced electricity from oil and natural gas, while Lijó et al. (2017) considered the average electricity production in the country mix of the site under study.

Following the issue of system boundaries, the consideration (or not) of biogenic carbon within the boundaries of LCA studies also deserves special mention. This biogenic carbon is temporarily stored in vegetation, litter, dead wood and soil (Cherubini and Strømman 2011). Therefore, this consideration is particularly important when dealing with bioenergy systems, since they use biomass as feedstock, whether energy crops, by-products or organic waste, which can be considered a temporary carbon storage. Biogenic carbon is defined as the carbon contained or derived from biomass that accumulates during plant growth as a result of photosynthesis (Wiloso et al. 2012). Conventionally, LCA studies do not assign any environmental burden to carbon dioxide emissions from biogenic sources (Brandão et al. 2013). In these cases, carbon neutrality is considered on the basis that the expected uptake of carbon dioxide from biomass growth equals the expected carbon emitted over the full life cycle, whether it is naturally decomposed or burned (Wiloso et al. 2012). Therefore, it is considered that there is no net increase in atmospheric carbon dioxide content and the benefits of temporarily removing it from the atmosphere and the impacts related to its subsequent emission are excluded from many LCA studies (Brandão et al. 2013; Wiloso et al. 2012).

However, with the aim of validating this assumption, the previously harvested biomass should be replaced by a new biomass growing in the short term. In this sense, the use of annual crops may not increase the amount of atmospheric carbon due to compensation by the relatively undelayed photosynthesis (Wiloso et al. 2012). Many authors disagree with this statement. Carbon sequestration during biomass growth can be considered as a negative emission in LCA. The argument to support this approach is that during the time between biomass harvesting and its decomposition or combustion, the concentration of carbon dioxide in the atmosphere decreases temporarily and radiative forcing is partially avoided.

Other authors support the idea that temporary storage of biogenic carbon may have a negative effect due to the change in the concentration gradient between atmosphere and oceans, causing oceans to absorb less carbon dioxide (Wiloso et al. 2012). The consideration of temporary carbon storage and delayed emissions within the system boundaries of LCA studies is discouraged by the ILCD Handbook, unless the goal of the study clearly includes it (Wolf et al. 2010). In any case, the inclusion of biogenic carbon within the system boundaries in LCA studies is still under discussion. Timing of emissions is usually not included in LCA of renewable energy. However, according to Ahlgren et al. (2015), when there are significant differences in time between CO_2 uptake and emissions from the system under study, this should not be ignored and discussed in the study, promoting efforts to quantify the impact.

However, biogenic carbon in biomass is not the only one to be considered; changes in biogenic carbon contained in the soil should be taken into consideration. By changing the way land is used, named as land use change (LUC), these storage pools may change until they reach a new equilibrium (

Cherubini 2010). The consideration of these carbon changes is directly related with cLCA studies (Carneiro et al. 2017) and has an important impact on the carbon balance of bioenergy systems due to the large quantities of carbon in soil. Therefore, emissions due to land use change may reduce GHG savings from bioenergy systems when comparing to fossil-based alternatives, especially when considering dedicated energy crops or agricultural and forestry residues as raw material (Cherubini 2010). Direct land use change (dLUC) occurs when the use of land is changed to produce energy crops for bioenergy purposes, displacing previous land use. Depending on the earlier use of the land and the energy crop to be established, carbon stock in soil can increase or decrease (Cherubini 2010). For example, if a forest land is converted into palm plantations, there would be a loss of carbon stocks; while, when the abandoned land is converted into sustainable maize cultivation, carbon stock may increase (Cherubini and Strømman 2011). Indirect land use change (iLUC) (or leakage) occurs when land currently used for feed or food crops is transformed into the production of feedstock for bioenergy and the demand for the previous land use (i.e. feed, food) remains, the displaced agricultural production will be shifted to other places where unfavourable land use change may occur. For instance, Buchspies and Kaltschmitt (2018) analysed different first and second-generation ethanol production, considering the mechanisms of LUC linked to straw removal in Germany, transformation from scrubland to soybean cultivation in Brazil and decrease in forest cover due to oil palm cultivation in Indonesia and Malaysia.

6.2.5 Building the Life Cycle Inventory

In LCA studies, the LCI is built by collecting data for each unit process defined at the system boundary and it is expressed on the basis of the FU selected. Collected data would include energy inputs, raw material inputs, ancillary inputs, other physical inputs, products, co-products and waste, emissions to air, discharges to water and soil and other environmental aspects. Two types of data can be distinguished: i) foreground data that refer to the process data required to produce the product under study and ii) background data that include data from processes required to produce generic materials, energy, transport and waste management. In accordance with ISO 14044, data quality requirements shall be specified to ensure compliance with the goal and scope of the LCA. It should include time-related coverage (data age and the minimum period of time for data collection), geographical coverage (area from which data should be collected for each unit process), technology coverage, accuracy (measurement of variability in data values for each process), completeness (percentage of flow that is measured or estimated), consistency (assessment of whether the methodology for data collection is uniformly applied to the data collection process), reproducibility (assessment of the extent to which information on the methodology and data values would allow an independent practitioner to reproduce the study), data sources and uncertainty of the information.

Specifically, regarding LCA studies of bioenergy systems, the calculation of different foreground data is required in different unit processes and at different stages of the life cycle, especially data related to direct emissions, such as those from storage or application of digestate, since they are not usually measured due to their difficulty. Estimating this type of data is a crucial issue in biogas LCA studies because they play an important role in the environmental outcomes. Therefore, to consider these emissions within the system, they are usually estimated using the methodologies available in the literature. Numerous studies have shown that the most important hotspots in biogas systems are associated with emissions, especially in the eutrophication and acidification impact categories (Lijó et al. 2017). However, there are several different methodologies and there is no a general consensus on which of them should be selected. And, therefore, in the literature, these emissions are generally estimated using different methodologies (De Vries et al. 2012; Fantin et al. 2015). These differences translate into disparities in LCA studies. For example, Bacenetti et al. (2017) used the Estimation of Fertilisers Emissions-Software (EFE-So) model to calculate emissions from the fertilisation of two non-conventional oilseed crops for biodiesel production in Italy. Moreover, Lijó et al. (2017) and Fantin et al. (2015) conducted a sensitivity analysis to quantify the effect on the results obtained on different methodologies.

Following the topic of data quality, uncertainty analyses are also recommended. In LCA literature, Monte Carlo simulations are the most commonly used methodologies, for example in Van Stappen et al. (2016) or Fantin et al. (2015); however, this assessment is not performed in most cases.

6.2.6 Selecting the Life Cycle Impact Assessment Methodologies and Impact Categories

According to the ILCD Handbook, the selection of the impact categories and characterisation models should be internationally accepted (Hauschild et al. 2011). In addition, the category indicators shall include those relevant for the specific study performed in accordance with the goal and scope, as well as to the results of the LCI. The characterisation model for each category indicator shall be scientifically and technically valid. Moreover, all the characterisation factors should not have significant gaps in the coverage of the impact category to which they relate. For the selection of relevant impact categories, initial knowledge based on experience gained from studies of similar systems may help to identify which impact categories are of significant global relevance and which may seem irrelevant to a specific system (Hauschild et al. 2011). In previous reviews focused on renewable energy production from biomass (Von Blottnitz and Currain 2007; Hijazi et al. 2015; Bacenetti et al. 2016), a great variation in the number of impacts considered and the Life Cycle Impact Assessment (LCIA) methods used to estimate them was found. In a review focused on agricultural biogas production carried out by Bacenetti et al. (2016) considering 105 studies, the number of impact categories ranged from 1 to 18 and LCIA method used included almost all known LCIA methods, such as EcoIndicator 99, CML 2001, Impact 2002+, ReCiPe and ILCD methods. Undoubtedly, the carbon footprint (also called global warming potential or climate change) was the most widely used environmental indicator. Nevertheless, different assessment methods were also used for this impact category: IPCC (2007), RED (European Union 2009) as well as the standard ISO/TS 14067:2013 (ISO, 2013). Moreover, tackling the carbon footprint alone offers a very limited version of the overall environmental performance of a bioenergy system. Regarding this issue, Venkatesh and Elmi (2013) criticised the importance of focusing on climate change and noted the importance of avoiding problem shifting; i.e. reducing the environmental impacts produced in climate change by increasing them to other impact categories, including acidification or eutrophication categories.

6.3 Conclusions

The LCA has been widely applied to assess the environmental impacts of renewable energy production systems. Although LCA has proven to be a valid methodology for environmental assessments of bioenergy supply chains, some methodological choices remain critical and challenging, and still lead to inconsistent conclusions.

This chapter analyses these unsolved issues and methodological choices to provide a solid basis for further harmonisation of the LCA evaluation activities. In particular, the aim is to improve the robustness of LCA results and the awareness about the methodological choices, as well as to make the outcomes of different studies comparable. Critical methodological factors such as goal definition, selection of FU, system boundaries and allocation have been discussed.

This chapter contributes to the current debate on harmonisation of environmental impact assessment using LCA for the different and more common renewable energy processes.

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Chapter 7
Energy and Environmental Assessments of Agro-biogas Supply Chains for Energy Generation: A Comprehensive Review

Carlo Ingrao, Jacopo Bacenetti, Giuseppe Ioppolo and Antonio Messineo

Abstract Over the years, the application of Life Cycle Assessment (LCA) to agricultural biogas energy source allowed to depict the environmental impact related to this renewable energy source as well as to highlight mitigation strategies oriented to improvement of Anaerobic Digestion sustainability. A review focused upon the recently published LCAs of agricultural biogas plants was carried out. The review highlighted a huge variability on environmental results due to the ways the feedstock mixtures are produced, managed and supplied; and the regions in which the plants are located. Differences were also related to the ways the energy produced were utilised, whether it was input to the national grid, and/or recycled within the system.

Keywords Biomass · Anaerobic digestion · Life cycle assessment
Energy performance · Environmental sustainability · Review

7.1 Introduction

Nowadays, fossil fuels are still the world's main energy sources (Volpe et al. 2014), though they are responsible for several problems deriving from both their production and utilisation (combustion), like: the exploitation and subsequent decrease of the

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natural reserves; the emission of Greenhouse Gases (GHGs) and other pollutants causing impacts to environmental categories like climate change, human health, and ecosystem quality (Collet et al. 2017). Those and other related problems are increasingly reviving interest in, and fostering efforts towards, developing new technologies to obtain clean energy (Volpe et al. 2014; Karray et al. 2017); not to mention that, due to political unrest in the world, diversification of energy sources is needed (Volpe et al. 2014). Furthermore, to make economic development sustainable there is an urgent need to operate drastic changes, in as a short time as possible, in the direction of efficient, accessibly priced, low-carbon energy supplies (Chiricosta et al. 2014).

The scarcity of resources, the increasing demand for materials and energies, the dematerialisation and substitution approach for sustainable development, are the clear outlines to be considered for implementation of equitable, sustainable post fossil-carbon societies (Ioppolo et al. 2014; Ingrao et al. 2016). Those outlines can be phrased mainly in terms of the transition: from fossil fuels to renewable energy sources; and from linear to circular economies centred upon closed-loop materials (Ioppolo et al. 2014; Ingrao et al. 2018).

A growing interest is, indeed, shown towards renewable energy sources, with many world regions and countries setting ambitious targets as, according to Yasar et al. (2017), such sources are abundant and environmentally friendly.

Biomass is considered as one of the main renewable energy sources and is expected to provide more than a half of the energy demand in the nearest future (Ertem et al. 2017). Biomass contains a lower amount of carbon dioxide (CO_2) than fossil fuels. It is regarded as carbon neutral, since the share of CO_2 that is emitted through combustion is balanced by the CO_2 previously fixed by photosynthesis (Karray et al. 2017).

Biofuels are referred as the fuels obtained from biological resources whether they are made directly or indirectly from photosynthesis (Yasar et al. 2017). Three generation of fuels can be acknowledged.

- First-generation biofuels are produced utilising edible feedstock like corn, soybean, sugarcane and rapeseed (Karray et al. 2017). When used for energy production, those crops become regarded as '*energy crops*'. Several studies documented that intensive exploitation of arable land for cultivation of those crops has negative effects in terms of direct and indirect land use change (d-, i-LUC), as other land surfaces need to be invested for food production. Therefore, there may be a negative impact upon the global stock and prices of food, and an increase of the amounts of GHGs that get to be emitted to the atmosphere (Ertem et al. 2017). The potential risks and attribution of iLUC effects to biofuels are still largely debated today by the environmental impact assessment community. However, the European Commission has been making ways to mitigate iLUC related problems by defining a precautionary threshold (7%) for the share of first-generation biofuels that are used in the transportation market (Rana et al. 2016).
- Second-generation biofuels are those made from agriculture and food industry residues, as well as from dedicated lignocellulosic feedstocks. These fuels are characterised by a set of advantages over the first-generation ones. The major

benefits are represented by the higher stock yields and the lower land requirements in terms of quality and quantity. However, some lacks in terms of economy viability at the large scale are observed for lignocellulose conversion to biofuels due to its strong resistance to degradation (Karray et al. 2017).

– Feedstock for third-generation biofuels is represented by micro- and macroalgae, which present further advantages compared to the previous feedstock categories (Karray et al. 2017). Considering its high photosynthetic effectiveness, fast biomass growth with no arable land required and resistance to contaminations from heavy metals, algae appears to be highly competitive compared to energy crops and second-generation fuel feedstocks (Ertem et al. 2017; Karray et al. 2017).

The use of biomass includes energy generation through a set of methods, such as pyrolysis, combustion, gasification, hydrolysis and fermentation. Amongst these, Anaerobic Digestion (AD) is a widely used biochemical process to produce biogas from biomasses containing high levels of organic matter (Nayal et al. 2016). Two end-products are generally released from biogas plants: biogas; and a nutrient-rich digestate. Biogas is a mixture of methane (CH_4), carbon dioxide (CO_2) and others compounds such as hydrogen sulphide (H_2S), carbon monoxide (CO) and hydrogen (H_2). Methane and carbon dioxide contents are strictly dependent upon the feedstock used and the plant operating conditions, and generally range between 50–60% and 40–50%, respectively (Negri et al. 2014). Digestate is a stabilised material (Nayal et al. 2016) that is generally subjected to a centrifugation treatment where a solid and a liquid fraction are obtained. The former is used as organic soil fertiliser or conditioner or animal bedding, whilst the latter is partly recirculated within the plant to feed the digester and partly used for fertigation activities (Ingrao et al. 2015; Nayal et al. 2016). So, it is understood that, overall, digestate can contribute to reduce both production and consumption of energy intensive fertilisers (Yasar et al. 2017).

Generation of electricity from biogas in those countries being part of the Convention on the Organisation for Economic Cooperation and Development (OECD) grew from 3.7 TWh in 1990 to 78.8 TWh in 2015. This has represented the third fastest growing renewable electricity source after wind and solar energy (Nayal et al. 2016). As highlighted by Nayal et al. (2016), that growth was driven by OECD Europe accounting for almost 80% of the entire OECD production in 2015. Nearly 14 ktoe of biogas primary energy was produced in the European Union (EU) in 2013: 69% of that energy volume was generated in decentralised agricultural plants, facilities for production of methane from Municipal Solid Waste (MSW), and centralised co-digestion plants. As many as 72% of biogas plants operating in the EU are installed in farms utilising agricultural waste, manure and energy crops (Nayal et al. 2016).

So, it is understood that biogas production has expanded hugely in Europe during the last 20–25 years. In this regard, Nayal et al. (2016) documented that in 2013 the world biogas production had reached 59 billion m^3, so representing a 5.5% increase over the previous years. This should be attributed also to biogas from agricultural feedstock AD being recognised by the EU as one of the renewable resources that can be used to produce 20% of the energy demanded in Europe by 2020 (EU 2009). In addition to this, electricity generation from biogas in AD plants was stimulated by subsidies at the regional, national and European level. Currently, Germany has the highest number of AD plants (about 9000), according to the German Biogas Association (2016), followed by Italy (1800 plants) (Negri et al. 2014; GSE 2017). In 2016, the installed capacity reached 4166 MW in Germany and 1406 MW in Italy corresponding to an electricity production of 29.41 TWh in Germany and 8.12 TWh in Italy. For Italy, this figure represents 2.56% of total electricity consumption (GSE 2017).

The feedstock used is generally a mixture of animal manure, energy crops, and agricultural residues, that is designed to maximise biogas production, based upon the biogas potential of each mixture component (Rana et al. 2016; Igos et al. 2016). Nayal et al. (2016) highlighted that around 70% of biogas plants operating in the EU are installed in farms using agricultural waste, manure, and energy crops.

Most AD plants in Europe are fed with cereal and grass silage and grain crops: amongst them cereal silage is favoured because of its high specific biogas production, high energy density and the ease of storage. 50–55% of EU biogas plants´ feedstock is originated from energy crops despite of the growing concern about using food to produce energy. In 2012, Germany used 2.5×10^6 ha of land for the growth of energy crops. Maize often results in the highest yields; therefore, it is the most preferred feedstock for energy generation; thus, 90% of biogas plants in Germany runs—at least partially—with maize as feedstock. According to Negri et al. (2014) about 10% of the agricultural area dedicated to maize in Italy is used to supply biomass to feed agricultural anaerobic digestion plants.

However, due to a set of environmental criticisms related to their cultivation, energy crops like maize and triticale are increasingly being replaced by agriculture and food industry residues. Co-digestion of those residues together with zootechnical effluents (animal manure and sewage) enables achieving a better nutrient balance in AD, and provides optimum carbon-to-nitrogen ratios, whilst decreasing the risk of ammonia inhibition (Nayal et al. 2016). Different types of residues from same geographical areas can be utilised, so allowing for creation and development of integrated waste management systems (Nayal et al. 2016). This generates considerable environmental gains, in terms of energy saving, reuse and recycling of residues within agriculture, and the reduction of CO_2 emissions (Pagés-Díaz et al. 2014; Nayal et al. 2016).

Life Cycle Assessment (LCA) coupled with energy yield analysis can be used to investigate the energy and environmental issues associated with agro-biogas supply chains: from feedstock production and management, to biogas AD and digestate treatment, until electricity and heat cogeneration (Ingrao et al. 2015).

Several LCA studies in Europe and worldwide focused upon environmental assessment of biogas production systems, so creating a solid knowledge base for stakeholders like LCA practitioners, farmers, engineers, company owners, policy- and decision-makers to contribute improvements of the efficiency of such systems and reduction of the related environmental impacts. At the same time, comparing different LCA studies can be challenging due to differences in scope and a lack of documentation. However, those studies are desirable to highlight those differences to lay down the foundations for creation of guidelines and regulations for application of LCA in the bioenergy field.

In this context, Bacenetti et al. (2016) reviewed and compared results of a relevant number LCAs of agricultural AD plants at the world level by considering both methodological and operational aspects. However, studies considered by Bacenetti et al. (2016) were published before 2016. This chapter integrates that paper, as it reviews LCAs that have been published in the next three years, to contribute further investigation of the environmental impacts and improvement potentials associated with AD plants. Set-up and developed like this, the chapter may contribute to further enrichment and understanding of the subject literature and knowledge.

Attention was focused upon AD because it is a well-established, eco-friendly technology to treat organic-matter-rich biomass, also in the form of residues and wastes, that is increasingly being deployed as a renewable energy generation technology (Nayal et al. 2016; Fusi et al. 2016). AD offers, indeed, a series of environmental benefits mainly related to odour control, improved air and water quality, improved nutrient management, flexibility and GHG emission reduction. Furthermore, AD systems enable reducing the release of phosphorus and of copper and zinc into surface waters, when the solid and liquid fractions of digestate are applied onto the soil (Nayal et al. 2016).

Finally, AD of agricultural feedstocks is currently acknowledged to be a viable solution for provision of renewable energy in rural areas where the energy is used locally and the heat cogenerated with electricity is fully exploited.

7.2 Review of the Latest Environmental Assessments in the Agro-biogas Energy Field

This section was dedicated to overviewing papers dealing with environmental assessment of biogas supply chains for energy generation. Based upon the review performed, six papers published in 2016, and nine in 2017 were selected: all of them regarded assessment of AD plants, so emphasising upon the interest and attention that are increasingly being shown towards such bioenergy production systems.

Nayal et al. (2016) investigated the production of a feedstock mixture in Turkey made from slaughterhouse wastes, vegetable wastes from cultivation and harvesting, poultry and cattle manure, and grass from cultivation, and its conversion into energy. For this purpose, they performed LCA to compare, on an equal feedstock basis, two options (AD vs. combined-cycle gasification) for production of the same amount of energy. The Functional Unit (FU) of the study was, indeed, represented by 8599 GJ of electricity provided to 865 houses for one year, and produced from a 10.68 kt/year feedstock mixture. The system boundaries included the phases of feedstock production and utilisation for production of the aforementioned electricity amount. Digestate treatment in the AD plant and administration as organic fertiliser were considered in the assessment.

Overall, the authors demonstrated that both plants are sustainable options compared to the use of hard coal (the most abundant fossil fuel in Turkey), but the environmental impact associated with the AD option is 50% lower than that associated with the gasification one. This should be attributed to the environmental gains resulting from the administration of digestate as organic fertiliser and the avoided usage of equivalently nutrient-rich mineral fertilisers. However, the largest impact to global warming amongst all the life cycle stages comes from the N_2O emission due to digestate application.

Pierie et al. (2016) assessed the overall renewability, sustainability and possible energy yields of biogas production pathways operating on locally available biomass waste flows in Netherlands. The authors considered that questions could be raised regarding the achievability, efficiency, and sustainability of the biogas production pathway when utilising biomasses from energy crops above all if transported them over longer distances. Considering 20 kt of fresh matter per year and the same setup for the AD plant, three different pathways were considered: green gas (part of the produced biogas is used in a small boiler to produce the needed heat for the digestion process), combined heat and power (CHP), and waste treatment. Concerning the multifunctionality issue, the digestate is considered to replace fossil-equivalent quality fertilisers whilst, in the scenario in which the biogas feeds a CHP engine, the produced heat is considered fully exploited and 1 km of pipeline to transport it is considered.

To indicate efficiency and sustainability, two reference scenarios were considered. The first one is the fossil reference scenario based on natural gas and the grey electricity average mix of the Netherlands, whilst the second scenario is the 'maize reference scenario' in which maize silage used as a feedstock is specially cultivated for use in the biogas production pathway.

Although literature indicated that there is sufficient bioenergy potential in local waste streams to reach the renewable production goal set for the year 2020, the authors highlighted that the average useful energy finally produced by the AD production pathway is significantly lower, often due to poor quality biomass and difficult harvesting conditions. Concerning the LCA methodological choices, the authors concluded that the choice of feedstocks, technologies, and the operational values of AD pathways (e.g. feedstock, transport, process) have a significant influence on the environmental impact, and the increased biomass use can claim valuable arable land for cultivation and/or effect biodiversity.

In another study, Collet et al. (2017) designed a CH_4 production system by combining AD and Power-to-Gas (PtG) technology. PtG consists in the utilisation of electricity to convert water into hydrogen (H_2) by electrolysis, and then to synthetize CH_4 from CO_2 and H_2. Several applications are acknowledged for CH_4. In this study, the authors investigated its combustion for heat generation, by considering three alternative plant options: biogas upgrading and CO_2 conversion into CH_4 via methanation; biogas upgrading into CH_4 through its direct methanation; and biogas upgrading without methanation. The first two options provided the utilisation of the PtG technology, whilst the third one did not. In all cases, the AD was fed only with sewage sludge and no other substrates were used. Furthermore, all the plant configurations provided part of the biogas produced to be combusted for production of heat for usage in the AD and biogas purification and compression step. That heat is mixed with the one developed during methanation, when the latter is provided.

In their study, Collet et al. (2017) performed a techno-economic and environmental assessment of the whole methane supply chain, considering the different plant options. The FU was regarded as to be 1 MJ heat produced, and the system boundaries included all the middle unit operations in which the system had been broken down into, from biogas production to CH_4 combustion and heat generation. Technical changes could be investigated to improve the economic and environmental performance of the process, like converting into CH_4 the CO_2 outlet from the biogas combustion stage (Collet et al. 2017). Based upon the findings of the study, it seems that the PtG technology is a prominent and competitive one. However, its economic and environmental performance could increase if renewable electricity and/or a different electricity mix was used to feed the electrolysis process.

In another study, Ertem et al. (2016) analysed the environmental performances of an agricultural biogas plant of a capacity of 500 kW comparing environmental impacts of flexible and the traditional baseload operation. The authors pointed out that flexible biogas supply is vital to balance the power generation and can be realised by biogas storage or flexible biogas production concepts. To this purpose, LCA was performed to detect the environmental impacts of the variety of feedstock in co-digestion scenarios by substitution of maize and the loading rate scenarios with a focus on flexible feedstock utilisation.

The evaluated AD plant operates with the co-digestion of maize, grass, rye silage and chicken manure. The selected FU was 1 kWh of produced electricity and the system boundary included crop production, purchase, ensilage, storage, AD, storage of residues and application of digestate for agricultural production, transport between multiple stages of the anaerobic digestion and, lastly, the biogas combustion and the supply of generated electricity into the grid and heat utilisation for temperature control at the fermenters and poultry housing. As in other studies (Pierie et al. 2016; Bacenetti et al. 2016; Lijo et al. 2017a), concerning the animal slurry only the collection and storage of the produced digestate were considered. The produced digestate was supposed to replace mineral fertilisers. The amounts of mineral fertiliser replaced by the application of digestate were calculated based on the digestate composition and fertiliser replacement values. Primary data were collected by surveys at the AD plants over 2 years, whilst background data related to the production of construction materials, agricultural tractor, CHP unit, energy, fuels, fertilisers and pesticides were taken from the Ecoinvent 2.2 database.

With regard to the outcomes, the authors highlighted that: (i) 10–45% of GHG emissions could be saved by changes in feed management; in particular, the substitution of maize with waste could involve a reduction up to 10% for Global Warming Potential and Acidification Potential; (ii) demand-based production would require 16% higher energy input.

Iordan et al. (2016) used the LCA approach to evaluate the environmental sustainability of electricity production through anaerobic co-digestion of sewage sludge and organic wastes. The selected FU was 1 MJ of electricity from sewage sludge, fats from food industry, sludge from septic tanks and other biological substrates in a Norwegian biogas plant. Because the feedstock is represented by four types of wastes supplied by different industries, no upstream impacts were considered for the substrates. The system boundaries included only the transport of the feedstocks to the plant, the capital infrastructure and its end of life, the use of biogas in a CHP plant for electricity production, as well as the management of the digestate. Final distribution and usage of electricity was not considered in the assessment. Since the heat and the digestate are not economically exploited, all impacts are attributed to the electricity production and no allocation is needed. Regarding the digestate, the authors did not carry out the allocation because the digestate market in Norway is still under construction and the biogas plants receive no revenues from recycling the digestate. The analysis relies on primary data from a biogas plant, supplemented with data from the literature. For background data Ecoinvent 2.2 was used. The environmental performances of the biogas system are benchmarked against a conventional fossil fuel system. The achieved results highlighted how the biogas system has better environmental performance than the fossil reference system for the acidification and particulate matter formation potentials. The sensitivity analysis carried out showed that the most sensitive parameter is the storage of the digestate.

The study carried out by Uusitalo et al. (2016) focused on the environmental assessment of the potential benefits arising from the exploitation of surplus heat by means of an organic Rankine cycle (ORC). Although heat is a typical co-product of all AD plants producing electricity via the combustion of biogas in a CHP engine, its exploitation is complicated. In fact, the surplus heat available for exploitation has a huge variation, during seasons, depending on the variation of the air temperature and, consequently, of heat self-consumption. Furthermore, the heat demand in agriculture is typically concentrated during the winter season, when also the heat self-consumption for heating of digesters is the biggest. In this context, the study carried out by Uusitalo et al. (2016) assessed the potential reduction of GHG emissions by using ORC for recovering exhaust gas heat of biogas engines. The study highlighted how the ORC is a suitable technological option for converting low-grade heat into electricity with relatively high efficiency. More in details, two scenarios (the first where only electricity from a gas engine is utilised, the second where electricity and heat from a gas engine are utilised) with four cases for each scenario were evaluated. The four cases are: (A) additional electricity is produced using average methods; (B) additional electricity is produced using marginal methods, (C) additional electricity is produced using biogas; (D) the ORC process is used. The comparison amongst the different cases is modelled using the system expansion method according to the ISO/TR 14049. According to system expansion, a similar number of products is produced in different cases to enable fair comparison. Therefore, the authors considered a similar amount of electricity and heat in all cases. Because with the ORC process more electricity is produced, for the cases A and C additional electricity had to be produced using other electricity production methods. The results of the study pointed out that GHG emissions can be reduced significantly if the thermal energy of the exhausted gases, otherwise lost during the process as heat waste, is utilised for additional electricity production by means of ORC. However, when the heat is already used in the form of heat power, the use of ORC does not necessarily lead to GHG emission reductions. The results also indicate that the working fluid leakages and production as well as the ORC construction materials and production have only marginal effects on the results from the GHG perspective.

Sometimes Cover Crops (CCs) are used as co-substrate in the feedstock mixture. CCs are crops that are planted on the field during the part of the year when the land would usually be left fallow. The main aim of this is to reduce erosion and improve the structure and water retention capacity of the soil (Igos et al. 2016). Furthermore, management of cropping systems with CCs also brings benefits related to soil characteristics, such as increased humus building, increased compaction, biological weed and pest control, moisture conservation, as well as improved nutrient cycling through avoided leaching and enhanced nitrogen sequestration (Igos et al. 2016). In the past, CCs were mainly cultivated as 'green manure', which meant that they were not harvested, but incorporated into the soil before the main crop was sown. If on one side, this contributes to higher yields of the main crop, on the other side new biomasses are needed for energy production, due to the growing scarcity and the societal dependence upon fossil fuels, as well as the increasing biomass demand (Igos et al. 2016). This aspect was investigated by Igos et al. (2016), who evaluated the environmental repercussions of planting rye as a winter CC after maize cultivation, and its utilisation as a co-substrate with maize and manure in an AD plant. This scenario was considered by the authors as an alternative to that providing leaving the land fallow during winter and only using maize in co-digestion with manure. An LCA was conducted by the authors for this purpose. The FU of the system was chosen to be 1 MJ energy. The system was split into three main sub-systems related to: feedstock production and management; feedstock supply to the AD plant and processing into biogas and digestate; biogas conversion into energy. The phase of feedstock production was represented by crop cultivation and manure management. Rye and maize, or maize only, depending upon the scenario considered, were co-digested with pig manure.

The obtained biogas is then used for cogeneration of both electricity (36%) and heat (64%), which are entirely reused to feed the plant energy demand. Digestate was treated as an organic fertiliser for cultivation of the maize that is utilised within the system.

The study highlighted that the usage of rye as a winter CC could lead to significant environmental benefits on a local scale, mainly related to: reduced nutrient demand and nitrate leaching; optimised land use; and avoided use of herbicides (Igos et al. 2016). However, the lower rye productivity and specific biogas potential compared to maize, generate increase of indirect environmental impacts due to the higher consumption of materials and energies for 1 MJ bioenergy production (Igos et al. 2016). Therefore, as also suggested by the authors, a trade-off should be found between mitigation of local environmental impacts and lower energy productivity.

Biogas from biomass AD can be used as fuels for Solid Oxide Fuel Cell (SOFC) systems, which represent valid alternative solutions to conventional power generation systems (Rillo et al. 2017). However, LCAs and related assessments need to be conducted on such complex systems of biomass transformation and energy generation to find rooms for improvements. Such was done by Rillo et al. (2017), who performed LCA to address the environmental issues associated with a SOFC based system, where biogas from sewage sludge AD was used as fuel to SOFC. The FU was identified in 1 kWh electricity produced by the plant. The system boundaries were at the plant gate and encompassed the biogas production in the AD stage, and the phases of manufacturing, operation and maintenance of the SOFC plant. Results showed that SOFC manufacturing is the most impacting due to the usage of large amounts of resources, materials, and energies. As regards plant operation, the highest environmental burdens come from the AD phase for biogas production. However, due to its superior environmental performance, overall SOFC can be considered as a valid alternative to conventional systems, such as internal combustion engines and micro gas turbines. Hence, it represents an interesting choice for cleaner manners of bioenergy generation.

Biogas production and utilisation for energy generation can be quite a valid option for smallholder farmers utilising waste products from household, farming and animal breeding activities. In most cases, biogas is produced and used onsite through conversion into energy (electricity and heat) or upgrading into high-value gas like (bio-)methane, as shown by the studies reviewed thus far in this chapter and those reviewed by Bacenetti et al. (2016). However, it can play multiple key roles in the development of rural new energies, because it is also well suited for small-deployment in rural areas where it can be easily integrated into farm systems and managed with relatively little operation and maintenance effort (Hou et al. 2017). As a matter of fact, this form of biogas known as Rural Household Biogas (RHB) has been increasing in popularity in areas where smallholder farmers predominate, such as Southeast Asia, China, and Africa (Hou et al. 2017). However, there is increasing concern about the real economic and environmental performance of RHB-based systems, which led an increasing number of those systems to be abandoned in China (Hou et al. 2017). These and related issues were assessed by Hou et al. (2017) who performed an LCA for evaluation of the net GHG mitigation effect of RHB systems integrated into small household farms; tracked RHB system deployment in different areas in relation to driving forces; and proposed policy recommendations to improve the effectiveness of biogas GHG mitigation solutions.

The authors documented that RHB systems can lead to a series of environmental benefits, mainly related to reduction of fossil fuel consumption, GHG emissions from manure storage and management, and chemical fertiliser inputs. However, when poorly operated, RHB systems can end up in increasing GHG emissions, aggravating nutrient surplus on farmland, increasing labour inputs, and cause economic loss. This emphasises upon the need for the design of those systems to be carefully matched to local conditions and farmers' needs with respect to manure management and energy requirements (Hou et al. 2017). Technical options such as small pumps to draw out digestate from digesters could improve operational efficiency and reduce labour requirements for RHB systems. Moreover, widespread adoption of more precise nutrient management planning would ensure efficient utilisation of digestate to realise potential fertiliser substitution effect (Hou et al. 2017).

Biogas is increasingly being produced from marine algae, mainly due to higher rates of CO_2 fixation, greater potential for carbon dioxide remediation and, hence, higher production yields per unit of area, compared to terrestrial biomass (Karray et al. 2017). In this context, Ertem et al. (2017) performed an LCA to investigate the energy efficiency and environmental sustainability of biogas production when the energy crops are partially replaced with macroalgae as feedstock to an industrial-scale AD plant. Two feedstock scenarios were, indeed, tested by the authors: chicken manure with macroalgae; and chicken manure with energy crops. In line with the previous studies, the boundaries of the analysed system included: manure collection; macroalgae and energy-crop cultivation; storage and handling of the three substrates; storage, treatment and handling of the digestate produced, cogeneration of electricity and heat from biogas; and lastly the transportation. Two FUs were chosen by the authors in their study, so to highlight whether and how results from comparison of the two scenarios are affected by methodological issues like the choice of the FU. Considering the function which AD plants are generally designed for, the FUs chosen by the authors were 1 kg feedstock mixture and 1 MJ produced energy.

The authors documented that a different FU leads to different results, so highlighting the importance of best interpreting the function of the system and the objective of the study to correctly operate the FU choice.

The study carried out by Lijo et al. (2017a) aimed at analysing the eco-efficiency of 15 agricultural biogas plants located in Northern Italy. To this purpose, LCA and data envelopment analysis (DEA) methodologies were combined aiming to identify efficient operational plants and proposing improvement measures for the inefficient ones. The 15 AD plants are fed with different feedstock: energy crops (maize and triticale silage, maize flour, organic fraction of municipal solid waste (OFMSW), food waste, animal slurry and glycerol. The FU selected was the production of 1 MWh of electricity. The system boundaries included the cultivation of energy crops, the transport and handling of all input materials, the anaerobic digestion process, the use of biogas in the CHP system and the digestate management. The production of organic wastes such as manure, food and industrial wastes was excluded since they were considered waste streams from other production systems. Concerning the digestate the authors considered that some plants produce more digestate than the required for the cultivation of their own crops. In this case, the avoided mineral fertilisation using ammonium nitrate was estimated according to the nitrogen replacement value of 65%. Concerning the heat cogenerated by the CHP, it was considered wasted except for one biogas plant located nearby a greenhouse. In this case, avoided production of the same amount of heat from natural gas was considered. With regard to the main outcomes of the studies, the authors pointed out that: (i) the production of electricity from biogas in all plants would imply environmental benefits compared with the average electricity production in the Italian grid; (ii) to improve the environmental performances of electricity from AD plants special attention should be paid to the feedstock selection since it has a key role in the overall eco-efficiency of the plant, due to their different origin and composition; (iii) besides the emission from digestate storage, feedstock production are the main environmental hotspots.

A second LCA of Lijò et al. (2017b) was carried out to identify the environmental consequences of feedstock selection in biogas production paying attention to the use of OFMSW. To this purpose, two real biogas plants were assessed and compared from a life cycle perspective. The firs plant performs the co-digestion of energy crops (78%) and animal waste (22%) whilst the second one consumes energy crops (4%), food waste (29%) and OFMSW (67%). The selected FU was 1 MWh of electricity produced whilst the system boundary includes maize and triticale cultivation, feedstock transport, bioenergy production, digestate management and use of surplus digestate. Waste production was considered outside of the system boundaries whilst its transport was included. Concerning the digestate, the following items were included: transport and spread, the production of the machinery required as well as diesel combustion emissions and field emissions from the application of digestate. When more digestate is produced with respect to the amount applicable over the agricultural area directly managed by the AD plant owner, the authors considered that the surplus digestate is used in other agricultural systems, reducing the use of mineral fertilisers by including the avoided products perspective. Nevertheless, unlike than in several previous studies (for more details see Pierie et al. 2016; Ertem et al. 2016, as well as the review carried out by Bacenetti et al. 2016), only avoided N fertilisers were considered as environmental credits. In fact, the agricultural soil in the area around the AD plant (Northern Italy) presents high contents of P and K, which makes the addition of P- and K-based fertilisers unnecessary. As for Lijò et al. (2017a), amongst the different inputs and outputs, feedstock production and emissions from digestate storage were identified as one of the main sources for the plant fed with agricultural biomass.

The study carried out by Arodudu et al. (2017) pointed out that previous LCAs for agro-bioenergy production rarely considered some agronomic factors with local and regional impacts. Based on the results of previous LCA studies the authors highlighted that depending on the assumptions in some studies the environmental impacts of producing bioenergy on arable land cannot be considered sustainable, whilst other researches consider the production of electricity from biogas one of the most effective direct emission reduction and fossil fuel replacement measures. In this context, Arodudu et al. (2017) improved LCA methods to examine the individual and combined effects of often overlooked agronomic factors (e.g. alternative farm power, seed sowing, fertiliser, tillage and irrigation options) on life cycle energy indicators. The system boundary considered in the evaluations involves cultivation of energy crops (ploughing, harrowing, ridging, seed sowing, fertiliser application, pesticide application, liming, irrigation, harvesting, etc.), transportation (e.g. from farm to input market, input market to farm, farm to bio-refinery, bio-refinery to farm, etc.) and conversion of biomass to energy. The manufacturing of the production factors directly consumed over the production process such as fuels, fertilisers, herbicides, lime, etc. was included too. Manufacturing, start-up and maintenance of machineries (e.g. tractors, irrigation systems, biorefineries, etc.) was excluded from the system boundary. The digestate was accounted for as fertiliser (N, P and K) and lime replacements without any consideration about the real need on the soil.

Finally, Table 7.1 summarises the main results from the review carried out. Regarding the feedstock, the papers reviewed present a wide variability. In particular, the AD plants studied are fed with maize, rye, cover crops, algae, agricultural waste (animal manure and slurry) as well as with OFMSW. This remarks how the feeding of AD

Table 7.1 Methodological aspects best representing the research topics addressed in the papers reviewed

Study	Geographic area	Technology	Functional unit	System boundary	Feedstock
Arodudu et al. (2017)	Tropic, sub-tropic and the temperate landscapes	AD for biogas	1 J of energy from maize ethanol and maize biogas	From cradle to AD plant gate	Maize silage
Ertem et al. (2016)	Northeast region of Germany	AD for biogas	1 kWh of electricity	From cradle to AD plant gate	Maize, rye grass silage, chicken manure
Ertem et al. (2017)	Northeast region of Germany	AD for biogas	1 kg feedstock mixture; and 1 MJ produced energy	From cradle to AD plant gate	Macroalgae, and chicken manure and energy crops.
Collet et al. (2017)		AD for biogas and power-to-gas technology	1 MJ of heat	From cradle to AD plant gate	Maize, grass, rye silage, and chicken manure
Hou et al. (2017)	China	Rural household biogas	1 Rural Household	From cradle to grave	Animal manure and kitchen garbage
Igos et al. (2016)	Flanders (Belgium)	AD for biogas	1 MJ energy (36% electricity and 64% heat)	From cradle to AD plant gate	Maize silage, rye, manure
Iordan et al. (2016)	Norway	AD for biogas	1 MJ of electricity	From cradle to grave	Sewage sludge and sludge from industry
Pierie et al. (2016)	Netherlands	AD for biogas	GJ of energy/km^2	From cradle to grave	OFMSW
Lijo et al. (2017a)	Italy	AD for biogas	1 MWh of electricity	From cradle to AD plant gate	Energy crops and OFMSW
Lijo et al. (2017b)	Italy	AD for biogas	1 MWh of electricity	From cradle to AD plant gate	Energy crops and OFMSW

(continued)

Table 7.1 (continued)

Study	Geographic area	Technology	Functional unit	System boundary	Feedstock
Nayal et al. (2016)	Turkey	AD for biogas versus combined-cycle gasification	8599 GJ of electricity	From cradle to AD plant gate	Slaughterhouse wastes, vegetable wastes from cultivation and harvesting, poultry and cattle manure, and grass
Rillo et al. (2017)	Not indicated	Solid oxide fuel cell fed with biogas	1 kWh of electricity	From cradle to AD plant gate	Rye and maize, or maize only, depending upon the scenario considered, were co-digested with pig manure
Uusitalo et al. (2016)	Europe	AD for biogas, plant with ORC	1 kWh of electricity	From cradle to AD plant gate	Not indicated
Yasar et al. (2017)	Pakistan	AD for biogas	1 tonne	From gate to gate (only digestate management is included)	Cow-dung, potato pulp

plants is still under investigation, especially considering the potential environmental benefits arising from the substitution of cereal silage (maize silage above all) with other feedstock coming from crops with low inputs requirement (e.g., cover crops) or from waste and alternative biomass (e.g. algae).

7.3 Discussions of Review Results

Based upon the review performed, some important points are worthy of being highlighted. It was observed that all studies were based upon application of LCA, often in combination with other methods like those for estimation of the on-field emissions deriving from farming activities, or of the energy balances associated with the plants. This proves once again LCA to be a valid methodology for environmental assessments of bioenergy supply chains. The systems investigated in those studies were based upon AD, sometimes integrated with other technologies to form more complex energy production systems, as done in Collet et al. (2017), and Rillo et al. (2017). In this regard, in line with Bacenetti et al. (2016), the review highlighted that, thanks to its flexibility and multifunctionality, AD can be considered as a useful technology to produce renewable energy, in that it enables converting several biomass streams into useful products and closing organic matter cycles.

In all the AD plants investigated, zoo-technical effluents (sewage and manure) were an important substrate of the biomass stock used to feed the plant, representing around 20% (Nayal et al. 2016) to 100% (Collet et al. 2017; Rillo et al. 2017). Those effluents were modelled using a zero-burden approach, so considering only their collection, transport to, and handling within, the plant. This has become by now a well-approved practice in LCAs of bioenergy systems that utilise animal breeding and agricultural wastes.

Different feedstock mixtures were tested and compared in those studies, with the aim of finding trade-off options between biogas yield and environmental sustainability. In most of those papers, mixing manure with biomass residues was proven as a valid solution for mitigation of the environmental impacts associated with the whole bioenergy supply chain. The choice of feedstocks, technologies and the operational values of AD plants was documented, however, to significantly affect the associated energy yield and environmental impact.

Furthermore, marine macroalgae were tested for usage in replacement of energy crops, highlighting their higher biogas potential and lower environmental impact (per MJ energy produced) compared to the most commonly used silages in the field, like those of rye, maize and grass (Ertem et al. 2017).

Another interesting aspect emerging from the review was about the potential environmental benefits related to the usage of cover crops as AD feedstock substrate. However, just a couple of studies were found in the literature to be dealing with the energy and environmental issues associated with both feedstock types (Igos et al. 2016; Ertem et al. 2017). Such a result emphasises upon the need for LCAs and other assessments for further investigation and improvement of those issues, so contributing to more sustainable AD-based plants.

The systems investigated in the studies reviewed provided the output biogas to be used, also within more complex plant systems like in Collet et al. (2017), Rillo et al. (2017), and Hou et al. (2017), for cogeneration of electricity and heat, or for conversion into bio-methane.

For all studies, the system boundaries were designed to include the main unit operations which the systems were broken down into: from feedstock production, transportation and handling to energy generation and utilisation. In this context, attention was shown towards computation of the on-field emissions resulting from soil management and those of N_2O generated by the administration of mineral and organic fertilisers, like the digestate fractions outlet from the plant. In addition to this, uncovered storage of digestate before and after treatment resulted as responsible of large impacts due to N_2O emission, so impacting upon climate change.

Allocation was performed by the authors when needed based upon the system investigated and, however, generally between biogas and digestate, and between the heat and electricity produced by the cogeneration plant. Some authors, however, did not carry out allocation due to the absence of a real market for the digestate outlet of an AD plant, and to biogas plants receiving no revenues from recycling the digestate. With regard to digestate, the review highlighted an increasing attention to the environmental consequences related to its different utilisation, as well as to the peculiar methodological challenges that the production of an organic fertilisers involves (recirculation-complete utilisation to fertilise the crops used to fed the digesters, substitution of mineral fertilisers considering its composition in NPK, substitution of mineral fertilisers considering only the real requirement of the different agricultural areas).

Primary data were used in those studies in combination with secondary data that were extrapolated from databases like Ecoinvent and/or from the subject literature. Primary data were collected onsite by performing multi-year surveys and/or, for some studies, derived from the design phase of the whole systems whether the latter had not been implemented yet, like in Collet et al. (2017) and Rillo et al. (2017).

Other differences amongst the studies reviewed were found in the application of the methodology, with regard to the study background and setup, the data inventoried, and the environmental impact assessment method. Some authors limited their assessments at the midpoint approach, whilst some others extended it to the endpoint approach. Some assessments considered a set of impact categories to best represent the system investigated, mainly related to global warming, non-renewable energy, acidification, eutrophication, particulate matter formation, and land transformation. For contrast, others (i.e. Nayal et al. 2016; and Hou et al. 2017) were focussed only upon GHG emission estimation.

One big difference that was easily recognised by the authors was related to both the function of the system investigated and the objective of the study. In those papers, the FU was indeed chosen to best represent the system under investigation and be consistent with the aim which the study was designed with. Those two aspects led to the FU being different from study to study, thereby making comparisons sometimes different to be conducted. In this regard, Ertem et al. (2017) chose two different FUs for assessment of the same system and showed that different FUs cause changes in the results. If the aim is to produce higher amounts of energy, substitution of energy crops with macroalgae is a liable solution because it would help solving the dilemma between energy and food production. In this way, bioenergy production yield would be maximised, and the iLUC related problems would be avoided. For contrast, when the attention is focussed upon the AD feedstock, it would be beneficial to analyse the whole system based upon the kg of feedstock produced. In this case, utilisation of energy crops other than macroalgae could be more favourable to mitigate the environmental impacts associated with the AD plant (Ertem et al. 2017).

In line with Bacenetti et al. (2016), differences were found in the feedstock mixture and in the regions in which: the plants were located; and the feedstock substrates were produced/obtained, managed and supplied. Differences were also related to the ways the energy produced were utilised, whether it was input to the national grid, and/or recycled within the system. In this regard, an interesting aspect was the one related to the potential environmental benefits deriving from the exploitation of the surplus heat by means of an organic Rankine cycle (Uusitalo et al. 2016).

According to the authors, those and other related differences generate huge variability in the results, and difficulties of making evaluations and drawing general conclusions on the environmental sustainability of AD plants.

7.4 Conclusions and Future Trends

Over the years, the application of LCA to agricultural biogas technology allowed to depict the environmental impact related to this renewable energy source as well as to highlight mitigation strategies oriented to improvement of AD sustainability.

Nevertheless, there are unsolved challenges and methodological choices that should be harmonised for improving the robustness of LCA results and to make the outcomes of different studies comparable.

To best model feedstock production, primary data should be collected also considering the wide geographic and temporal variability of cultivation practices and biomass yield. For this reason, the use of secondary data may affect the reliability of the results, especially if not duly adjusted. Concerning digestate emissions, primary data collection is expensive, hazardous and time-consuming; consequently, the use of secondary data is frequently inevitable. Nevertheless, site-specific data should be used to assess these emissions, as they are deeply affected by climatic conditions (Bacenetti et al. 2016).

Moreover, the choice of the Life Cycle Impact Assessment method should be carefully evaluated considering the goal of the study and the selection of impact categories. When the study aims at assessing AD plants fed by energy crops, an impact assessment method able to properly quantify the impact categories affected by fertiliser related emissions (e.g. acidification and eutrophication) should be used. Other methods, possibly different than these, should be considered when a comparative LCA of differently sized biogas plants is conducted to highlight differences in term of plant construction building and maintenance.

Finally, in line with Bacenetti et al. (2016), environmental LCAs are increasingly being relevant for marketing strategies, supply chain management, and politic decision-making. A higher level of transparency and a harmonisation of the preparation of biogas LCAs would be desirable to improve the comparability of LCA results. This could stimulate the creation of biogas-specific technical standards to guide and regulate the assessment, and communication, of the energy and environmental performances of biogas-derived energy systems.

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Chapter 8
A Review on Potential Candidate Lignocellulosic Feedstocks for Bio-energy Supply Chain

Amalia Zucaro, Angelo Fierro and Annachiara Forte

Abstract In the context of an increased bio-based economy characterized by both reduced dependence upon imported fossil fuels and reduced greenhouse gases emissions, bio-fuels and the other bio-based supply chains have reached a worldwide expansion. Taking into account the high environmental impact of the agricultural production and the potential conflicts among food, energy and environment, this review provides an overview of the opportunities and constraints specifically related to the environmental performance of different candidate lignocellulosic feedstock in the Italian context. Peer-reviewed Life Cycle Assessment (LCA) studies were analysed and compared on a mass basis. Several biomass-based supply chains from wood and herbaceous residues or dedicated crops on marginal and fertile lands (under different fertilization management) were considered. A cradle-to-farm gate attributional LCA approach was applied to assess the environmental profile and the linked major hotspots as useful information to evaluate the most promising feedstock for bio-energy or integrated biorefinery systems. The results have demonstrated that short rotation forestry and medium rotation forestry cultivation systems, characterized by restrained mineral fertilization, can have a better environmental performance than herbaceous crops under both standard and reduced fertilization management, offering substantial benefits for almost all investigated impact categories.

Keywords Environmental performance · Annual and perennial crops
SRF and MRF · Dedicated feedstock · Residual biomass · Energy crops

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8.1 Introduction

The rapid growth in population and industrialization gave rise to increase the energy demand and the dependence on fossil-based products. There is a clear scientific evidence that the global change arises from human influence and it is strictly related to the fossil fuel consumption (IPCC 2014). Therefore, the interest in developing environmentally friendly supply chains from renewable feedstocks has considerably increased (Forte et al. 2016). Over the last decades, the transition to a decarbonised energy system brought to increase the bio-energy production, with an expected growth of bio-fuels, such as bioethanol (EtOH) and biodiesel (Gomiero 2017). In this context, the exploitation of lignocellulosic energy crops for bio-energy or other bio-based productions are increasingly considered as a strategy to not affect food security and reduce environmental impacts (Solinas et al. 2015). The European Union (EU) encourages the employment of second generation feedstock, such as energy crops or waste raw materials (Directive 2009/28/EC). In the Italian context the energy crops have strongly grown over the last years as well, mainly driven by dedicated subsidization policy (Bartoli et al. 2016), with a rising bio-fuel oriented policy for greenhouse gases (GHG) and fossil energy saving in the transport sector (D. Lgs. 03/03/2011 n.28; COM15 final, 2014).

The lignocellulosic materials are considered as the promising feedstock for bio-based industrial processes due to their chemical features and composition (Anwar et al. 2014). Such materials are considered natural and renewable resource essential to the functioning of modern industrial societies even if much of the lignocellulosic biomass is still disposed of by burning (Anwar et al. 2014). This biomass can potentially be converted into different high value products including bio-fuels, chemicals, and cheap energy sources (Anwar et al. 2014; Zucaro et al. 2016a).

However, the environmental performance of bio-fuels, bio-materials and bio-chemicals from lignocellulosic biomass, over the entire production chain, needs to be carefully investigated. The Life Cycle Assessment (LCA) has been widely recognized as one of the most suitable analytical approaches to deeply analyse the environmental performance of processes or products (Bessou et al. 2013).

At the present time, the enhanced use of biomass is strictly connected to the widespread opinion that bio-based products are less pollutant than their fossil-counterparts and do not contribute to net CO_2 emissions. The pertinent scientific literature shows controversial results and highlights the crop phase as the major environmental hotspot of several bio-based supply chains (Forte et al. 2017; Zucaro et al. 2017), due to the farming managements (Milà i Canals et al. 2006; Bessou et al. 2013) and the site-specific conditions for the local emissions (Bessou et al. 2013). For this reason, the environmental performance of dedicated crops or residues biomass should be subject to a constant evaluation and monitoring in site-specific conditions for effective territorial environmental friendly bio-based strategies. In this regard, in the Italian context, preliminary studies were carried out comparing the environmental performance of different oleaginous biomasses (Cocco et al. 2014; Forleo et al. 2017), whilst there is a lack of comprehensive evaluation of the environmental profile of alternative lignocellulosic feedstock for bio-energy/biorefinery purposes. The present work is a literature review of peer-reviewed articles on the LCA of lignocellulosic biomass (dedicated and residual ones) referred to the Italian context. Specifically, different lignocellulosic biomass productions were analysed and compared by means of a cradle-to-farm gate attributional LCA approach to assess the environmental profile and the related major hotspots for the largest number of impact categories. All the resulting information will serve as a useful base to identify the main environmental pros and cons of lignocellulosic bio-based routes in the Italian context.

8.2 Methodological Issues

8.2.1 Papers Selection and Clustering

This review was designed to summarize and critically address the LCA studies for lignocellulosic biomass production for bio-based supply chains in the Italian context. Scientific literature, published in the last 10 years, was investigated through the following e-resources: Scopus, Google Scholar and Sciencedirect. Afterwards, this work focused only on the full attributional LCA studies applied to biomass-based supply chains published in peer-reviewed journals or in peer-reviewed conference proceedings, with a specific focus on the crop phase. The selected studies are reported in Table 8.1, associating an identification number, consistently used thought all figures, to each work and summarizing the most relevant information about key parameters such as: (i) the biomass feedstock, (ii) the type of land used, (iii) the functional unit (FU), (iv) the system boundaries, (v) allocation procedures, (vi) the applied impact assessment methods (IAM) and (vii) the linked analysed impacts. For the present review, the selected system boundaries were from cradle-to-farm gate and the FU was set to 1 kg of total lignocellulosic dry biomass production (thought the specific crop life cycles), since the biomass yield is a key parameter influencing the environmental performance of the farm systems and the whole bio-based supply chains (Bosco et al. 2016). In order to standardize and properly compare the different studies, when necessary the FU was converted to the selected one and the results were extrapolated to match the cradle-to-farm gate system boundary. Additionally, since the choice of the life cycle IAM was not always consistent among the selected studies (Table 8.1), to extensively discuss the results for the largest available number of LCA impact categories, the authors re-elaborated, by means of SimaPro 8.2.0 software (Pré 2018), the results from their own studies (Forte et al. 2015, 2016, 2017; Zucaro et al. 2015, 2016a, b, 2018) moving from the ReCiPe to the ILCD or CML methods. The cumulative energy demand (CED) was also evaluated applying the single-issue method to the data available by the authors. All data were clustered in the following three groups: (i) woody lignocellulosic biomass through short and medium rotation forestry (SRF-MRF), (ii) perennial herbaceous crops, (iii) annual herbaceous crops.

The only available data for agricultural straw residues (herbaceous, straw) was kept separate. In this regard, the results by the ILCD IAM allowed the comparison of environmental impacts among all the three groups (SRF-MRF, herbaceous perennial and herbaceous annual); whilst the results by the CML IAM provided a further specific focus on potential differences between herbaceous perennial and annual feedstock (see Table 8.1 and Sect. 8.2.2 for additional details).

8.2.2 *Statistical Analysis*

Statistical analyses were performed using the Sigma Plot package (Sigma Plot 2012). The ANOVA 'One Way Analysis of Variance' test ($p < 0.05$) was used to check significant differences among: (i) the environmental impacts of the SRF-MRF, herbaceous perennial and herbaceous annual feedstock by the ILCD IAM; (ii) the key agronomic input (N, P and K fertilizers and diesel) required per 1 kg of dry SRF-MRF, herbaceous perennial and annual biomass produced. The t-test ($p < 0.05$) was used to further investigate the impacts of the perennial herbaceous crops *versus* the annual herbaceous feedstocks through the CML IAM. For each feedstock, the relationships among the environmental impacts and the key agronomic parameters (N, P and K fertilizers and diesel input per 1 kg of dry biomass) were investigated through the Pearson Product-Moment Test and linear regression analysis.

8.3 Results and Discussion

Figure 8.1a shows a significantly lower impact in terms of Climate Change (CC) for the woody biomass compared to both the herbaceous perennial and annual feedstocks, likely linked to the combined effect of the restrained fertilizer inputs (Table 8.2) and the higher biomass yield related to the whole crop life time of willow and poplar crops compared to the herbaceous cultivations. Indeed, SRF-MRF crops are characterized by a higher nitrogen-use efficiency and a reduced use N-fertilizer input (often applied only as organic N in the pre-plant phase) (Banacetti et al. 2012, 2016; Djomo et al. 2015). Otherwise, notwithstanding the CC impact resulted linearly related to the K, P fertilizers and diesel input (Table 8.3), no clear separation was observed among the groups in relation to these parameters, due to comparable fertilization schemes and fuel consumption patterns (linked to the high mechanization for the biomass collection) for woody and herbaceous crops.

The inclusion of the soil carbon dynamic in the GHG inventory might amplify the outcome of comparative analyses between perennial (herbaceous) and annual crops (Bessou et al. 2013), since the former are usually recognized to entail a potential long-term soil carbon storage (SCS) thanks to: (i) a longer C turnover of the more extensive rooting systems (Monti and Zatta 2009); (ii) limited soil management (planting and related tillage, to be shared for the whole lifetime) (Monti et al. 2009); (iii) a reduced

Table 8.1 Selected LCA studies about lignocellulosic crops

	Feedstock	Paper	Land	System boundary	FU	Allocation	LCIA-M	Impact category[a]
1	Wood SRF-Poplar	Banacetti et al. (2012)	Fertile	Cradle-to-farm gate	1 ha of poplar plantation	n.a.	IPCC-GWP, CED	Global warming potential and cumulative energy demand
2	Wood MRF-Poplar							
3	Wood SRC-Poplar	Banacetti et al. (2016)	Fertile	Cradle-to-farm gate	1 t (tonnes) of dry-chipped biomass	LHV of biomass	ILCD method	CC, OD, HTc, HT, PM, POF, A, FE, TE, ME, FEx, MFRD
4	Wood SRC-Willow							
5	Wood SRC-Poplar (best clone)							
6	Wood SRC-Willow (best clone)							
7	Herbaceous Perennial- Switchgrass;	Monti et al. (2009)	Fertile	Cradle-to-farm gate	Hectare and energy based	n.a	CML 2	GWP, OLD, Ac, Eu, T-t, MW-t, FW-t, H-t, A-d.
8	Herbaceous Perennial- Giant reed							

(continued)

Table 8.1 (continued)

	Feedstock	Paper	Land	System boundary	FU	Allocation	LCIA-M	Impact category[a]
9	Herbaceous Perennial-*Miscanthus*							
10	Herbaceous Perennial-Cardoon							
11	Herbaceous Perennial-Switchgrass	Fazio and Monti (2011)	Fertile	Cradle-to-grave (electricity/heat/transport fuels	Land—(hectare) and energy—(Joule)	n.a.	CML 2 and Eco-indicator 99	GWP, OLD, Ac, Eu, T-t, MW-t, FW-t, H-t, A-d.
12	Herbaceous Perennial-Giant reed							
13	Herbaceous Perennial-*Miscanthus*							
14	Herbaceous Perennial-Cardoon							
15	Herbaceous Annual-Fibre *Sorghum*							
16	Herbaceous Perennial-Giant reed	Forte et al. (2015)	Marginal	Cradle-to-farm gate	1 kg of dry matter	n.a.	ReCiPe midpoint (H)	CC, OD, TA, FE, ME, POF, PMF, TET, FET, MET, WD, FD

(continued)

Table 8.1 (continued)

	Feedstock	Paper	Land	System boundary	FU	Allocation	LCIA-M	Impact category[a]
17	Herbaceous Perennial-Giant reed	Zucaro et al. (2015)	Marginal	Cradle-to-farm gate	1 kg of dry matter	n.a	ReCiPe midpoint (H)	CC, OD, TA, FE, ME, POF, PMF, WD, FD
18	Herbaceous Annual-Fibre *Sorghum*							
19	Herbaceous Perennial-Giant reed	Bosco et al. (2016)	Fertile	Cradle-to-plant gate	1 ha and 1 t of dry biomass	n.a.	CML	Energy balance, gross and net greenhouse gas emissions, eutrophication potential, acidification potential, ozone layer depletion potential, photochemical ozone creation potential
20	Herbaceous Perennial-Giant reed		Marginal	Cradle-to-plant gate		n.a.	CML	
21	Herbaceous Perennial-Cardoon	Zucaro et al. (2016b)	Marginal	Cradle-to-farm gate	1 kg of dry matter	Economic	ReCiPe Midpoint (H)	CC, OD, TA, FE, ME, POF, PMF, WD, FD

(continued)

Table 8.1 (continued)

	Feedstock	Paper	Land	System boundary	FU	Allocation	LCIA-M	Impact category[a]
22	Herbaceous Annual-Fibre *Sorghum*							
23	Herbaceous Straw-Wheat straw	Forte et al. (2016)	Fertile	Cradle-to-farm gate	1 kg of dry matter and 1 kg of bio-based 1,4-Butanediol	Economic	ReCiPe midpoint (H)	CC, OD, TA, FE, ME, POF, PMF, WD, FD n
24	Herbaceous Perennial-Giant reed	Zucaro et al. (2016a)	Marginal	Cradle-to-wheel	1 kg of dry matter and 1 km driven	n.a.	ReCiPe midpoint (H)	CC, OD, TA, FE, ME, POF, PMF, WD, FD
25	Herbaceous Perennial-Giant reed high-input	Zucaro et al. (2018)	Marginal	Cradle-to-farm gate	1 kg of dry matter	n.a.	ReCiPe midpoint (H)	CC, OD, TA, FE, ME, POF, PMF, WD, FD
26	Herbaceous Perennial-Giant reed low-input							
27	Herbaceous Annual-Fibre *sorghum*	Forte et al. (2017)	Marginal	Cradle-to-farm gate, cradle-to-wheel	1 kg of dry matter and 1 km travelled	n.a.	ReCiPe Midpoint (H)	CC, OD, TA, FE, ME, POF, PMF, WD, FD
28	Wood SRC-Poplar/Willow Casale Monferrato	Djomo et al. (2015)	Fertile	Cradle-to-farm gate	Hectare based	n.a.	Energy output and balance	Energy ratio, net energy yield

(continued)

Table 8.1 (continued)

	Feedstock	Paper	Land	System boundary	FU	Allocation	LCIA-M	Impact category[a]
29	Wood SRC-Poplar Ostiano							
30	Wood SRC-Poplar Pisa							
31	Wood SRC—Poplar Bagni di Tivoli							

[a] Acronyms LCIA-M: life cycle impact assessment method, FU: functional unit, for ILCD impact categories: climate change (CC), ozone depletion (OD), human toxicity, cancer effects (HTc), human toxicity, non-cancer effects (HT), particulate matter (PM), photochemical ozone formation (POF), acidification (A), freshwater eutrophication (FE), terrestrial eutrophication (TE), marine eutrophication (ME), freshwater ecotoxicity (FEx) and mineral, fossil and renewable resource depletion (MFRD), acronyms for CML impact categories: global warming potential (GWP), ozone layer depletion (OLD), rainfall acidification (Ac), water eutrophication (Eu), terrestrial ecotoxicity (T-t), marine water ecotoxicity (MW-t), fresh water ecotoxicity (FW-t) human toxicity (H-t), abiotic depletion (A-d). acronyms for ReCiPe impact categories: climate change (CC); ozone depletion (OD), terrestrial acidification (TA), freshwater eutrophication (FE), marine eutrophication (ME), photochemical oxidant formation (POF), particulate matter formation (PMF), terrestrial ecotoxicity (TET), freshwater ecotoxicity (FET), marine ecotoxicity (MET), water depletion (WD), fossil depletion (FD)

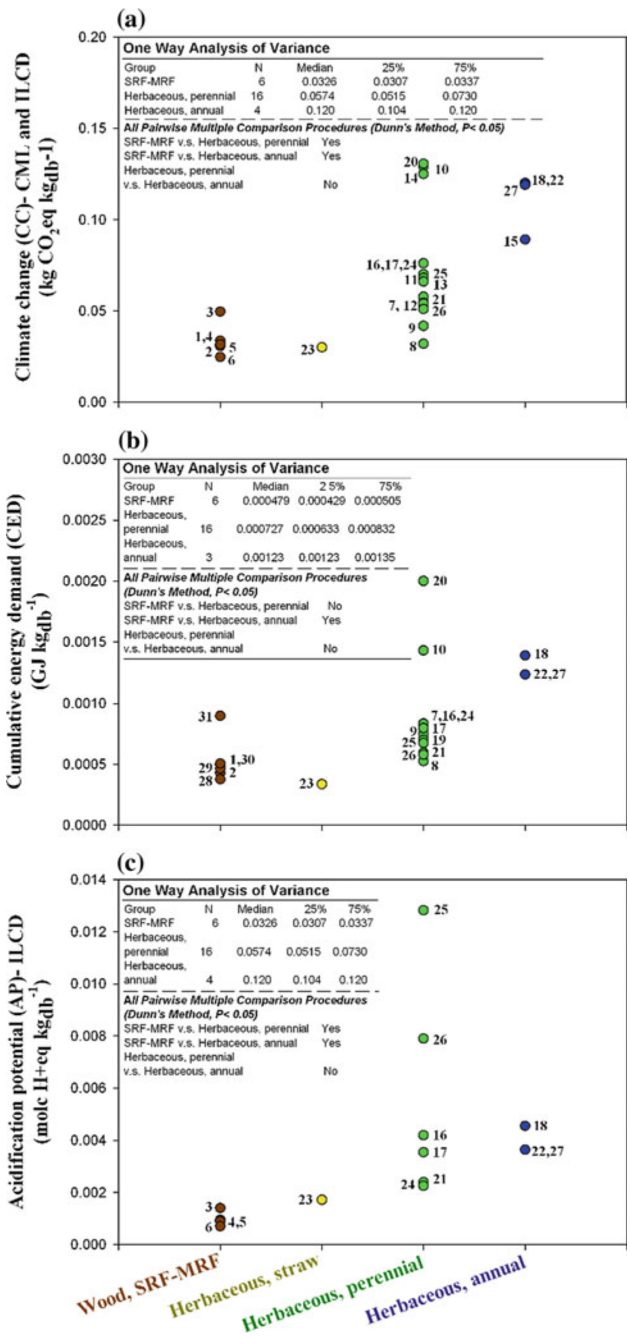

Fig. 8.1 **a** CC, **b** CED and **c** AP impacts of the different lignocellulosic feedstock. The inset table shows the results of the ANOVA-one way analysis of variance ($p < 0.05$)

Table 8.2 Kruskal-wallis one way analysis of variance on ranks ($p < 0.05$) for the key agronomic input (N, P and K fertilizer and diesel) required per 1 kg of dry biomass produced through SRF-MRF, herbaceous perennial and herbaceous annual cultivations

N-fertilizer input (kg kg^{-1} of dry biomass)				
Cluster	Number	Median	25%	75%
SRF-MRF	12	0.000234	0.00223	0.00323
Herbaceous, perennial	23	0.00522	0.00381	0.00667
Herbaceous, annual	7	0.00667	0.00428	0.00811
All pairwise multiple comparison procedures (Dunn's method, p<0.05)				
SRF-MRF versus Herbaceous, perennial		Yes		
SRF-MRF versus Herbaceou, annual		Yes		
Herbaceous, perennial versus Herbaceou, annual		No		
P-fertilizer input (kg kg^{-1} of dry biomass)—$p = 0.978$				
Cluster	Number	Median	25%	75%
SRF-MRF	12	0.000297	0.000252	0.000297
Herbaceous, perennial	23	0.000416	0.0000483	0.00107
Herbaceous, annual	7	0.000554	0.000	0.00164
K-fertilizer input (kg kg^{-1} of dry biomass)—$p = 0.197$				
Cluster	Number	Median	25%	75%
SRF-MRF	12	0.00109	0.00084	0.00109
Herbaceous, perennial	23	0.000178	0.000	0.000985
Herbaceous, annual	7	0.00046	0.000	0.00254
Diesel input (kg kg^{-1} of dry biomass)—$p = 0.249$				
Cluster	Number	Median	25%	75%
SRF-MRF	12	0.00499	0.0024	0.0326
Herbaceous, perennial	23	0.00415	0.00308	0.0064
Herbaceous, annual	7	0.00825	0.00612	0.0102

Table 8.3 Significant correlations among analysed environmental impacts and key N, K, P and diesel inputs per kg of dry biomass produced (pearson product-moment test)

Impact	N-fertilizer (kg kg_{bd}^{-1})	P-fertilizer (kg kg_{bd}^{-1})	K-fertilizer (kg kg_{bd}^{-1})	Diesel (kg kg_{bd}^{-1})
CC (kg CO_2 eq kg^{-1}_{bd})	0.752*** (27)	0.613*** (26)	0.439* (27)	0.823*** (27)
CED (MJ kg^{-1}_{bd})	0.832*** (22)			0.821*** (22)
POF (kg NMVOC eq kg_{db}^{-1})		0.624* (14)	0.542* (14)	
OD (kg CFC-11 eq kg_{db}^{-1})	0.813*** (23)	0.636** (23)		0.826*** (23)
PM (kg PM2.5 eq kg_{db}^{-1})	0.752** (14)			0.753** (14)
TE (molc N eq kg_{db}^{-1})	0.666** (14)			0.620* (14)

*$p < 0.05$; **$p < 0.01$; ***$p < 0.001$. Sample size in parentheses

risk of soil erosion (Angelini et al. 2009); (iv) an increase in soil carbon content and biodiversity (Angelini et al. 2009). Although there is a general consensus on the importance of the below-ground biomass in withdrawing C from the atmosphere (Monti and Zatta 2009) only few studies provided quantitative data on roots of energy crops and the possible plant CO_2 uptake (Monti and Zatta 2009). Therefore, in this review the direct estimate of SCS were not included. For the perennial-giant reed (GR) crop preliminary measures of SCS have highlighted a potential clime change mitigation showing in some cases a net sink of atmospheric CO_2 (Forte et al. 2015; Zucaro et al. 2018).

The results achieved from the evaluation of the CED impact category underlined a significant difference between dedicated woody crops and annual herbaceous crops (Fig. 8.1b). The differences between the SRF-MRF group and herbaceous perennial or between herbaceous perennial and annual were not significant (Fig. 8.1b). In the first case the result was affected by the poplar feedstock cropped in Bagni di Tivoli (see Table 8.1 for additional details), subjected to an annual cultivation management comparable to the herbaceous perennial crops. For the second case, the GR cultivation on marginal soil (point 20, Table 8.1), in spite of higher rates of N fertilization, produced less than half biomass respect to the same GR crop on fertile soil (Bosco et al. 2016). This finding underlined that the high energy demand requested to produce fertilizers might not significantly affect long-term productivity (Cadoux et al. 2014; Djomo et al. 2015).

Table 8.3 shows a linear dependency between the CED impact and the required inputs of N-fertilizer and diesel per kg of dry biomass produced, with an average highest impact for the annual feedstock according to the average highest fuel consumption (Table 8.2). In this regard, the amount of fertilizers and choice of mechanical harvest

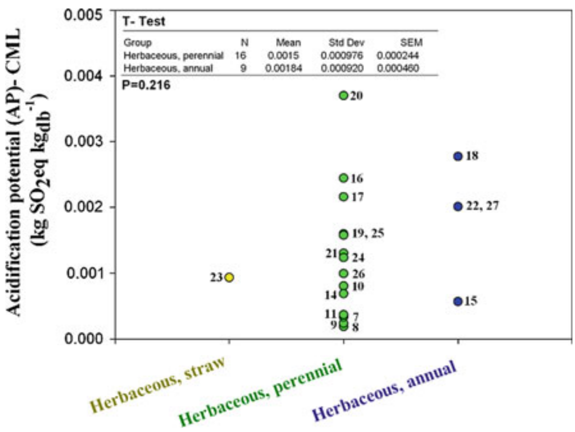

Fig. 8.2 AP of the herbaceous lignocellulosic feedstock. The inset table shows the results of the comparison of perennial versus annual feedstock

yard can largely affect the depletion of fossil resources (Bosco et al. 2016; Zucaro et al. 2018).

For the Acidification potential (AP), the statistical results by the ILCD methods (Fig. 8.1c) show significant differences between the SRF-MRF and the herbaceous groups (both annual and perennial), highlighting much lower impacts for woody crops. The comparative analysis by the CML method (Fig. 8.2) confirms the slight (not significant) differences between the AP impact generated by the herbaceous perennial and annual crops. Relevant tradeoffs may occur in the assessment of AP impacts with both IAMs, due to no clear assessment of Direct Field Emissions (DFE) as highlighted by several authors (Forte et al. 2015; Mbonimpa et al. 2016). Specifically, comparing the same crops in some of the reviewed papers (Monti et al. 2009; Fazio and Monti 2011) the contributions of both N fertilization and harvest operations were on the whole lower compared to the average share highlighted in the other studies (Forte et al. 2015; Zucaro et al. 2018). This was most likely due to differences in the DFE included and the chosen calculation methodology (Bessou et al. 2013; Forte et al. 2015); however only some of the studies at the national level reported the detailed accounting procedure of each DFE analysed (Forte et al. 2015, 2016; Bosco et al. 2016; Zucaro et al. 2016a, 2018). Therefore, no positive correlations emerged between the main hotspot inputs (fertilizers and diesel consumption) and the linked target AP impacts. Nevertheless, as highlighted by some authors (Forte et al. 2015 and Zucaro et al. 2018) the volatilized ammonia (NH_3) emissions, linked to N fertilization practices, highly influenced (up to 70–75%) the acidification impacts.

The results achieved for ozone depletion (OD) were shown in Fig. 8.3a. For both investigated IAMs, ILCD and CML, the OD impact was measured in kg CFC-11 eq. For this reason, the results were processed and presented together. The OD results show significant differences (Fig. 8.3a) among groups due to the higher dependency on the use of fertilizers and diesel consumption (Table 8.3) mainly linked to the upstream halocarbures emissions (Zucaro et al. 2018). Indeed, the constrained use of mineral fertilizers for woody crops (Table 8.2) has produced a lower OD impact, linearly related to the N-fertilizer input (Table 8.3).

The results achieved for Photochemical Ozone Formation (POF) category (Fig. 8.3b) did not show a significant difference among clusters. The average POF value for woody crops was less than the herbaceous one. Indeed, whilst for the woody crops the mechanization of field operations (reaching almost 90% of impact) has been detected as the main responsible of POF impact (Bacenetti et al. 2016), for the perennial and annual crops the total POF impact was influenced by both upstream (from fertilizer and agricultural machinery productions) and downstream (from machinery on-field operations) emissions (Forte et al. 2017). The POF regression analysis (Table 8.3) highlighted the importance of upstream NO_x emissions emitted during fertilizer manufacturing showing a linear dependency in the use of P and K fertilizers. These results showed the importance in the assessment of the whole production chain, underlining how the different accounting of the indirect emissions might produce marked differences in the results evaluation.

The investigation of particular matter (PM) impact category pointed out a net separation among the investigated clusters (Fig. 8.3c). Nevertheless, the differences between herbaceous crops were not significant (Fig. 8.3c), also in this case due to the overlapping crop management for both perennial and annual lignocellulosic feedstocks. The PM correlation diagrams (Table 8.3) shows a clear dependence by: (i) the increase of N-fertilizer, mainly related to ammonia (NH_3) volatilization, due to the specific NH_3 emission factor (DFE calculation) and the scheduled fertilization rates (Bacenetti et al. 2016; Forte et al. 2017), (ii) the use of diesel in the agricultural machinery (reaching in same case 40–60% of PM impact) (Table 8.3). Therefore, more reliable estimate of NH_3 emissions from Mediterranean cropped lands (Sanz-Cobena et al. 2008) as well as the monitoring of the on-field mechanization (Zucaro et al. 2015) would be very beneficial.

The eutrophication potential (EP) was largely affected by the fertilizer application (Monti et al. 2009; Bacenetti et al. 2016; Zucaro et al. 2018) and to a lower extent by the sulphur dioxide emissions from the combustion of diesel fuel (González-García et al. 2013). In the ILCD method three different EP impact categories are considered: terrestrial (TE), freshwater (FE) and marine (ME) (Fig. 8.4).

For TE significant differences among SRF-MRF and the perennial and annual lignocellulosic clusters were highlighted (Fig. 8.4a). TE values for woody crops were lower than the TE impact generated by both herbaceous crops mainly due to the lower fertilizer inputs. TE impacts were driven by the N-fertilizer input (Table 8.3), due to the key role of NH_3 emissions (about 90%). Also the agricultural mechanization tuned the TE impact (Table 8.3) as highlighted by Bacenetti et al. (2016) and González-García et al. (2013).

The FE values for SRF-MRF cluster were higher than the FE impact generated by the herbaceous crops (Fig. 8.4b). For the annual and perennial herbaceous crops the P-fertilization (when applicable, see Table 8.1 for details) was the highest contribution (Forte et al. 2015). The difference between woody crops and perennial herbaceous was significant (Fig. 8.4b) showing a clear separation of the two clusters, but only for FE the SRF-MRF group displayed higher impact than the perennial one. Currently

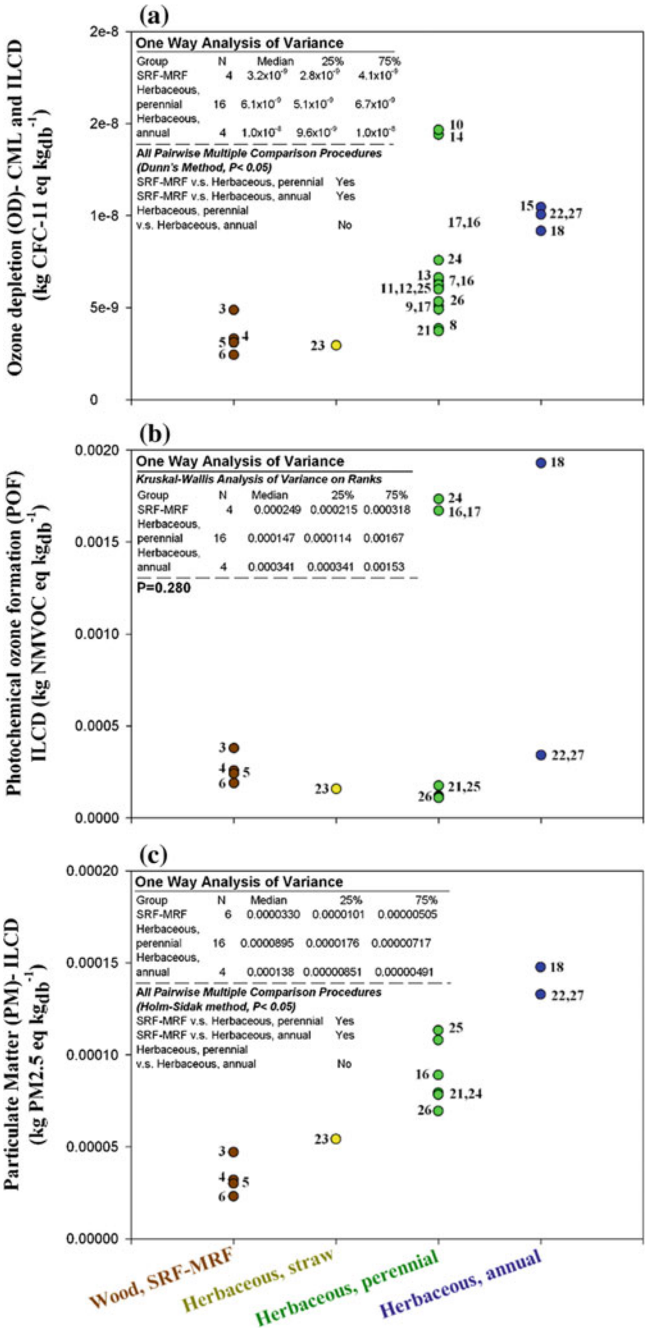

Fig. 8.3 **a** OD, **b** POF and **c** PM impacts of the different lignocellulosic feedstock. The inset table shows the results of the ANOVA-one way analysis of variance ($p < 0.05$)

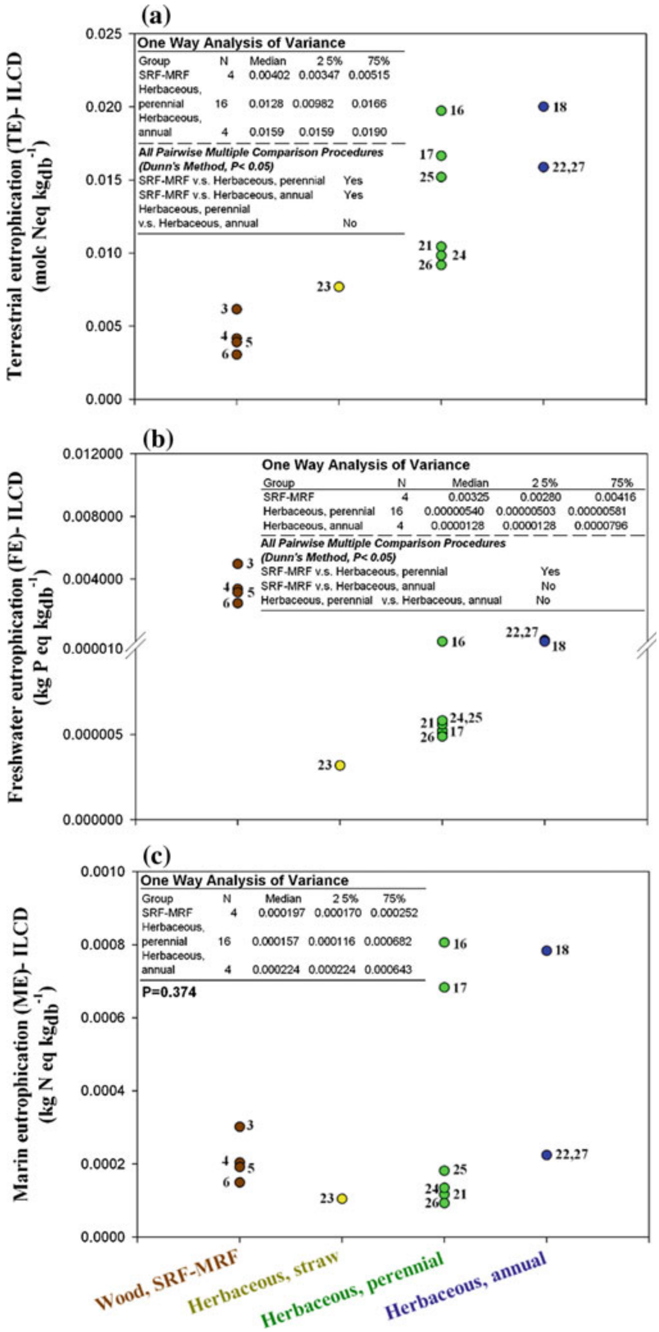

Fig. 8.4 **a** TE, **b** FE and **c** ME impacts of the different lignocellulosic feedstock. The inset table shows the results of the ANOVA-one way analysis of variance ($p < 0.05$)

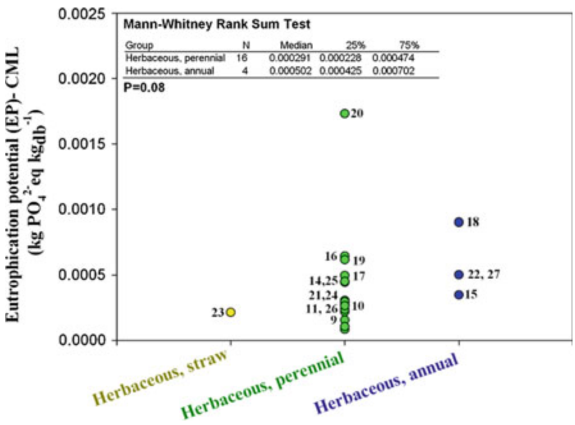

Fig. 8.5 EP impacts of the herbaceous lignocellulosic feedstock. The inset table shows the results of the comparison of perennial versus annual feedstock

the higher variability in the estimation of site-specific factors for P discharge and diffuse N emissions from soil to aquatic ecosystems (nitrate, NO_3^- leaching) may produce significant errors in the calculation of FE impact (Ortiz-Reyes and Anex 2018). The correct estimates of P discharges and nitrate losses to groundwater via leaching are a key challenge to be achieved.

Both emissions are dependent on local conditions, transport mechanisms, soil P concentrations, conservation measures such as managed riparian zones, and how fertilizer is incorporated into the soil (Ortiz-Reyes and Anex 2018). Therefore, also for this impact category the strongly dependency on agricultural management (e.g. fertilization rates) as well as on site-specific soil and climate conditions (Brentrup et al. 2004) requires appropriate DFE calculation procedure (Brentrup et al. 2004; Ortiz-Reyes and Anex 2018).

The combined effects of NH_3 emissions and the risk of nitrate losses considerably affected the ME impact (Forte et al. 2017; Zucaro et al. 2018) but also the mechanization of on-field operations cannot be neglected (Bacenetti et al. 2012; Forte et al. 2015). Nevertheless, the results of ME highlighted a less impact for willows and poplar crops compared to the other investigated lignocellulosic production showing not significant differences among groups (Fig. 8.4c).

The evaluation of EP impact category with CML method is shown in Fig. 8.5. The results achieved by the t-Test highlighted similar EP results for all investigated lignocellulosic herbaceous crops due to the comparable fertilization management. Two points are stand out: (i) the fibre sorghum cultivated in marginal land (point 18 Fig. 8.5 and Table 8.1) as combined results by the NH_3 volatilization after urea supply and the upstream and downstream emissions related to phosphorus input (Zucaro et al. 2015) and (ii) the GR cultivation on marginal soil (point 20 Fig. 8.5 and Table 8.1) producing about 170% higher EP impact compared to the same crop in fertile soil (Bosco et al. 2016). This finding highlighted the key role of biomass yield that can be considered as the main driver of environmental results of crop phase (Bacenetti et al. 2012; Bosco et al. 2016).

8.4 Concluding Remarks

The main findings achieved in this review can be useful for operators and stakeholders involved in the implementation of new bio-based supply chains, in particular regarding the choice of the best lignocellulosic feedstock to use from both a productive and an environmental point of view. Different management intensities have highlighted different environmental performance within the same group and in the cultivation of the same crop in different soils (marginal *versus* fertile). The choice of an applied method depends on the scope and objective of the study and the accounting of its limitations in results interpretation should be always discussed. Similarly, the calculation of the DFE emission needs to be clearly assessed considering the crop- and site- specific characteristics. The final outcome of this review has preliminarily highlighted for the investigated impact categories the SRF and MRF crops as the most promising lignocellulosic feedstock to supply bio-energy and/or biorefinery networks. Nevertheless, to routinely assess the environmental sustainability of bio-based processes a transparent environmental life cycle standardized procedure should be achieved.

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Chapter 9
Life Cycle Assessments of Waste-Based Biorefineries—A Critical Review

Serena Righi

Abstract In recent years advanced biorefineries based on organic residues and waste have gained increased attention for their potential to obviate first-generation biorefineries environmental burdens. During the conceptual design phase of an advanced biorefinery the role of Life Cycle Assessment (LCA) is crucial for providing information on its environmental performances, better solutions, preferable process setup, more suitable feedstock, trade-off, and so on. This review focuses on advanced biorefineries LCAs in order to accomplish a synthesis of the state of the art from the methodological point of view. Some main methodological issues have been analyzed and discussed on 24 LCAs. Attention has been drawn to functional units, system boundaries, inventory data collection, allocation methods and multifunctionality management approach. Results show different approaches and solutions to the analyzed aspects but some clear addresses can be pointed out. It has been observed that LCA of biorefineries can be classified in three different types in base on focal aim, and then functional units are consequentially defined. A large variability has been observed regarding system boundaries even if "cradle-to-gate" appears the most common. Inventories are mainly based on secondary data due to the very innovative features of the analyzed technologies. No general consensus has been observed concerning allocation of environmental impact between co-products.

Keywords Biorefinery · Agricultural residues · Biomass residues
Life cycle assessment

9.1 Introduction

The Paris Agreement on climate change (UNFCCC 2015) is also an agreement about energy. In fact, transformative change in the energy sector, which accounts

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for roughly two-thirds of all anthropogenic greenhouse-gas emissions, is essential to reach the objectives of the Agreement (IEA 2015). The World Energy Outlook 2016 (WEO-2016) has processed all the Paris climate pledges to examine future energy trends using the World Energy Model. In its main scenario, New Policies Scenario, WEO-2016 foresees a 30% rise in global energy demand to 2040 with an increase in consumption for all modern fuels, i.e., natural gas, oil, coal, nuclear (IEA 2016). As far as oil is concerned, demand slows over the projection period, but tops 103 million barrels per day by 2040, compared with 92 million barrels per day in 2016 (British Petroleum 2017). Over the longer term, oil demand in WEO-2016's main scenario concentrates in freight, aviation and petrochemicals, areas where alternatives are scarce. These three sectors account for all of the growth in global oil consumption.

As in WEO-2016 forecasts, oil will continue to be a foundation of the global energy system for many decades to come but the transition to a low-carbon era is the inevitable choice to tackle global warming and achieve sustainable development. The low-carbon transition in some cases is already driving a boom in innovation and emerging businesses (OECD/IEA/NEA/ITF 2015).

Green economy can be considered as a means to make the transition to a sustainable and low-carbon economy (OECD 2009) and bio-based economy as a key element for both green economy and low-carbon transition, being able to replace fossil fuels on a large scale, not only for energy applications, but also for chemicals and materials applications (Scarlet et al. 2015).

In this context, biorefining approach for integrated production of energies and products is emerging as a fundamental pillar of bioeconomy and sustainable development. As stated during the Global Bioeconomy Summit 2015, "there is no scientific evidence that the material use of biomass provides greater sustainability benefits than the energetic use, or vice versa. On the contrary, there is evidence that the combined energetic and material use ("biorefinering") of biomass has the potential for large sustainability benefits" (Ree and Jungmeier 2015).

9.2 Biorefineries

The International Energy Agency (IEA) Bioenergy Task 42 defines biorefining "the sustainable processing of biomass into a spectrum of bio-based products (food, feed, chemicals, and/or materials) and bioenergy (biofuels, power and/or heat)" (IEA 2017).

Task 42 of IEA Bioenergy has developed a biorefinery classification system based on four main features that identify, classify and describe the different biorefinery systems: platforms, energy/products, feedstocks, and conversion processes (if necessary). The platforms are intermediates connecting different biorefinery systems and their processes. Among the most important platforms which can be recognized in biorefineries are: biogas, syngas, hydrogen, C6 sugars, C5 sugars, and lignin. The number of involved platforms is an indication of the system complexity. The two biorefinery product groups are energy and products. In energy-driven biorefineries the biomass is primarily used for the production of secondary energy carriers (biofuels, power and/or heat); process residues are sold or are upgraded to added-value bio-based products, in order to optimize the benefits of the full biomass supply chain. In product-driven biorefineries the biomass is fractionized into a portfolio of bio-based products, after which the process residues are used for power and/or heat production. The two main feedstock groups are “energy crops” from agriculture (e.g., starch crops, short rotation forestry) and “biomass residues” from agriculture, forestry, trade, and industry (e.g., straw, bark, wood chips from forest residues, used cooking oils, waste streams from biomass processing). Finally, four main conversion processes are identified: biochemical, thermochemical, chemical, and mechanical processes (Cherubini et al. 2009a; IEA 2009).

The emerging advanced biorefineries based on more sustainably derived biomass feedstocks and cleaner thermochemical and biological conversion technologies promise to be a more sustainable and environmentally benign system than the conventional biorefineries (IEA 2009).

The application of Life Cycle Assessment (LCA) to advanced biorefineries appears fundamental to assess their environmental performances and measure the overcoming of limitations shown by conventional biorefineries.

The aim of this chapter is to analyze the case studies dealt with the application of LCA to advanced biorefineries in order to understand which methodological choices have been applied and to provide comments and recommendations.

9.3 Method

The literature has been searched via the bibliographic database Scopus using as keywords synonyms for LCA (“life cycle analysis”, “life cycle assessment”, and “LCA”) and “biorefinery” (or “biorefineries”). The search words have been connected by Boolean operators and applied to titles and abstracts.

In order to focalize on the most environmentally promising technologies and feedstocks, only studies that consider residues as feedstock and embrace biorefineries combining energetic and material use have been included in this analysis. Twenty-four LCAs fulfilled this inclusion criterion (Table 9.1).

The review focuses on some methodological aspects selected as key issues influencing LCA outcomes. These issues include: functional unit (FU), inventory data, system boundaries, and methods to handle multifunctionality (Ahlgren et al. 2015; Cherubini et al. 2009b; Muench and Guenther 2013; Saraiva 2017).

The LCA studies have been divided in three types following the method proposed by Saraiva (2017). Type 1 includes LCAs aiming at comparing different process configurations to obtain the same group of products, these LCAs have generally input-related FUs; type 2 comprises LCAs focused on a main product and aimed to compare this main product to others with the same function, these LCAs have generally output-related FUs; and type 3 includes LCAs that aim to compare the use

Table 9.1 Characteristics of the LCA studies included in the review

Code	Authors	Type	Feedstock	FU	Syst. boundaries
1	Cherubini and Ulgiati (2010)	1	Corn stover, wheat straw	477 t/y feedstock	Cradle to gate
2	Daful and Görgens (2017)	1	Sugarcane residues	1 t lactic acid (LA)	Cradle to gate
3	Ekman and Börjesson (2011)	2	Potato juice and glycerol	1 kg propionic acid	Cradle to gate
4	Ekman et al. (2013)	1	Onion waste, birch bark	1 t feedstock	Gate to gate
5	Falano et al. (2014)	2	Wheat straw, forest residues	1 L EtOH	Cradle to gate
6	Fazard et al. (2017)	1	Sugarcane residues	65 t/h bagasse	Cradle to gate
7	Gilani and Stuart (2015)	1	Wood chips	Biorefinery portfolio	Cradle to gate
8	González-García et al. (2011)	2	Forest residue	1 t cellulose	Cradle to gate
9	González-García et al. (2016a)	1	Pinus bark chip	100 kg pinus chip	Cradle to gate
10	González-García et al. (2016b)	2	Forest residue	285 kWh	Cradle to grave
11	Jeswani et al. (2015)	2	Wheat straw, forest residues	1 L EtOH	Cradle to gate
12	Karlsson et al. (2014)	2	Straw and forest residues	1 MJ EtOH	Well-to-tank
13	Liu and Shonnard (2014)	2	Wastewater	1 MJ EtOH + 1 kg Kac	Cradle to grave
14	Mandegari et al. (2017)	1-2	Sugarcane residues	65 t/h bagasse	Cradle to gate
15	Morales et al. (2017)	2	Forest residue	1 kg glucose	Cradle to gate
16	Mu et al. (2010)	2	Agro—forest residues, waste	1 L EtOH	Cradle to gate
17	Nascimento et al. (2016)	2	Coconut unripe husk	1 g cellulose	Cradle to gate
18	Parajuli et al. (2017)	1	Wheat straw	1 MJ EtOH + 1 kg LA	Cradle to gate
19	Piemonte (2012)	2	Forest residue	1 kg EtOH + 1 kWh	Cradle to gate
20	Pourbafrani et al. (2013)	2	Citrus waste	Biorefinery portfolio	Well-to-wheel
21	Spatari et al. (2010)	2	Corn stover	1 L EtOH	Cradle to gate

(continued)

Table 9.1 (continued)

Code	Authors	Type	Feedstock	FU	Syst. boundaries
22	Tonini et al. (2016)	1-3	Agro—industrial residues	1 t feedstock	Cradle to gate
23	Uihlein and Schebek (2009)	3	Straw	1 t feedstock	Cradle to gate
24	Vaskan et al. (2017)	1	Palm empty fruit bunches	1 t feedstock	Well-to-wheel

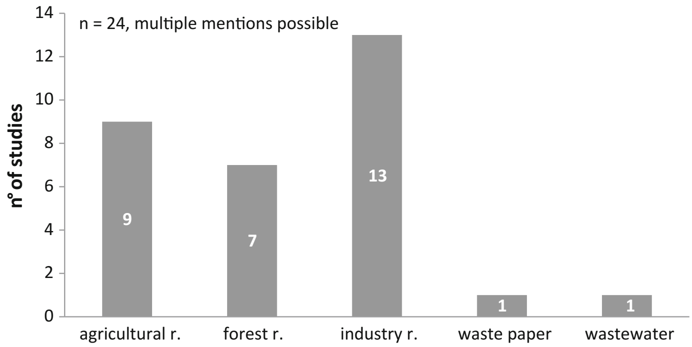

Fig. 9.1 Residues and waste considered in reviewed biorefinery LCAs

of different feedstock to obtain the same group of products, also these LCAs have generally input-related FUs.

Figure 9.1 stresses the key role that agricultural, forest and industrial residues play as feedstock of advanced biorefineries. In all reported cases, agricultural residues are corn stover or wheat straw or both of them. In one case, grass from natural areas is also cited (Tonini et al. 2016). Attention is focused on corn stover and wheat straw because of their abundance all over the world. They are the main agricultural residue available in North America (Kurian et al. 2013), moreover the latter is also prevalent in Asia and Europe (Kim and Dale 2004). Also forest residues are largely present and analyzed. Special attention is paid to industrial residues. Investigated cases are very varied: from food industry to pulp and paper manufacture, from biodiesel production to hardboard facilities. Little consideration is given to wastewater and waste paper. Organic fraction of municipal solid waste and sewage sludge are not present.

9.4 Results

9.4.1 Functional Unit

Biorefineries are supposed to produce two or more products (often fuels and materials) therefore the FU definition is particularly important and, in some cases, a difficult issue. As reported by Saraiva (2017) and Ahlgren et al. (2015) the FU choice is strictly related to the aim of the study and to the final recipients. This review confirms the observation of Saraiva (2017) on the relationship between LCA study type and input-related or output-related FUs: types 1 and 3 have input-related FUs while type 2 has output-related FUs with the sole exception of Parajuli et al. (2017). According to Ahlgren et al. (2015), four types of FUs have been identified: the feedstock (studies 1, 4, 6, 9, 14, 22, 23, 24), the main product or the main function (studies 2, 3, 5, 8, 10, 11, 12, 15, 16, 17, 21) and a combination of output products (studies 7, 13, 18, 19, 20). Generally, feedstock and main products or functions as FUs provide results easier to interpret and compare. On the contrary, the choice of the whole portfolio of the biorefinery as FU is more difficult to interpret and communicate but offers the possibility to have an overall view on the environmental performances of the entire biorefinery.

9.4.2 System Boundary

The system boundary determines which processes of a system are assessed. Table 9.1 above shows that the majority of the studies in the review (18 of 24) apply "cradle to gate" system boundaries. Two studies perform a "cradle to grave" analysis and two studies are "well to wheel". The Table 9.1 shows that only one of the 24 reviewed studies is "gate to gate", and only one is "well to tank".

"Cradle to gate" is the most common choice both when the objective is to compare different process configurations and when the objective is to compare a main product to one with the same function; both for energy-driven and product-driven biorefineries. Even if the prevalent approach is a "cradle to gate", the processes included in the system boundaries are different. The "cradle to gate" system is expected to include the biomass supplying, i.e., the production of the biomass, its harvesting or collection, and the transportation to the refinery. Figure 9.2 shows the number of studies including biomass production, collection and transportation; the processing of biomass in the biorefinery is not shown since all studies, obviously, include it. Figure 9.2 also presents the number of studies which includes the emissions due to the land use change (LUC) and the indirect land use change (iLUC) into the system boundaries.

The impacts of feedstock cultivation are included in the system boundaries only in 50% of the studies (studies 2, 3, 6, 7, 8, 10, 13, 14, 16, 18, 23, 24). In almost half of the studies (10 out of 24), the agricultural inputs required to grow the biomass

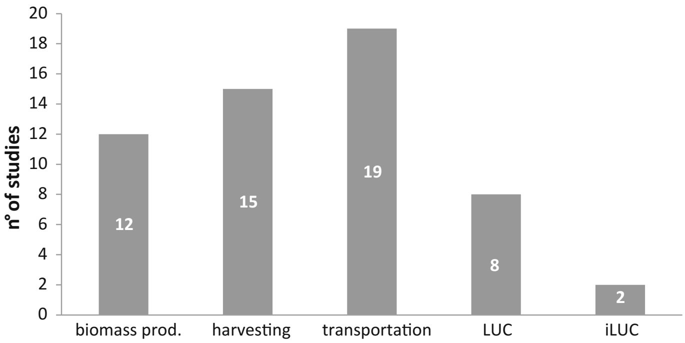

Fig. 9.2 Studies including the processes: biomass production, harvesting/collection, biomass transportation, land use change, indirect land use change

are not accounted for because they are assumed to be completely allocated to the main product (studies 1, 5, 9, 12, 15, 17, 19, 20, 21, 22). In the remaining two cases, the inclusion of cultivation phase is not possible: Ekman et al. (2013) implemented a biorefinery "gate to gate" analysis therefore the cultivation phase is excluded and Jeswani et al. (2015) considered forest residues where there is not a cultivation phase.

Biomass harvesting (if it is a residue) or collection (if it is a waste) is included in fifteen LCAs (studies 1, 2, 3, 5, 6, 7, 8, 10, 11, 12, 14, 16, 21, 23, 24). When it is not included the explanation is often that, as cultivation, this phase is allocated to the main product.

Transportation is generally included in reviewed LCAs. Excluding the "gate to gate" analysis (Ekman et al. 2013), very few of the studied assessments completely exclude this process (studies 9, 15, 17). In one case, transportation is not applicable since the study examines a modification of a treatment process and there is no biomass to be transported (Liu and Shonnard 2014). Often, the transportation phase is clearly detailed and information about distance, transport (road, train, or ship) and vehicle type is given.

Advanced biorefinery plants seem to be a good solution to overcome the GHG emissions associated with land use change (LUC) due to first-generation biorefineries (Cherubini et al. 2009a). Anyway, LUC effects can be induced also by agricultural and forest residue collection (Lal 2005). The removal of agricultural residues from fields may contribute to some concerns regarding soil quality, decrease in soil organic carbon (SOC), soil erosion, crop yields (Cherubini and Ulgiati 2010). Eight of the examined LCAs include LUC effects within the system boundaries. GHG emissions caused by the removal of residues from field are included in the climate change impact category by five studies (n° 1, 12, 18, 19, 21). Generally, the hereunder terms are included: (i) emissions due to SOC change (ii) nutrients replacement by synthetic fertilizer and (iii) decrease in N_2O emissions (Cherubini and Ulgiati 2010; Karlsson et al. 2014; Parajuli et al. 2017; Spatari et al. 2010). Piemonte (2012) does not state clearly how LUC effects are included. The three remaining studies include LUC

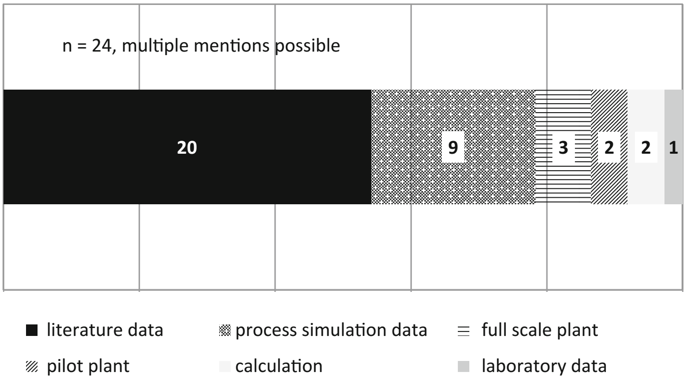

Fig. 9.3 Inventory data sources

effects only when dedicated crops are directly or indirectly involved in the system boundaries (Ekman and Börjesson 2011; Falano et al. 2014; Vaskan et al. 2017). Indirect land use change (iLUC) effects are very rarely analyzed. Parajuli et al. (2017) consider iLUC effects as the upstream consequences of occupying a productive land during the biomass production. Tonini et al. (2016) estimate the iLUC effects due to the diversion of agricultural residues from feed to bioenergy market.

None of the studied LCAs includes farm machinery and infrastructure of cultivation phase, construction, and decommissioning of the refinery phase. The exclusion of equipment and infrastructure is sometimes justified by its negligible contribution to total environmental impact owing to the long lifetimes of farm and industrial installations (Daful and Görgens 2017; Falano et al. 2014; Jeswani et al. 2015; Liu and Shonnard 2014). A further explanation is the poor and scarcely reliable information available (Liu and Shonnard 2014; Tonini et al. 2016; Uihlein and Schebek 2009). The studies which perform both environmental and techno-economic assessment always include the purchased and installed cost of equipment and facilities (Mandegari et al. 2017; Morales et al. 2017; Vaskan et al. 2017) but still environmental impacts are not accounted for.

9.5 Life Cycle Inventory Data

To the best of our knowledge, there are not LCAs of commercial waste-based biorefineries at present and most studies are based on conceptual designs. In the most of cases, life cycle inventory (LCI) of the reviewed studies is not based on primary data derived from experimental measurements but come from scientific literature and/or estimates (Fig. 9.3).

As it is possible to observe from Fig. 9.3, in nine studies (codes 2, 5, 6, 11, 14, 15, 16, 20, 24) life cycle inventories are based on process simulation software. These softwares are powerful tools, used in chemical engineering, to model and evaluate new processes and design plants. They allow LCA practitioners to obtain reliable energy and mass balances. Three reviewed studies report the use of data from full-scale plants but these data do not refer to the biorefinery but to the feedstock supplying (Ekman and Börjesson 2011; González-García et al. 2011) or to existing plant that could undergo integrations to exploit the new feedstock (Gilani and Stuart 2015). González-García et al. (2016a, b) present LCI for the foreground systems based on pilot plant. The first study analyzes a batch operation pilot plant with a capacity of 100 kg processed dried chips. The second study analyzes a pilot plant system with a feedstock flow rate of 336 kg/day. Finally, Nascimento et al. (2016), who study production of cellulose nanocrystal from coconut fiber, obtain data regarding cellulose nanocrystal extraction from measurements in laboratory.

9.6 Allocation Issue

The process of allocation is necessary when more than one product enters or leaves a process and the consumptions and the emissions of this process have to be partitioned among the products. Biorefineries produce several outputs so that it is necessary to allocate the impacts between them. Biorefineries based on residues present the issue to allocate both flows to feedstock production and to biorefinery output products (Ahlgren et al. 2015).

9.6.1 Feedstock Production

LCAs of biorefineries based on agricultural, forest, domestic, and industry residues have to solve the problem of how to manage the input and output flows related to the process leading to feedstock production.

LCAs applied to waste management systems often omit the environmental impacts from upstream life cycle stages prior to the collection. This approach is called the "zero burden assumption" (Ekval et al. 2007). In this respect, the "zero burden" approach is often applied also to waste and residue-based biorefineries. Where "zero burden" approach is not applied, typical allocation methods used in the reviewed studies include system expansion or are based on mass, energy content, and economic value (Fig. 9.4).

The "zero burden" approach is used when the feedstock does not have a market price since it is still considered waste or when it remains unutilized (e.g., agricultural residues left on-field). In case of agricultural residues or forestry industry, all the cultivation inputs required to grow the biomass are not accounted for because they are assumed to be completely allocated to the main product (e.g., wheat, timber, boards, etc.).

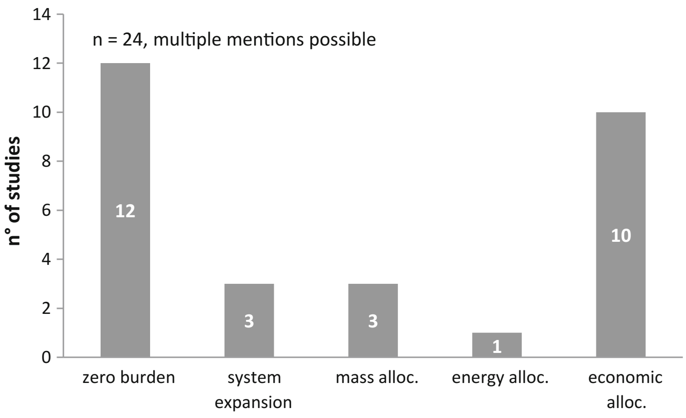

Fig. 9.4 Types of allocation of the feedstock production phase

As it is possible to observe from Fig. 9.4, the choice to apply an economic allocation is as common as the "zero burden" assumption. This approach is prevalent when residue has a market and therefore an economic value. Daful and Görgens (2017), Fazard et al. (2017) and Mandegari et al. (2017) propose the economic allocation among the co-products of a sugar mill (i.e., sugar, molasses, bagasse, and filter cake). Quite common is also the economic allocation of environmental burden between wheat and wheat straw (Jeswani et al. 2015; Mu et al. 2010; Parajuli et al. 2017; Uihlein and Schebek 2009). Ekman and Börjesson (2011) distribute on an economic basis the production load between biodiesel and glycerol and between potato starch and potato juice, and Vaskan et al. (2017) between palm fresh fruits and empty fruit bunches. All these residues have a market value or it is obtainable from the energy content (see Daful and Görgens 2017).

Mass and energy allocations are quite unusual and they are, generally, applied as alternative methods in sensitivity analysis (Ekman and Börjesson 2011; Pourbafrani et al. 2013). Only González-García et al. (2016b) apply mass allocation as main approach in a case where environmental burden of a sawmill activity has to distribute among bark, sawn timber and residual wood. The reviewed studies show that the economic value of main products and by-products differs significantly, which motivates the use of economic allocation as main allocation approach. Moreover, allocation by physical relationships (either mass or energy content) cannot reflect the underlying relationships between the co-products under an economic-value-driven multiproduct system.

The system expansion approach is put on when a technological innovation is integrated at an existing industrial process in order to valorize a process residue and the LCA comprises the whole system. Gilani and Stuart (2015) analyze the integration of an extraction pretreatment to a paper mill to obtain hemicellulose from wood chips. González-García et al. (2011) apply a LCA to a paper mill turned into a biorefinery producing also ethanol and lignosulfonates. Finally, Liu and Shonnard

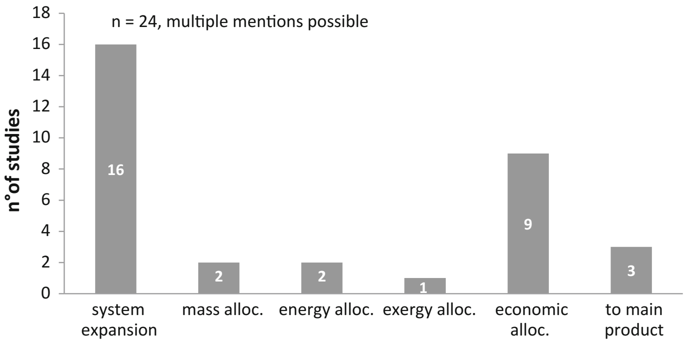

Fig. 9.5 Multifunctionality management approaches

(2014) examine an integrated production system between a forest hardboard facility and an ethanol biorefinery.

The two studies performing the sensitivity analysis show that the environmental performances of biorefineries are dependent on the allocation method applied (Ekman and Börjesson 2011; Pourbafrani et al. 2013).

9.6.2 Multifunctionality

All studies selected for this review are biorefineries combining energetic and material use of biomass to produce a spectrum of bio-based products and bioenergy. Therefore, all these studies deal with multifunctional processes and offer different approaches to solving multifunctionality problems.

As shown by Fig. 9.5, expanding the product system to include the additional functions related to the co-products is by far the most common approach. This is the approach applied by the majority of studies belonging to type 1 (i.e., focusing the biorefinery concept as a whole) and by the four consequential LCAs included in the review. In fact, while allocation is part of the traditional attributional (or descriptive) method, system expansion is part of the consequential (or change-oriented) LCA method that seeks to capture the environmental change caused by a certain activity, and to consider the effect on the whole system rather than on the single product under examination (Ekvall and Weidema 2004).

Liu and Shonnard (2014) and Nascimento et al. (2016) apply the mass allocation approach. The first study analyzes the use of mass allocation in a sensitivity analysis where the base case is the system expansion. Nascimento et al. (2016) applied both mass allocation and economic allocation since the proposed bio-products are still in the development phase with no consolidated insertion in the market.

Energy allocation is also rarely applied (Fazard et al. 2017; Pourbafrani et al. 2013) and that coherently with the different functions of products generated by biorefineries. Energy allocation is judged not appropriate when fuels, energy, and chemicals are produced at the same time.

González-García et al. (2016b) assess the application of exergy allocation in a sensitivity analysis where the base case is the total allocation to the main product (electricity). In this study, exergy-based allocation between the net electricity and the net heat produced is applied with 70 and 30% as partitioning factors for electricity and heat, respectively.

In five cases, economic allocation is the base approach and in four cases it is applied in the sensitivity analysis. This attests the relevance of this approach to biorefineries LCA analyzes. Since one of the most important indicators of a biorefinery is the economic benefit, the economic allocation is often the default method to partition the input–output flows and environmental burdens, according to the respective value and quantity of co-products. As stated by Ardente and Cellura (2012) despite the fact that economic allocation has potential limitations arising from the variability of prices and the low correlation between prices and physical flows, it is still suitable to illustrate the properties of complex systems, like the multiproduct biorefinery where the prices of co-products differ widely.

Three reviewed studies define a main product with the all inventory data assigned to it. In these cases the multifunctionality is not treated by system expansion and the co-products are ignored. Ekman and Börjesson (2011) do not include into the system boundaries the residues of glycerol and potato juice fermentation sent to anaerobic digestion to obtain digestate and biogas. González-García et al. (2016b) do not include in the base case the heat production in a CHP but just the electricity. Morales et al. (2017) do not consider the lignin contained in the solid phase of cellulose hydrolysis that is sent to an incineration plant for steam production. In all the three cases, the main product of the biorefinery is much more significant than the co-products.

It is noteworthy that seven studies carry on a sensitivity analysis on the multifunctionality management approach. General conclusion is that the choice of co-product allocation method significantly influences the life cycle assessment and each method presents disadvantages. According to Pourbafrani et al. (2013), system expansion may not be appropriate where co-products represent a significant production output or there is not a definitive primary product. Allocating impacts on the basis of market value is subject to volatility in product prices. The main drawback of energy allocation in the case of biorefinery operations is that not all products may be energy products. Mass allocation can be not proper when the prices of co-products differ widely.

9.7 Conclusion

The chapter provides a synthesis of the state of the art of LCAs applied to advanced biorefineries with the aim of investigating the methodological approaches. Only studies that apply life cycle analyzes, consider residues as feedstock and embrace biorefineries combining energetic and material use have been included in this analysis. Twenty-four LCAs have been selected. Functional units, system boundaries, inventory data collection, allocation methods and multifunctionality management approaches have been examined.

First, this review shows three main types of functional units: (i) related to the quantity of feedstock in input, (ii) related to the quantity of the main product in output, and (iii) related to the whole portfolio in output of the biorefinery. The choice of the FU is dependent on the study aim. In the first case, the aim could be to compare different processes or different feedstock used to produce the same group of products. In the second case, the focus is on the impact related to the main product of the biorefinery. In the last case, the aim is to assess the global performance of the biorefinery.

Second, the review focuses on the system boundaries, five processes are analyzed: feedstock cultivation, harvesting/collection, transportation, and GHG emissions due to land use change (direct and indirect). This issue is faced by the different authors in very different ways. It is evident that while harvesting and transportation are generally included, LUC and iLUC effects are ignored by most of the studies. Agricultural and forest activities often follow the "zero burden" approach and therefore are not included.

Thirdly, the review faces the problem of inventory data collection. An extensive use of literature data is evident since experimental data are few and they come from laboratory or pilot plants. Very interesting is the use of process simulation software to model and evaluate new processes and design plants.

Then, the analysis deals with the allocation issue of the environmental burdens related to the feedstock production. The "zero burden" approach results the most common and this methodological choice appears justified by the type of feedstock, residues and waste, often without a market value. On the other hand, many authors have decided to apply an economic allocation using the economic value of the feedstock—if any—or estimating it by its energetic value or future projections.

As far as the multifunctionality issue is concerned, the most common approach is to expand the system boundaries including all the co-products of the biorefinery. The system expansion is obviously used by consequential LCAs but often it is applied also in attributional ones. Quite frequent is the economic allocation among main product and co-products. It is interesting to note the use of sensitivity analysis in order to assess how results vary on the basis of multifunctionality choices.

On the basis of these results, while some methodological differences appear justified by the different aims of the studies, others appear to be the effect of different assumptions and they create inconsistency among the studies.

The large variability in methodology approaches highlights the need for a stricter harmonization to improve the comparability of these LCA evaluations. For example, despite the "cradle-to-gate" approach appears the most chosen, sometimes a "cradle-to-grave" one, including EoL, would have provided interesting information. Also the issue of environmental burdens due to the agricultural phase appears a very important topic to be faced and not always the choice seems due to the origin of the feedstock (agricultural/forest residue or waste).

The current analysis also highlights the need for robust methods to include some issue such as, for example, LUC and iLUC effects that are so important for bio-products but so little incorporate in LCA studies.

Finally, it is important to underline that an inherent limitation of this review is the limited number of analyzed key issues. Other issues should be addressed, for example life cycle impact assessment methods, sensitivity and uncertainty analyzes, attributional versus consequential approach.

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Chapter 10
Life Cycle Analysis of the Production of Biodiesel from Microalgae

Massimo Collotta, Pascale Champagne, Warren Mabee, Giuseppe Tomasoni and Marco Alberti

Abstract In recent years, there has been growing interest in third-generation biofuels, i.e., fuels from algal biomass. Considering microalgae, the production and transformation processes are currently under study by researchers across the world, as microalgae appear to be a promising alternative to meet our sustainability goals in the energy sector. Considering the Life Cycle Assessment (LCA) applied to biofuels from microalgae, a number of studies have been published to date, covering a wide geographical range and analyzing several process configurations. This chapter presents the microalgae-to-biofuel process and a review of the published LCA studies in the field. The findings show that the majority of these studies do not have access to primary data but only to secondary data sources. Most studies do not consider the whole process, but only some of the process stages, thus limiting the relevance of the results to the specific context to which they refer. Only about half of the studies reviewed consider the impacts of water and land use, and only two present a detailed analysis of the economic and social impacts. For this reason, further efforts are still necessary in order to obtain a comprehensive sustainability assessment of this potential solution to the energy problem.

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10.1 Introduction

It is a common belief that the development of green fuel technologies with low CO_2 emissions can help meet global energy requirements in a more sustainable fashion, reducing our over-reliance on fossil fuels, which currently meet 80% of the world's energy demand (Medeiros et al. 2013). In this context, growing evidence has illustrated the high potential for biofuels to improve the sustainability of the energy sector, especially for those countries and regions where fossil fuel availability is limited (Stephens et al. 2010).

For this reason, the exploitation of biomass for energy, and particularly liquid biofuels for use in transport, have been of increasing interest to policymakers, even though first- and second-generation biofuels, based on the use of crops, have received criticism (Crutzen et al. 2008), primarily associated with the use of energy crops and fertile land that generally lead to higher environmental impacts and to an increase in crops prices.

Among the different biomass feedstocks, microalgae has shown great potential as a sustainable feedstock for biofuels (also referred to as third generation biofuels), particularly for biodiesel, especially because microalgae are highly efficient lipid producers (Rickman et al. 2013; Leite et al. 2013). In particular, the lipid content of microalgae may reach up to 70% on an algal biomass dry weight basis mainly depending on species and cultivation conditions (Banerjee et al. 2002).[1]

Microalgal feedstocks have been investigated for different applications and products and several technologies have been proposed and investigated for the commercial production and transformation of microalgae (Grierson et al. 2013; Campbell et al. 2011). Nonetheless, the sustainability of the commercial production of microalgae-based biodiesel has yet to be proven, both from the environmental and economic point of view. The most promising directions that researchers have identified points to year-round cultivation, the ability to use wastewater as a nutrient source, higher solar energy yields and minimal use of arable land (Batan et al. 2011; Dismukes et al. 2008; Williams and Inman 2009). Moreover, it should be noted that microalgae can be cultivated in both salt and fresh water environments, and they are suited to areas where the cultivation of crops could be marginal, challenging, or expensive (Hiibel et al. 2015).

In this chapter, after having introduced the process for the production of biodiesel from microalgae and having analyzed the alternative technological pathways for the different steps of the process, we present a literature review on the environmental performance of microalgae in the production of biodiesel. The review highlights the lack of primary data and high production costs as the main weaknesses, while

[1]Chlorella Vulgaris, with standard Nitrogen fraction, has a lipid content of 175 g/kg with a low heating value of 17.5 MJ/kg.

a promising solution seems to be the use of co-products or by-products from other industrial processes.

10.2 The Microalgae-to-Biodiesel

The process for the production and exploitation of biodiesel from microalgae generally follows the scheme outlined in Fig. 10.1 and consists of seven main steps that can employ different technologies/chemicals/processes.

During cultivation, microalgae are grown in water (or wastewater) and supplied with nutrients, such as nitrogen and phosphorus, and a carbon source, mainly coming from inorganic CO_2. For microalgal cultivation, two alternative technologies have traditionally been employed: open ponds, i.e., shallow oval ponds exposed to air and light, which are likely to have lower operating costs, despite having higher net energy ratios and lower productivity rates (Collet et al. 2011; Chisti 2007); and photobioreactors, i.e., enclosed chambers for microalgal growth subjected to natural or (in northern climates) artificial light. These generally have higher operating costs and productivity. The use of wastewater throughout the process seems to be a promising manner to improve the environmental and economic sustainability of algae cultivation (Shrestha et al. 2013; Ficara et al. 2014; Ge and Champagne 2016). Similarly, flue gas from industrial sites (e.g., cement plants, power generation plants, etc.) has been evaluated as a potential source of CO_2 (Ge and Champagne 2016; Collotta et al. 2016). For this reason, the co-location of microalgal production facilities with wastewater treatment plants (or anaerobic digestion facilities), providing access to nutrients, waste energy and CO_2, could maximize the use of waste resources in an integrated resource management approach and increase the techno-economic feasibility of the overall process (Collotta et al. 2016, 2017b, 2018; Davis et al. 2016; Slade and Bauen 2013; Powers and Baliga 2010).

For the harvesting, which brings algae concentration from about 0.2% to about 20%, different pathways are also utilized. The most commonly considered is flocculation (stimulating the formation of solids flocs within the microalgal slurry) and centrifugation, and sometimes in combination (Collotta et al. 2017a; Lardon et al. 2009). The energy required for harvesting could be decreased through process innovations; for example, increasing PO_4 concentration in the growth medium can lead to a phenomenon called auto-flocculation in which the microalgae aggregate in flocs and then precipitate from the culture medium (Clarens et al. 2011). Other approaches have explored the harvesting phase, adopting the high pressure CO_2, without requiring the addition of coagulants, in order to separate algae from suspension (Lee et al. 2015).

Dewatering is an important stage as it is an energy-intensive process. This stage is often required to increase the percentage of algal biomass from about 20% to 90–95%, depending on the lipid extraction process requirements. A variety of technologies have been explored for this step including belt dryers, solar and steam dryers, natural

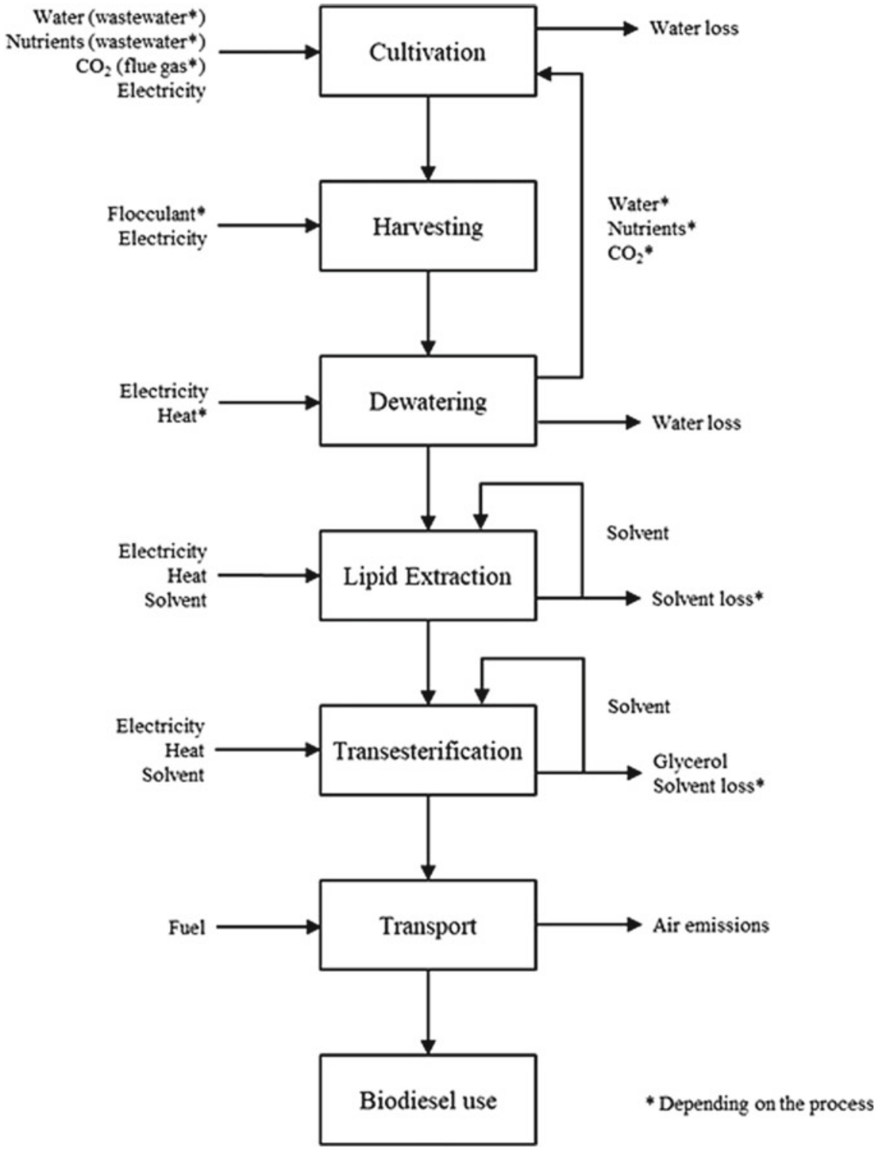

Fig. 10.1 Generalized scheme for the microalgae-to-biodiesel system

gas dryers and co-combustion with coal (Powers and Baliga 2010; Clarens et al. 2011; Lardon et al. 2009; Stephenson et al. 2010; Yang et al. 2011).

Different approaches are also used for the lipid extraction phase, the separation of lipids from the remainder of the biomass, which generally employ a solvent or co-solvent system, supercritical CO_2, and in some cases a prior or simultaneous cell disruption technique such as drill pressing, (Brentner et al. 2011) or dry degumming (Cox et al. 2014), microwave, sonication, freezing, etc. (Harris et al. 2018). More advanced approaches currently under exploration include the use of switchable hydrophilicity solvents (SHS) at room temperature (Boyd et al. 2012), the CO_2 expanded methanol approach (Paudel et al. 2015) or liquid CO_2, which present better lipid extraction yields (Paudel et al. 2015).

In the transesterification phase, lipids and alcohols are transformed into methyl or ethyl esters and glycerol. This reaction can be driven with esterification, sonication with a direct esterification and the Honeywell UOP™ process, which involves hydrogenation to produce synthetic hydrocarbons followed by selective hydrocracking and distillation (Brentner et al. 2011; Cox et al. 2014). Direct transesterification can also be adopted, which using supercritical conditions combine the lipid extraction and transesterification in a single phase with wet biomass (Brentner et al. 2011).

The transportation phase is the last step before biofuel usage and is generally implemented using trucks or pipelines, depending on the volumes produced and/or location of the plant. Production facilities should be placed at the most convenient location, for instance close to end users, close to the feedstock supply or close to a cement plant (as a source of CO_2) or a wastewater treatment plant (as a source of water and nutrients) (Stephenson et al. 2010; Powers and Baliga 2010; Collotta et al. 2016, 2018; Batan et al. 2016).

Finally, the end-use of the energy product is considered. Baseline comparisons between bio-based product (e.g., biodiesel, biojet) and their petroleum-based counterparts suggest that the impact of some substitutions—for instance, replacing coal-fired electricity—may lead to more significant environmental offsets than others (U.S. Energy Information Administration 2016). Understanding the end-use of the microalgae-based energy product is essential to understanding the overall impact of the system.

10.3 Literature Review of the LCA Studies on Biodiesel from Microalgae

The application of the Life Cycle Assessment (LCA) methodology to the production of biodiesel from microalgae is an ongoing endeavor, and the results obtained are affected by a high level of uncertainty, mainly because of the lack of large-scale production facilities and, consequently, because of the scarcity of primary data.

In this review, 24 LCA studies analyzing microalgae-to-energy systems have been identified and reviewed. Table 10.1 summarizes the main characteristics of these studies. As it can be seen, only 6 of the studies have used primary data for the life cycle inventory, while the majority have used sensitivity analyses to reduce the impact of uncertainty on the results. Nine studies evaluated systems in the European Union, but none are located in Italy.

Table 10.1 Main characteristics of the LCA studies considered

LCA paper	References	Year	Functional unit	Tool	Database	Country	Primary data	Sensitivity analysis
1	Baliga and Powers (2010)	2009	Production of 1 l of biodiesel	GREET 1.8a and crystal ball	n/s	USA		✓
2	Batan et al. (2011)	201 1	Production of 1 MJ of biodiesel	GREET 1.8c	n/s	USA		
3	Brentner et al. (2011)	2011	Production of 10 GJ of algal methylester	Simapro	Ecoinvent 2.2	USA		✓
4	Campbell et al. (2011)	2010	Transport of 1 t km	Simapro 7.0	ESAAE, AusLCI, Ecoinvent	Australia		✓
5	Clarens et al. (2010)	2009	Production of 317 GJ of biomass	Excel and crystal ball	Ecoinvent	USA		✓
6	Clarens et al. (2011)	2011	1 Vehicle kilometer traveled	Excel and crystal ball	Ecoinvent	USA		
7	Collet et al. (2011)	2010	Production of 1 MJ of electricity	n/s	Ecoinvent	EU	✓	
8	Collet et al. (2014)	2014	Combustion of 1 MJ of algal methylester	n/s	Ecoinvent	EU	✓	
9	Collotta et al. (2016)	2016	Production of 1 kg of dry algal biomass	Simapro 7.3	Ecoinvent	EU	✓	
10	Collotta et al. (2017a, b, c)	2017	Production of 1 kg of chlorella vulgaris	Simapro 7.3	Ecoinvent	EU		✓
11	Collotta et al. (2018)	2017	Production of 1 kg of lipids	Simapro 7.3	Ecoinvent	EU	✓	✓

(continued)

Table 10.1 (continued)

LCA paper	References	Year	Functional unit	Tool	Database	Country	Primary data	Sensitivity analysis
12	Cox et al. (2014)	2013	Production of 100 MJ of biojet	Simapro 7.3.3	Ecoinvent	Australia		✓
13	Gnansounou and Raman (2016)	2015	Production of 1 kg of biodiesel	Simapro 7.3.3	Ecoinvent	India		✓
14	Khoo et al. (2011)	2010	Production of 1 MJ of biodiesel	n/s	n/s	Singapore		✓
15	Lardon et al. (2009)	2009	Combustion of 1 MJ of fuel	n/s	Ecoinvent	EU		
16	Malik et al. (2015)	2014	Production 1 Mt of biocrude	MRIO	MRIO	Australia	✓	
17	Montazeri et al. (2016)	2016	Production of 1 kg of biodiesel	GREET	Ecoinvent 2.2 and US LCI	USA		
18	Slade and Bauen (2013)	2012	Production of 1 MJ of energy	Gabi 4	n/s	UK		
19	Stephenson et al. (2010)	2010	Combustion of 1 t of biodiesel	openLCA 1.4	Ecoinvent 2.2	UK	✓	✓
20	Soomro et al. (2016)	2016	Harvesting of 1000 kg of biomass	n/s	n/s	China		
21	Tsang et al. (2015)	2014	1 h of operation a the marine vessel	n/s	n/s	USA		✓
22	Yang et al. (2011)	2010	Production of 1 kg of biodiesel	Gabi 5	Ecoinvent 2.2	USA		✓
23	Holma et al. (2013)	2013	Production 1 MJ of biofuel	n/s	Different sources	EU		✓

As it can be seen from Table 10.2, the upstream process stages are included in the system boundaries in almost all of the studies considered (the cultivation phase is always considered), while the downstream stages are more frequently neglected, especially for what concern the use of the residual biomass, the transportation of the biofuel and its use. This result is particularly relevant, since a number of studies have shown the importance of the definition of system boundaries (Tillman et al. 1994).

The impact categories or indicators considered in the LCA studies under review are shown in Table 10.3.

Since the production of biofuels from microalgae is often cited as a solution to the climate change problem, (Medeiros et al. 2013) it is not unexpected that Global Warming Potential (GWP), which is representative of the combined emissions of several greenhouse gases (primarily CO_2, N_2O, CH_4), is quantified in almost all of the studies (22 of 23).

Within the biofuel life cycle, greenhouse gases mainly come from fossil fuel combustion for the generation of electricity and heat; the use of fuels for product transportation; and the manufacturing and use of chemicals in the process. Other GWP-related impact factors have also been noted in the studies under review, including energy use (10 studies), fossil resource depletion (6 studies), and abiotic depletion (consumption of natural but non-renewable resources—4 studies).

Greenhouse gas emissions are primarily related to energy consumption in the harvesting, dewatering/drying, lipid extraction, and transesterification phases. Some studies argue that harvesting and dewatering could contribute up to 20–30% of operational costs (Uduman et al. 2010; Grima et al. 2003), while other studies identified the lipid extraction and transesterification phases as having the highest energy demands (Khoo et al. 2011).

Given the relevance of energy use, it also has to be highlighted that the related impact factors change significantly with a changing energy mix (Itten et al. 2012), and thus depend on the country or region where the specific study is located. Moreover, it is evident that a microalgae-to-energy system that utilizes waste heat, or derives electricity from an onsite anaerobic digestion plant, could substantially reduce fossil greenhouse gases emissions and likely decrease overall operational costs.

With reference to the land and water requirements, also important when evaluating microalgal production systems, it has to be noted that water availability, which is dependent upon geographic location, is often the most critical issue affecting the feasibility of the process and its operational costs. While both fresh and salt water can theoretically be used; however, fresh water allows for a reduction in operational costs since, in the case of seawater, salts have to be extracted via processes such as evaporation, for example (Gendy and El-Temtamy 2013). In addition, sunlight potential and temperature of the location have been shown to influence the productivity of algae cultivation systems in terms of growth rate (Medeiros et al. 2013). The need for sunny days also defines the potential land requirements for algal production in outdoor environments, as in the case of open raceways ponds (Malik et al. 2015). In particular, in countries or regions with high land costs, open pond cultivation may become unfeasible, unless it can be located within existing industrial facilities.

Table 10.2 Production stages and processes included in selected LCAs

Process steps	Technology/chemical	LCA papers
Cultivation	Open raceway ponds (ORP)	1, 2, 3, 14, 18
	Photobioreactor (PBR)	3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14,15, 16, 17, 18, 19, 20, 21, 22, 23
Harvesting	Flocculation	3, 5, 6, 10, 14, 15, 18, 19, 20, 21, 22, 23
	Filtration	3, 10, 20, 21,
	Natural/gravity settling	6, 7, 8, 13, 23
	Mechanical press	15
	Dissolved air flotation	4, 13, 22
Dewatering	Centrifugation	1, 2, 3, 5, 7, 8, 11, 12, 13, 14, 17, 18, 20, 22, 23
	Dryers	1, 6, 11, 15, 17, 19, 22
	Homoginization	14, 18
Lipid extraction	Hexane (+methanol/ethanol)	1, 2, 4, 6, 11, 12, 13, 14, 15, 17, 18, 19, 22
	Supercritical CO_2	3, 22
	Other	3, 11, 12, 23
Transesterification	Methanol + ROH	1, 2, 3, 4, 6. 13, 14, 15, 17, 18, 19, 21, 22
	Methanol + Acid	2, 3, 21
	Supercritical methanol	3
	Honeywell UOP™	12
	Other	3, 16, 23
Residual biomass use	Anaerobic digestion, CH_4-energy	3, 6, 7, 8, 11, 12, 17, 18, 22, 23
	Animal feed	12, 17
	Soil amendments	6
	Landfill	3, 6, 22
	Other	9, 13, 22
Transportation	Truck	9, 10, 11, 18, 19
	Conveyor	2
Biofuel use	Biodiesel	3, 4, 6, 15, 18, 19
	Biojet	12
	Combustion/co-generation	6, 7, 23

Table 10.3 Impact categories utilized in reviewed LCAs

Impact category/indicator	LCA papers
Global warming potential	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14, 16, 17, 18, 19, 20, 21, 22, 23
Ozone depletion	7, 8, 9, 10, 11, 15, 19, 23
Human toxicity	7, 8, 9, 10, 11, 15, 19
Photochemical oxidation	1, 7, 8, 9, 10, 11, 15, 19
Ionizing radiation	7, 8, 9, 10, 11, 15
Acidification	1, 7, 8, 9, 10, 11, 15, 19
Eutrophifcation	3, 5, 7, 8, 9, 10, 11, 12, 15, 17, 19
Respiratory effects	19
Ecotoxicity	12, 19
Marine toxicity	15
Water use	1, 3, 5, 6, 12, 18, 21, 23
Land use	1, 3, 5, 7, 8, 9, 10, 11, 12, 13, 15, 23
Energy use	1, 2, 3, 5, 6, 12, 14, 16, 17, 20, 22, 23
Abiotic depletion	1, 7, 8, 9, 10, 11, 13, 15, 18, 19, 23
Life cycle costs	4, 20
Economic stimulus of microalgae-to-energy	16
Unemployment index	4
Full-time equivalent workers required (FTE)	16

Moreover, cultivation on arable land may raise concerns regarding impacts on food supply.

Although water and land use are clearly important impact factors to consider, only 13 of 24 studies under review included land use, eight examined water use and 12 considered eutrophication. This would suggest that water and land use are not monitored as regularly as greenhouse gas emissions, probably because of a paucity of data or a limited understanding of their importance.

10.3.1 Economic and Social Impact Assessment

One clear issue that emerges from the analysis of the LCA studies is that several process developments are still required for the production of algal biofuels to be economically viable. In fact, while many have speculated that biofuels from microalgae bring to environmental benefits, at the same time they have been presented to have a low economic feasibility due to the high costs associated with dewatering and lipid extraction (Campbell et al. 2011).

However, in any case, most studies have drawn from bench-scale operations, because of the absence of commercial facilities, and few of the analyses have assessed the potential economic impacts process scale-up. The integration of capital and operating costs would represent a key complement to the environmental impact assessment and, it would also be beneficial to consider the effects of specific policy measures, such as renewable fuel mandates, carbon pricing, or excise tax exemptions.

Three the 24 LCA studies reviewed have incorporated some economic considerations, and one in particular presented an innovative hybrid LCA model that integrated economic and social analyses along the supply chain (Malik et al. 2015). The life cycle costing methodology was used in one of the studies (Campbell et al. 2011), which defined a quite comprehensive model for tracking total production cost, including not only plant facility and main operational costs, but also items often neglected, such as the costs associated with research and development, design, failures, contribution margin loss, corrective and preventive maintenance and plant final disposal.

Other studies estimated the impact of increased or decreased water volume or arable land use on the production costs (Li et al. 2008; Borowitzka and Moheimani 2015) or the feasibility of using regional waste streams as resources (CO_2, wastewater and waste heat) for the algae cultivation. (Collotta et al. 2016).

With reference to the social impact assessment of biofuels from microalgae, two studies adopted their use as an impact category to track. In particular, one study (Brentner et al. 2011) examines employment through the unemployment index, while the other simply tracked the full-time workers required to operate the designed system (Malik et al. 2015). These studies suggested a higher number of employees for microalgae-to-energy systems compared to comparable food and nutraceutical production (10 employees) as well as conventional crude oil production facilities (29 employees). The implication is that the effect that microalgae-to-energy facilities may have on host communities, given the labor force demand derived by this plant, should be considered for a complete analysis. Although this is a first step towards the assessment of the social sustainability of biofuels from microalgae, to have a comprehensive and more reliable assessment, other factors need to be included, such as human rights, labor conditions and health and safety benefits, as well as corruption and their effects on the legal system (Ekener-Petersen et al. 2014).

10.4 Conclusions

In recent years, many advances have been achieved through the research and development of microalgae-to-energy systems. The LCA methodology, as an eco-design tool, can provide a relevant contribution to guiding this development towards sustainability direction.

Considering the state of the art regarding the application of LCA to biofuels from microalgae, one of the clearest evidences is the heterogeneity of the system boundaries adopted. In particular, the review highlighted a wide range of process configurations. Few of the LCAs currently published in this field consider the full range of process stages, most of them investigating five or fewer stages, with the most commonly omitted stages involving the transportation of biofuel to end users and end product use. Although such studies can give a relevant contribution in the specific context to which they refer, they are generally limited in contributing to evaluations of the environmental impacts of an integrated microalgae-to-energy scenario.

Another relevant aspect to highlight is that many LCAs have focused primarily on GWP (as measured via greenhouse gas emissions), while water and land use, highly significant in microalgal production systems, were not nearly as well quantified and analyzed in the selected studies. This is likely due to the fact that researchers have focused on the potential for microalgae-to-energy systems to meet global warming challenges. However, an important lack of primary and secondary data have been highlighted in these systems. Moreover water depletion remains an important topic to investigate for future commercial applications.

Finally, it should also be noted that, for the most part, published LCAs do not take into account the economic and social impacts of microalgae-to-energy systems. In fact, only two studies introduced some aspects of the economic and social benefits of biodiesel production. Although the integration of economic and social considerations in the sustainability assessment of microalgae-to-energy systems still presents a high level of uncertainty, due to their early technological development stage, a comprehensive sustainability assessment is crucial both to provide an impetus for the development and deployment of these technologies, and to give reliable profitability assessment.

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Chapter 11
Comparative Life Cycle Assessment Study on Environmental Impact of Oil Production from Micro-Algae and Terrestrial Oilseed Crops

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Abstract Global policies for reducing fossil fuel dependency and CO_2 emissions have fostered the development of low carbon sustainable energy. Since first generation biofuels may generate environmental burdens related to agricultural production, second and third generation biofuels from lignocellulosic feedstock and algae-to-energy systems have been developed. In this study, the Life Cycle Assessment methodology is applied to compare quantitatively, utilizing primary data, the impacts of the first generation in respect to the third-generation biofuels. Results show that micro-algae are neither competitive yet with traditional oil crops nor with fossil fuel. The use of renewable technologies as photovoltaics and biogas self-production might increase the competitiveness of micro-algae oil. Further investigations are however necessary to optimize their production chain and to increase the added value of co-products.

Keywords Biofuel · Sustainability · Oil crops · Micro-algae
Life cycle assessment

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11.1 Introduction

In the past few decades, the idea of using biofuels, mainly for transport use, has been developed in order to achieve several goals: (i) to reduce fossil fuel dependency; (ii) to decrease greenhouse gas emissions; (iii) to generate new employment and new sources of income for farmers. It is important to point out that the introduction of biofuels in the transport market and further progress towards low-emission technologies have been both driven by policy decisions, especially in the EU Directive EU 2015/1513. The application of various biomass feedstock, such as rapeseed, soybean, canola, corn and lignocellulosic crops as bioenergy source has been a common topic in the literature (Spinelli et al. 2013; Forte et al. 2015). Some recent publications (Forte et al. 2016; Zucaro et al. 2016) are referred to site-specific studies evaluating the environmental performance of biofuels, more often in comparison with fossil counterpart and/or among several biofuels products. However, the evidence that first generation biofuels (produced from edible parts of agricultural crops) can generate several environmental burdens, typically related to agricultural production (e.g., eutrophication, ecotoxicity, loss of biodiversity), and competition with food and land use change (Zah et al. 2007), has led to new solutions as second- and third-generation biofuels (Mamo et al. 2013), from lignocellulosic feedstock and algae-to-energy systems, respectively. Second-generation biofuels produced from non-food lignocellulosic crops, agricultural residues, or agro-industrial waste are considered more sustainable since they avoid land use change or competition with food crops. However, at the current state the production path to liquid biofuels from lignocellusic materials is still far from the technical and economical sustainability (Sims et al. 2010; Tabssum and Qazi 2017).

Accordingly, algae-to-energy systems are receiving great attention from both academic and industrial sectors. The narrative identifies several advantages in using micro-algae for bioenergy production, compared with conventional crops, such as:

- ability to be cultivated on marginal lands and therefore not incurring in land use change (Searchinger et al. 2008);
- semi-continuous to continuous harvesting;
- variable lipid content in the range of 5–50% dry weight of biomass;
- high exponential growth rates potential to utilize carbon dioxide (CO_2) from industrial flue gas (1 kg of dry algae biomass utilizes about 1.83 kg of CO_2) and nutrients (especially nitrogen and phosphorus) from wastewater (Chisti 2007; Cantrell et al. 2008).

For these reasons, they are an attractive feedstock for biofuel production (Malcata 2011). Moreover, some authors consider that micro-algae can be cultivated in mudflats or deserts where the carbon stock is close to zero, furthermore they could be an interesting alternative to energy crops which often lead to carbon stock losses through land use change (IFPRI 2011). Although many efforts have been made to optimize both the medium and processes parameters, the development of cost-effective and highly efficient cultivation systems must be significantly improved for large-scale industrial production (Brentner et al. 2011). According with forecasts of the International Energy Agency (IEA), world energy consumption is expected to increase by 53% between 2008 and 2035 (1.6% per year), stimulated in particular by the industrial and transport sector. Increasing demand for personal travel in the growing economies, freight and goods transportation system expansion along national and international routes are the main drivers of the utilization growth rate, which is expected to increase by 1.4% per year from 2008 and 2035 (IEA 2011). Algae may play a key role in producing biofuels (biodiesel, ethanol, methane, hydrogen) in view of depletion of fossil resources. Large research efforts, in recent years, have led to a variety of micro-algae based life cycle assessments (LCA) (Collet et al. 2013). Prior studies have shown that different algae harvesting options, reactor configurations, culture conditions, and cultivation assumptions yield divergent results concerning algae's environmental and energy performance. In any case, algae show higher environmental impacts than terrestrial crops in almost all the considered categories (Clarens et al. 2010). Many research efforts have been focused on this topic, among which the "EnerBiochem" project as a part of Italian National Operative Program (PON) for Research and Competitiveness, 2007–2013. The project aimed to study the feasibility of an integrated biorefinery, based on the opportunity of co-producing of biofuels together with bio-based chemicals, using marginal lands in the administrative scale of Campania Region (Southern Italy). The purpose of the project was also to identify an environmental and economical sustainable production for the development of a problematic region. Within the multidisciplinary framework and the several goals of this project, several biomasses (including micro-algae) have been considered as energy feedstock for the biorefinery. The results presented in this work are part of the activities performed inside the EnerBiochem project. It is an attributional LCA applied to satisfy two goals: (i) to evaluate the environmental hot spots in site-specific production chain of biodiesel from terrestrial oil seeds and micro-algae feedstock; (ii) to evaluate quantitatively, utilizing primary data, if the first generation of bio-fuels is environmentally unfavorable respect to the third generation.

Furthermore, the study explored the possibility to enhance the environmental performances of micro-algae oil through the application of renewable energies in the production process.

11.2 Materials and Methods

11.2.1 Description of the Analyzed Systems

11.2.1.1 Terrestrial Crops Oil System

This analysis has used average primary data of two crops (rapeseed and sunflower) grown in the years 2012–2014, using traditional farm practices, in experimental plots located in Campania Region (Southern Italy). The total cultivated area consists of

5 ha of flat land with sandy-loam soil texture, average annual rainfall 920 mm yr^{-1} and average annual sun insolation 10.8 MJ m^{-2} yr^{-1}. The two crops were cultivated in polluted marginal areas. Such areas, because of the adverse conditions for growing food crops were undergoing to a progressive abandonment. Experimental data relative to soil carbon storage are not presented in this study since, due to the short experimental period (3 years), they are poorly representative.

The same amount of N and K fertilizer was provided to both crops, while sunflower crop has required 100% more phosphorous and 52% more fossil fuel than rapeseed and a rescue irrigation of 280 m^3 ha^{-1}. Soil local N_2O emissions, due to N fertilization, were calculated by applying an emission factor (EF) of 0.8% measured in Mediterranean crops (Fierro and Forte 2012).

11.2.1.2 Micro-Algae Oil System

As reported in the literature, micro-algae biomass production using raceway pond shows a higher net energy ratio respect to the use of photo-bioreactors (Jorquera et al. 2010). Generally, open pond cultivation systems are the most frequently industrially applied because of their low cost of investment and operational capital. On the other hand, in more recent decades the development of different types of closed photo-bioreactors were considered and compared to open ponds; closed photo-bioreactors have increased photosynthetic efficiency and higher production of biomass (Wang et al. 2012; Darzins et al. 2010). However, the main problems for closed photo-bioreactors are the high initial cost, the maintenance operations and the specificity of strains (only micro-algae strains with particular physiologies can be used) (Harun et al. 2010).

Scenedesmus obliquus is a freshwater micro-alga that can grow in wastewaters of different origins showing good adaptability and it is widely used for outdoor cultivation and application for biofuels production (Hodaifa et al. 2008). Therefore, the algae strain *S. obliquus* has been cultivated in a raceway pond with the use of livestock wastewater as nutrient source. The choice of this specific strain was due to the capability of *S. obliquus* in purifying wastewater in order to minimize environmental impacts. Other strains should be preferred if the aim of the production is biofuels (increasing lipid content by cultivation under nitrogen starvation) (Lardon et al. 2009) or biogas (increasing carbohydrates content) (Baskar et al. 2012).

The quantities of materials required for cultivation and harvesting equipment, e.g., raceway pond, centrifuge, etc., were estimated to determine the environmental burden associated with the construction of the facilities. The lifetime of raceway pond and centrifuge were assumed to be 10 and 20 years, respectively. Livestock wastewater ($0.5\%_{v/v}$) was used as nutrient source instead of chemical fertilizers. After cultivation step, micro-algae slurry was sent to a flocculation step (recovery efficiency 88%). Natural illumination was used as light source for micro-algae growth. Micro-algae biomass was finally recovered by a centrifugation step (recovery efficiency 95%). All these treatments are high electricity consuming.

11.2.2 LCA Assumptions and Life Cycle Inventory Analysis

The LCA study was performed in accordance with the ISO 14040. The first LCA parameters that have to be defined are: (i) the functional unit and (ii) the system boundaries. The definition of such parameters should be subjected to the precise identification of the goal and scope of the analysis that in the case of this study is the comparison between oil production processes from micro-algae and terrestrial oilseeds crops for energy purposes. Therefore, the chosen functional unit should be the embodied energy (MJ) in 1 kg of produced oil.

As far as the system boundaries are concerned, a "cradle-to-gate" analysis was performed including a cultivation phase and oil extraction phase.

Primary data from the experimental plots of rapeseed and sunflower cultivated in Campania and from lab-to-pilot scale (100–3000 L) production of micro-algae (*S. obliquus*) carried out in the framework of EnerBiochem project form the basis for the life cycle inventory (LCI).

The oil extraction phase of terrestrial crops (via a chemical refining method) was referred to literature data (Figueiredo et al. 2012; Schneider and Finkbeiner 2013). Data from the literature were also used to determine the micro-algae oil recovery system by solvent extraction and the recovery system by a stripper column for separation of micro-algae oil/hexane stream (Stephenson et al. 2010). Inventory data are reported in Tables 11.1, 11.2 and 11.3.

The facilities for the oil extraction (buildings, machineries, etc.) are included in the system boundaries, but their inputs are negligible because of time spreading and utilization for other productions.

The cake in both crops was considered as substituted of soybean meal (avoided product), the most common source of protein for cattle breeding (D'Avino et al. 2015).

The oil content for the selected micro-algae strain is 5.2% (primary data). This figure is at the lower limit, as the lipid content is dependent on the growing conditions and ranges from 5 to 15% in a open pond, while reaches 25% in a photo-bioreactor under N-starvation. Data from literature were used to determine the micro-algae oil recovery system by solvent extraction and the recovery system by a stripper column for separation of micro-algae oil/hexane stream (Stephenson et al. 2010). The total heat requirement of the re-boiler was estimated $\sim$1.6 kJ kg^{-1} oil entering in the distillation column (hexane recovery > 99.5%). Electricity production is based on the Italian energetic mix and heat is produced with natural gas burned in industrial gas boilers (Ecoinvent Centre 2013).

11.2.3 Sensitivity Analysis

Sensitivity analysis was performed on micro-algae oil system changing the source of the most crucial parameter which is electricity consumption.

Table 11.1 Material and energy fluxes for the production of sunflower and rapeseed oil

Seed production		Rapeseed	Sunflower
Input			
Nitrogen fertilizer	kg	174	174
Phosphate fertilizer	kg	109	217
Potash	kg	96.2	96.2
Diesel	kg	47	61.9
Lubricants	kg	0.603	0.76
Water for irrigation	kg	0.00	2.80
Steel for agricultural machinery	kg	5.29	5.10
Seeds	kg	2.00	4.76
Products and by-products			
Seed produced	kg	4400	3850
Residues in field as such (dry matter)	kg	9300	6500
Refined Oil			
Input			
Seeds	kg	4400	3850
Water	kg	2090	2440
Bentonite	kg	9.71	8.90
Hexane	kg	4.56	4.18
Phosphoric acid	kg	1.47	1.30
Sulphuric acid	kg	3.61	2.44
Nitrogen liquid	kg	0.902	0.827
Charcoal	kg	0.361	0.331
Soda	kg	5.41	4.96
Heat natural gas	MJ	2940	2697
Electricity	kWh	174	159.87
Products and by-products			
Oil yield	%	41.0	43.0
Oil	kg/ha	1804	1655
Cake	kg/ha	2600	2195

Therefore, three alternative scenarios for algae oil productions were proposed

- Scenario 1—conventional electricity;
- Scenario 2—solar energy;
- Scenario 3—electricity from biogas produced by algae cake.

The energy source in the Scenario 1 is taken from Ecoinvent Database as "Electricity Medium Voltage Production IT, at grid" (Ecoinvent Centre 2013).

Table 11.2 Mass and energy flow generated by the production of 1 kg/m^2 day of algae

Input	Unit	Amount
Pond		
Concrete	kg	12,700
Steel (structure greenhouse)	kg	800
Copper (connection cables)	kg	18.4
PVC (pipeline connections)	kg	1320
PE (covering greenhouse)	kg	26,950
Electricity		
Air pumping	kWh	0.298
Nutrient pumping	kWh	1.06
Pump system		
Iron	kg	0.072
Fertilizer tank		
Concrete	kg	97.750
Nutrient		
N fertilizer	kg	0.029
K fertilizer	kg	0.02
P fertilizer	kg	0.011

Table 11.3 Mass and energy flow generated for harvest of 1 kg/day of microalgae

Input	Unit	Amount
Sedimentation/flocculation system		
Sodium hydroxide	kg	0.232
Centrifugation (centrifuge model MSE 220 V, 2 A, 150 mL/min)		
Steel	kg	0.032
Electricity	kWh	0.065

In the Scenario 2 the use of renewable energy as photovoltaic (PV) systems (single-SI panel) was investigated using data from Ecoinvent 2.2 database. These devices absorb incident illumination and produce a supply of electrons, which can be used by an external circuit with conversion efficiencies of up to 25 and 22% in small laboratory and full modules, respectively (Wenham et al. 1994; Green et al. 2012; Beardall et al. 2009). Therefore, electricity can even be produced from this same solar resource via the use of photovoltaic modules connected to the grid (Parlevliet and Moheimani 2014). Tredici et al. (2015) have shown positive results in terms of energy balance by integrating a PV system in the photo-bioreactor. The cake in both scenarios was considered as substituted of soybean meal (avoided product). The equivalent amount of avoided product was calculated as reported in the literature (Baliga and Powers 2010).

In the Scenario 3, the use of micro-algae cake for biogas production has been considered and was evaluated using data from literature (Collet et al. 2011). Biogas yield is affected by the composition of the algae biomass that in turn is partially determined by the algae growth conditions and by the biomass pretreatment (Sialve et al. 2009).

An investigation carried out with de-oiled micro-algae biomass obtained a biogas yield of 376 mL g^{-1} dry matter (DM) (0.376 m^3 kg^{-1}) from *Chlorella sp.* and 338 mL g^{-1} DM (0.338 m^3 kg^{-1}) from *Scenedesmus sp.*. Some authors reported in their extensive reviews (Scorupskaite and Makareviciene 2014; Ward et al. 2014) a methane yield of 240 mL g^{-1} VS (volatile solids) for *S. obliquus*. Sialve et al. (2009) summarized different experimental results in which methane yield varies from 0.09 to 0.45 L g^{-1} VS (0.09–0.45 m^3 kg^{-1}) depending on the species and culture conditions. Biogas purification is usually achieved by bubbling it into pressurized water. Electricity potentially produced at cogeneration with biogas engine is assumed to substitute fossil energy in the production process of algae biomass. Assuming a biogas production from anaerobic digestion of algae cake of 0.240 m^3 kg^{-1} VS in accordance with literature (Ward et al. 2014), the production of 3 m^3 of methane for 0.9 kg of algae oil has been calculated. As the electricity produced from 1 m^3 of purified biogas in a cogeneration unit is about 2 kWh (Piccini et al. 2007), only 20% of the electricity consumed in the micro-algae oil production process could be substituted from biogas-derived energy.

This scenario corresponds to a system expansion approach since the co-product (cake) is used inside the system boundaries.

11.3 Results and Discussion

11.3.1 Comparison Between Oil from Conventional Crops and Micro-Algae Oil

The comparison between oil from sunflower and rapeseed and oil from micro-algae was performed with the ReCiPe Midpoint Method (Goedkoop et al. 2012). A system expansion approach was applied in order to valorize the co-product (cake) and to minimize the low yield in oil of micro-algae. Results in Fig. 11.1, show that micro-algae oil production process has much higher environmental impacts compared with sunflower oil and rapeseed oil. The large impacts are due to the heavy energy demand (electricity and heat) and material consumption for the algae biomass production. A deviation from this trend is shown in the case of terrestrial ecotoxicity and freshwater ecotoxicity. In the case of terrestrial ecotoxicity the better performance of micro-algae oil is due to the big amount of avoided product (soybean meal) which possesses a high environmental burden. As far as freshwater ecotoxicity is highly influenced by the strong fertilization with phosphate as mentioned above. Moreover, water depletion

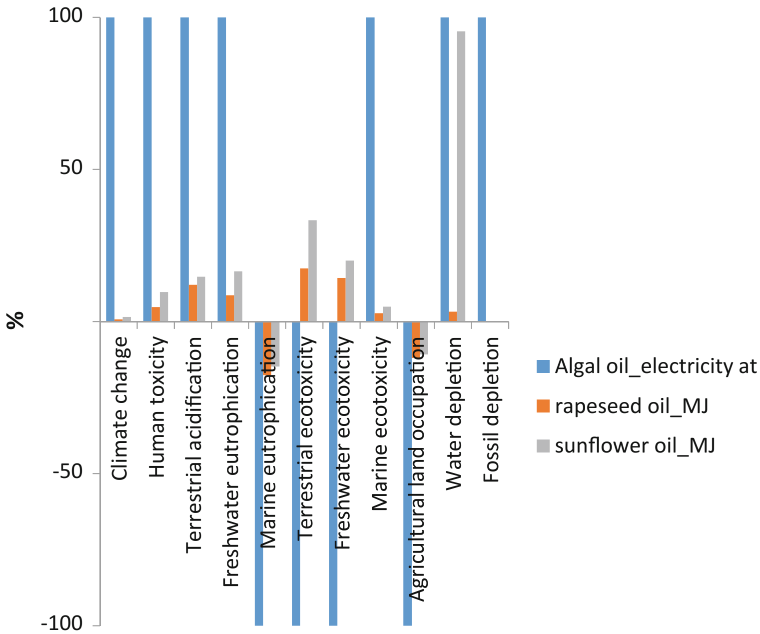

Fig. 11.1 Comparison among sunflower oil, rapeseed oil and micro-algae oil production processes

is affected by auxiliary irrigation, which in our primary data was made on sunflower only.

11.3.2 Sensitivity Analysis

The algae cultivation stage has the largest electricity requirement for air and nutrient pumping into the raceway pond, water pumping due to evaporation lost and pumping algae slurry for harvesting stage.

For these reasons, as previously discussed in Sect. 6.2.3, two alternative energetic scenarios have been evaluated besides the base case of Italian electricity mix (Scenario 1): use of photovoltaic technology (Scenario 2) and use of biogas produced from micro-algae cake (Scenario 3).

In Fig. 11.2 it is reported the comparison of the three scenarios calculated with ReCiPe Midpoint Method.

The scenario with photovoltaic energy seems to be the most environmentally convenient in almost all the impact categories while the scenarios with biogas and conventional electricity are similar in five categories even if biogas is the worst. The difference between biogas and conventional electricity increases in the impact categories where the influence of the credits from the avoided product "soybean

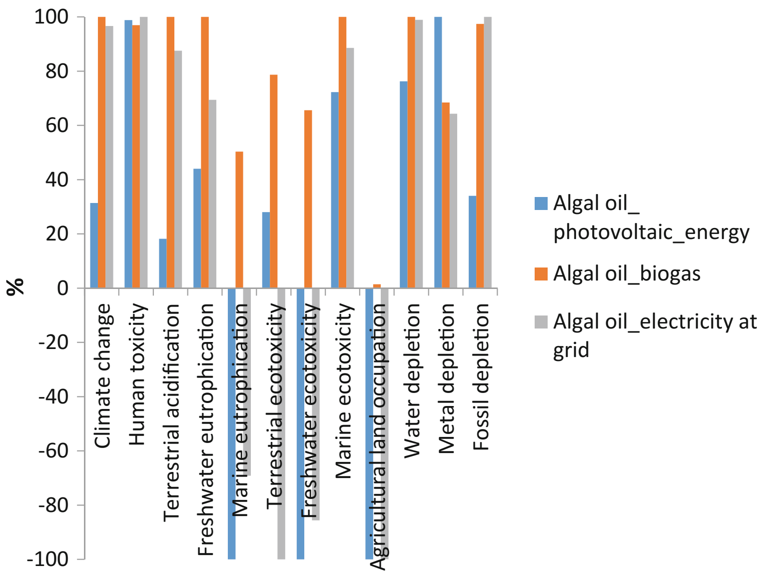

Fig. 11.2 Comparison among three scenarios for micro-algae oil production with different energy sources (electricity from biogas, photovoltaic, Italian electricity mix)

meal" highly affects the results: freshwater eutrophication, marine eutrophication, terrestrial ecotoxicity, freshwater ecotoxicity, agricultural land occupation.

A decrease of about 68% in climate change and 66% in fossil depletion can be calculated if the energy source "Italian mix" is substituted by photovoltaic system.

Exception to this trend is represented by the following impact categories: human toxicity and metal depletion, due to heavy metals and chemical reagents necessary in panel production technology.

Also in the context of sensitivity analysis, another key parameter to check at this point appears to be the avoided product. In fact the huge amount of residual biomass in the micro-algae oil production, which was considered as substitute of soybean meal, strongly affects many impact categories. For this reason Scenario 3, where the biomass was used to produce biogas was penalized. Therefore, the convenience of using the residual cake for biogas production or as animal feed was evaluated moving the residual cake out of the system boundaries, as compost for 1 and 2. In Fig. 11.3 the results of the comparison with ReCiPe Method are shown. It is evident that with these last assumptions the biogas scenario improves its position in the trend.

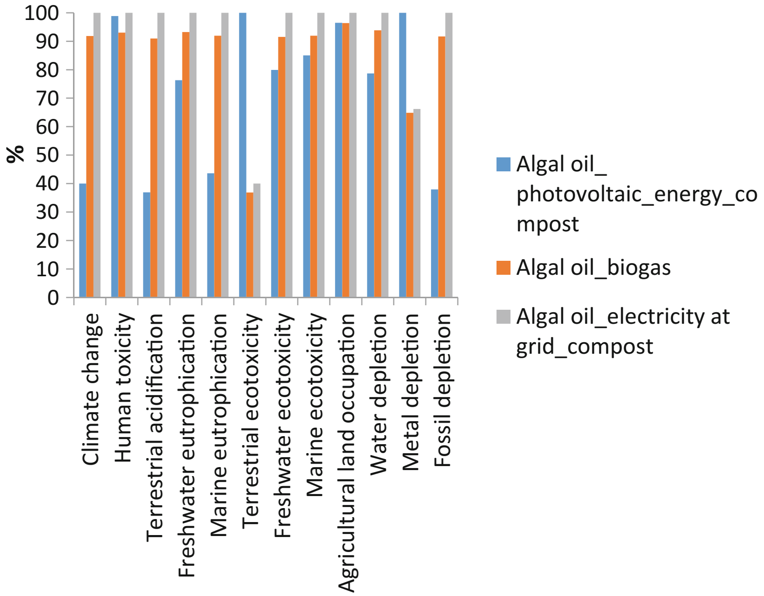

Fig. 11.3 Comparison of three scenarios for micro-algae oil production with different destination of residues for Scenarios 1 and 2

11.4 Conclusions

As reported in the literature, despite their high potential as sustainable energy feedstock, micro-algae are not yet competitive with the traditional oil crops in both economic feasibility and environmental impact (Reijnders 2008). The main achievement of this work is to have a confirmation of these findings by an LCA analysis based on primary data coming from a case study of an integrated biorefinery. A further important result is that the use of renewable technologies as photovoltaics and biogas self-production could increase the competitiveness of micro-algae oil reducing its demand of non-renewable energy sources (Dassey et al. 2014). In fact, this can reduce the costs of production, recovery, and extraction of oil from micro-algae, allowing a cheaper and more efficient production of biofuel or value added crops in remote locations far from sources of electrical power. The investment costs for the PV plant should be obviously assessed case by case.

The main hindrance to their application on industrial scale still consists on the high energy demand in terms of electricity, heat, and nutrients. Use of renewable energy in algae oil production chain has shown that there is a significant possibility to reduce its environmental impact. Even if this is still not enough to match the performances of terrestrial oil crops, the expected increase in world population resulting in growing need of arable land, will lead to favor second- and third-generation biofuels that do not compete with food production. From this perspective, algae could play an important role. Further investigations are necessary to optimize their production chain and to increase the value of all useful co-products as proteins, omega3 fatty acids, nutraceutics and other molecules suitable for pharmaceutical and cosmetic industry, which possess higher added value than biofuels.

Accordingly, in the future, these topics have to be approached by means of integrated and holistic methodologies, in order to evaluate the actual feasibility of bioenergy sources. In fact, many uncertainties still revolve around the technical and economic feasibility and the effectiveness of bioenergy to satisfy the energy demand of developed societies. These uncertainties and the complexity of the issue require multi-criteria studies to achieve representative results (Gomiero 2015).

The need to produce an integrated site-specific assessment is particularly obvious for complex production chain where natural capital (land availability, soil characteristics, solar input, water availability, and so on) as well as typical local human managements (mainly agricultural managements and new technological improvements) can affect the overall production chain.

For these reasons, the decision to build a biorefinery production system in a territory should be the subject of integrated evaluation with multi-criteria approach to obtain a more reliable picture of the system.

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