

Review

Life Cycle Analysis of the Bioethanol Production from Food Waste—A Review

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Abstract: Food Waste (FW) because of its composition is considered as an ideal feedstock for the production of biofuels and in particular bioethanol. The production of bioethanol from lignocellulosic materials has been studied over a long time. The process consists of the stages of pretreatment, enzymatic hydrolysis, fermentation and product recovery. However, the legal framework regarding biofuels has established specific environmental criteria for their production which are regularly updated. The most common tool for the assessment of the environmental performance of a process or product is the Life Cycle Analysis (LCA). In the present review, the results of LCA studies on the production of bioethanol from food waste are presented. Significant differences are observed among the studies in terms of the methodological choices made. Despite the high heterogeneity observed which does not allow a direct comparison among them, there is strong evidence that the production of bioethanol from food waste is an eco-friendly process which can substantially contribute to Green House Gas (GHG) emissions savings.

Keywords: biofuels; environmental impact; fermentation; sustainability; waste

1. Introduction

Nearly 1.3 billion tons of food products per year are lost along the food supply chain and this trend is expected to continue in the coming years according to Food and Agriculture Organization (FAO). Food Waste (FW) is produced at any step of the supply chain, from the agricultural production site to the processing plant and the consumption phase. Reduce, reuse and recycle, also known as the 3R's concept, is considered the best practice on which the management of food waste should be based [1].

However, the disposal at landfills remains one of the most popular options across the world. Food waste composition depends on its origin of production and may therefore comprise a mixture of carbohydrates (starch, cellulose, hemicellulose), lipids and proteins. According to Giroto et al., (2015), FW can be converted into biofuels or energy by the following processes: (a) transesterification of oils and fats for the production of biodiesel, (b) fermentation of carbohydrates for the production of bioethanol or biobutanol, (c) anaerobic digestion for the production of biogas (methane rich gas), (d) dark fermentation in order to produce hydrogen, (e) pyrolysis and gasification, (f) hydrothermal carbonization and (g) incineration.

Bioethanol is a typical green bioenergy source, preferable due to its renewability, non-pollution and its potential to replace gasoline in transport sector. It is mainly produced from edible materials such as sugarcane and corn. Use of agricultural land for growing energy crops is not recommended because it limits the land available for growing food crops, making bioethanol indirectly costly as compared to the cost of fossil fuels. Food wastes have the potential to serve as substrates for bioethanol

production, providing a viable solution in reducing Greenhouse Gas (GHG) emissions. Furthermore, bioethanol production from such wastes can reduce the cost of waste disposal as well as the cost of ethanol production [1].

The environmental and social benefits of exploiting waste in order to produce biofuels are reflected in the recently adopted legislation at international level as well as in the guidelines of international organizations such as the UN 2030 Agenda for Sustainable Development adopted in 2015 [2]. More specifically, biofuels' production from waste is associated with the Sustainable Development Goals 7, 12 and 13 namely Affordable and Clean Energy, Responsible Consumption and Production and Climate Action. In the European Union (EU), specific provisions for the production of biofuels from waste are included in the Renewable Energy Directive 2009/28/EC [3] and the recast of the Renewable Energy Directive known as RED II [4]. According to the European legislation advanced biofuels are defined as the 'biofuels that are produced from the feedstocks listed in Part A of Annex IX.' According to this piece of legislation, biofuels that are produced from the biomass fraction of municipal solid waste, the biomass fraction of industrial waste as well as bio-waste from households qualify as advanced biofuels and therefore count for a sub-target of 3.5% within the 14% target for renewable energy in transport in 2030. Furthermore, double counting of advanced biofuels towards the targets will continue. In the US, on the contrary, there are not detailed provisions concerning the feedstocks used for the production of biofuels. The focus is on the reduction of Green House Gases. More specifically, the Energy Independence and Security Act (EISA) biofuels has set GHG reduction thresholds compared to a 2005 petroleum reference for different types of biofuels. For example, a 60% reduction is required for cellulosic biofuels, a 50% for advanced biofuels from renewable biomass and a 20% for conventional biofuels. Low-carbon transport fuels are also supported by different states for example, Low Carbon Fuel Standards (LCFS) have been introduced in California and other North American regions [5].

Despite the differences observed at international level in the legislation concerning biofuels in general or more precisely cellulosic ethanol, there is a common ground: biofuels are assessed regarding their environmental performance compared to that of their fossil counter partners. The most common tool to assess the environmental impact of a product or process is the Life Cycle Analysis (LCA).

The use of lignocellulosic materials such as agricultural residues (corn stover, rice straw, sugarcane) as feedstocks for ethanol production has been extensively studied over at least 20 years. Therefore, numerous LCA studies exist which have been summarized in previous reviews [6,7]. The present study aims to fill the gap reviewing the LCA studies in the emerging field of the exploitation of food waste as feedstock for ethanol production.

2. Materials and Methods

Methodology—Principles of LCA

The present section is a brief introduction to the LCA methodology necessary for the non-expert in order to follow the analysis below. It does not intend by no means to provide a thorough description of the methodology and the way the LCA practitioners deal with the technical challenges they face. An LCA analysis is divided in different steps which are summarized in Figure 1.

Goal and scope definition is the first step of a LCA study, where the purpose of the assessment is established and decisions are made about the details of the product system being studied. This step aims at clearly defining goals in order to ensure that the analytical aims, methods, results and intended applications are optimally aligned. When goals are defined, identifying the intended applications and the degree of analytical depth and rigor of the study is critical. This should also be reflected in the defined study limitations. The scope definition of the LCA study includes the following elements: description and characteristics of the studied product, functional unit and reference flow, system boundary, assessed impact categories and related impact assessment methods, assumptions/limitations.

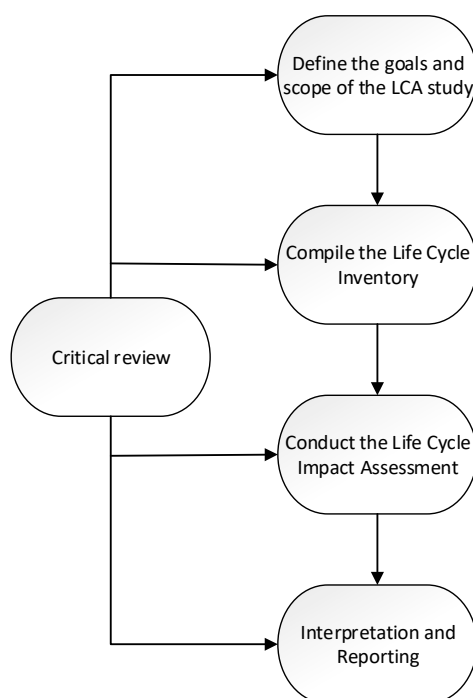


Figure 1. Phases of a Life Cycle Analysis (LCA) study (based on ISO 14040:2006).

Once the goals and scope of the study have been defined, the **Life Cycle Inventory** is prepared. The LCI is the compilation and quantification of inputs and outputs for a given product system throughout its life cycle. As a consequence, the LCI is the basis for calculating the potential environmental impacts of the analyzed product or process. It consists of detailed tracking of all the flows in and out of the product system, including raw resources or materials, energy by type, water and emissions to air, water and land by specific substance. It is important that all resource use and emissions associated with the life-cycle stages included in the defined system boundary are captured in the LCI. The following Life Cycle Stages may be included in the LCI (depending on the functional unit and the system boundaries that have been defined in the previous step): raw material acquisition and pre-processing; agricultural production; capital goods; production; product distribution and storage; use stage; logistics; end-of-life. In cases of “multifunctionality” that is, when a process or facility produces more than one function/good/“co-product,” the inputs and emissions linked to the process are partitioned between the product of interest and the other co-products in a predefined manner.

Based on the data of the Life Cycle Inventory, the **Life Cycle Impact Assessment (LCIA)** is undertaken in order to calculate the environmental burden of the product/process across the selected impact categories and models. The impact categories are classified in two groups: Midpoint and Endpoint. Midpoint indicators such as human toxicity, ozone layer depletion, global warming and eutrophication are oriented to the impacts. The impact captured in midpoint indicators can be further exploited to determine the damage categories in endpoint indicators. Endpoint indicators focus on the damage. They represent a quantification of the effects of the emissions on the object that should be protected: ecosystem, human health and resource availability. Table 1 presents a list of the most common impact categories found in LCA studies and their acronyms. The LCIA may be complemented with normalization and weighting but this is an optional step.

Table 1. Acronyms and definitions of the most common impact categories found in LCA studies.

Acronym	Impact Category
ADP	Abiotic Depletion Potential
AP	Acidification Potential
ED	Ecosystem Diversity
EP	Eutrophication Potential
FEP	Freshwater Eutrophication Potential
GHG	Green House Gas emissions
GWP	Global Warming Potential
HH	Human Health
HTP	Human Toxicity Potential
LUC	Land Use Change
MEP	Marine Eutrophication Potential
ODP	Ozone layer Depletion Potential
POP	Photochemical Oxidation Potential
TEP	Terrestrial Eutrophication Potential

The final step in an LCA analysis, the **Interpretation of the results** of the LCA study serves two purposes:

- to ensure that the LCA model corresponds to the goals of the study and fulfils its quality requirements.
- to generate meaningful conclusions and recommendations, for example, implementation of technological changes which will lead to environmental improvements.

It is during the Interpretation of the results when the most relevant impact categories, life cycle stages, processes are identified. In this case most relevant refers to those having the biggest environmental impact, usually reaching 80% cumulatively.

3. Results & Discussion

A literature search has been performed using as keywords the words “waste” + “LCA” + “ethanol.” However, the category waste is quite broad and vague so the results retrieved were further refined in order the definition of waste to be narrowed to food waste (which comes either from industrial plants or from households). Finally, 16 case studies concerning LCA for bioethanol production using different lignocellulosic materials (which are classified under the above definition of waste) were selected. Table 2 summarizes the key issues and general considerations of each study. From the information provided in the table, it is clear that a direct comparison between the selected studies is not possible because they vary widely in most of the components of an LCA analysis.

3.1. Analysis of the Studies Concerning the Functional Unit

A biofuel functional unit is often used in order to assess the environmental impact for the conversion of waste to bioethanol via fermentation. But even in this case, there is no single functional unit which could allow a comparison. Other studies use as functional unit 1 TJ bioenergy [8], others 1MJ ethanol [9,10] or 1kg ethanol [11]. When the focus of the study is on the ethanol production technologies then a volumetric unit of ethanol (i.e., 1L or 1 ton) is preferred as the functional unit [12–14]. A blended unit such as E85 (a 5% blend of ethanol with 85% fossil-based gasoline) is also found in the literature since bioethanol is often blended with gasoline [15]. However, the focus of a study may not be the production of ethanol but the management of the waste (municipal or industrial). In this case, different functional units such as 1 ton wet biowaste [9], 1 ton MSW [16] or even the total amount of waste in a given area [17] may be used.

Table 2. Main characteristics considered in the reviewed publications.

Reference	Feedstock	Functional Unit	Process	System Boundaries	Impact Categories	Key Impacts
[13]	palm oil frond	1 ton of anhydrous bioethanol	transportation, milling, juice extraction, pretreatment, fermentation, EtOH purification	gate-to-gate	ADP, AP, EP, GWP, ODP, HTP, FEP, MEP, TEP, POP	Conversion of OPF petiole juice to bioethanol could potentially generate high negative impacts to all the evaluated categories.
[9]	biowaste	1 ton of municipal wet biowaste, 1MJ ethanol	Pretreatment, hydrolysis, fermentation	Cradle-to-grave	GHG, eutrophication, toxicity, PM	−15 kg CO ₂ eq/ton biowaste compared to the current waste treatment methods. Sensitivity analysis conducted for investigating the impact of increased enzyme dosage to the overall environmental performance of the system showed that, the increased ethanol production due to increased enzyme dosage has a smaller impact to the system environmental performance compared to the effect of increased enzyme dosage.
[5]	MSW	1 MJ of liquid biofuel (butanol and ethanol), 1 ton MSW treated	Steam pretreatment, hydrolysis, fermentation, product recovery	Cradle-to-grave	GHG	GHG emissions results vary from −566 gCO ₂ eq/MJbiofuel (under US policies that employ system expansion approach) to +86 gCO ₂ eq/MJbiofuel and +23 gCO ₂ eq/MJbiofuel (under initial and current EU policies that employ energy-based allocation). Significant contribution of downstream wastewater treatment to GHG emissions. Increased acidification impact because of chemicals in pretreatment. Net negative emissions may be obtained by E65 blend in Ecuador.
[18]	Lignocellulosic waste from banana packaging plant	1MJ of energy released during ethanol combustion in a passenger car	Simultaneous saccharification fermentation with steam explosion pretreatment	Well-to wheels	GWP, AP, EP	
[19]	brewery waste	74.22 tons of lignocellulosic stream	Reconditioning and storage, autohydrolysis pretreatment, XOS purification, fermentation and bioethanol purification	cradle-to-gate	AP, EP, GWP, ODP, POP, HTP, FEP, MEP, TEP	Two environmental hotspots identified: the production of steam required to achieve the large autohydrolysis temperature (responsible for contributions higher than 50% in categories such as acidification and global warming potential) and the production of enzymes required in the simultaneous saccharification and fermentation (>95% of contributions to terrestrial and marine aquatic ecotoxicity potentials).
[14]	bagasse	1 ton bioethanol	Pre-treatment, enzyme generation and SSCF, ethanol purification and recovery and evaporation units	cradle-to-gate	ADP, GWP, AP, EP, ODP, POP, TEP, FEP, MEP, HTP	All scenarios assessed have environmental benefits over the combustion of bagasse in the sugarmill.

Table 2. Cont.

Reference	Feedstock	Functional Unit	Process	System Boundaries	Impact Categories	Key Impacts
[10]	HFW and agricultural residues	1 MJ bioethanol (99.7% bioethanol)	Modelling of bioethanol production following the approach of Tonini et al. (2015)	cradle-to-gate	GHG	GHG EFs ranged from −639 for household food waste to −1 g CO ₂ eq./MJ for maize stover compared to fossil fuels.
[20]	sugarcane	The functional unit (f.u.) for Well-to-Tank (WtT) LCA is 1 ton of sugarcane and for Tank-to-Wheel (WtW) f.u. is 1 km of car operation in the case of ethanol (vs. gasoline)	Juice extraction, Hydrolysis, Fermentation, Distillation, Cogeneration	Well-to wheel	climate change, fossil depletion, human toxicity, freshwater toxicity, freshwater eutrophication	All evaluated scenarios demonstrate positive values of Climate change and Fossil depletion reduction as compared to the reference systems. However, it shows less efficiency in Human toxicity, Freshwater ecotoxicity and Freshwater eutrophication impacts for “only fuels” scenarios.
[21]	Food processing and retail waste	1t waste	Simultaneous saccharification fermentation	Cradle-to grave	GHG	Negative GHG emissions and almost 500% improvement (compared to corn ethanol production).
[15]	Citrus waste	Functional units: 1MJ of E85, 1 kWh of generated electricity utilizing biomethane, 1kg of limonene and 1kg of digestate	Acid hydrolysis and fermentation (removal of inhibitor compounds (limonene), AD of residuals)	Well-to wheels	GHG	134% reduction in GHG with the use of E85 compared to gasoline. Significant savings resulting from on-site electricity generation and fertilizer displacement if the ethanol biorefinery is integrated with biogas production.
[16]	MSW	1 ton of MSW	Selective hydrolysis of cellulose fraction of MSW, fermentation and distillation	Cradle-to gate	HH, ED	Ethanol production proves to be the best alternative to avoid human health and ecosystems diversity impacts.
[12]	MSW	1L of denatured ethanol produced in Washington State	Dilute acid enzymatic hydrolysis and fermentation	Cradle-to grave	GWP, AP, EP, smog air, PM	Significant contribution of acid and enzyme production for pretreatment to energy consumption and acidification potential. High degree of uncertainty in the impacts of enzyme production.
[22]	Banana pulp, fruit, flower stalk and peel	Net energy analysis of a plant capable of processing 4000 kg/day of banana fruit and its residual biomass	Dilute acid and enzyme hydrolysis, fermentation, distillation	Cradle-to grave	NEV, ER	Energy ratio of 1.9 for fruit and pulp estimated, slightly higher than ER for corn ethanol. Low ER when fruit was co-fermented with cellulosic residue.

Table 2. Cont.

Reference	Feedstock	Functional Unit	Process	System Boundaries	Impact Categories	Key Impacts
[17]	Household waste: 1. Refuse Derived Fuel (RDF) and 2. Biodegradable Municipal Waste (BMW).	total amount of waste treated in the integrated waste management system/ MJ of fuel equivalent	integrated waste management system, taking into account recycling of materials and production of bioethanol in a combined gasification/bio-catalytic process.	cradle-to-gate/ cradle-to-grave	GHG	Bioethanol from RDF—this saves up to 196 kg CO ₂ eq. per ton of MSW, compared to the current waste management practice in the UK.
[23]	MSW	15 dry MMT MSW available for converting to ethanol in California	Dilute acid, prehydrolysis, enzymatic hydrolysis, fermentation	Cradle-to grave	GHG, LUC	A complete MSW-to-ethanol facility in California would displace 110PJ of fossil energy with a slight increase in GHG emissions. Landfilling of lignin residue is recommended over incineration to achieve improved GHG benefits.
[24]	MSW	1 ton of wet MSW treated; 1km distance travelled	Selective hydrolysis of cellulose fraction of MSW, fermentation and distillation	Cradle-to grave	GHG	At an ethanol yield lower than 166L/ton, MSW-to-ethanol conversion results in higher emissions than landfilling with LFG recovery. Higher well-to-wheels emissions for ethanol than gasoline, corn ethanol and lignocellulosic ethanol.

3.2. System Boundaries Analysis

Concerning system boundaries, these may include all upstream and production inputs and outputs to produce a given quantity of ethanol to the (a) factory gate, (b) the distribution network or (c) combustion of fuel ethanol in vehicles depending on the study. A generic simplified flow chart for a fermentation process which is the most common process to convert waste to ethanol is shown in Figure 2.

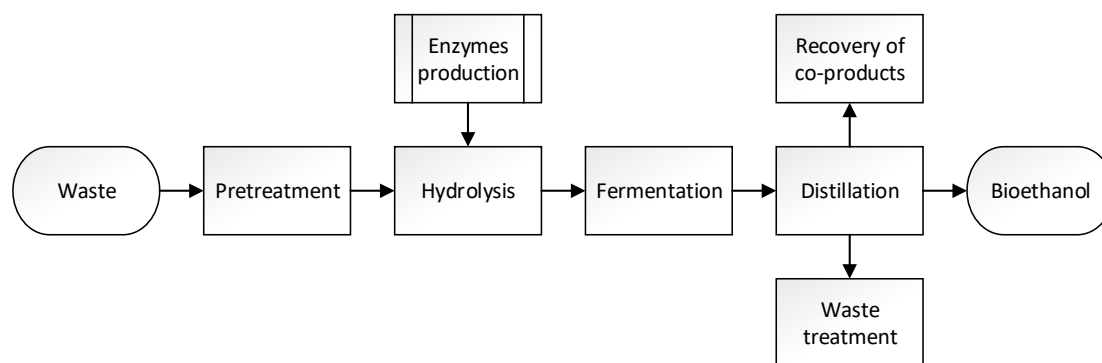


Figure 2. Simplified diagram of the bioethanol production process.

As it has been demonstrated in previous studies, the production of bioethanol from waste is heavily dependent on the composition of the waste [25,26]. The composition of waste not only defines the final ethanol production but it also affects the specific processes contributing to the total environmental burden. The first step of the process is a pretreatment which aims to reduce particle size and increase surface area. For example, household food waste may contain more readily available sugars for the production of ethanol than industrial waste. Thus, the pretreatment stage is not very demanding in terms of energy (heat) or additional materials (acid, alkali, organic solvents etc.). The same is true for some industrial food wastes with high sugar content, for example, palm oil frond [13], banana waste [18], brewery waste [19] and fruit syrups [21], which can be utilized for the production of ethanol without energy-intensive pretreatment processes. Once the pretreatment step has been concluded, the hydrolysis step follows. In that step relative enzymes (amylases, glucoamylase, cellulases, hemicellulases) hydrolyse carbohydrates namely, starch, cellulose and hemicellulose into simple sugars. Enzyme production is an energy and steam intensive process [27,28]. As far as enzyme production is concerned, glucoamylase production has been shown to contribute the highest toward GHG emissions, when a cradle-to-gate approach is considered [29,30]. Such an analysis (cradle-to-gate) comparing the environmental output of the production of five enzymes used in ethanol fermentation was conducted by Novozymes, the largest industrial supplier in the world [27]. The GHG emissions in glucoamylase production are attributed to the electricity or energy requirements. On the contrary, the impacts for cellulase are associated with the chemicals and nutrients used during the production. However, the environmental impact of cellulase in the production of lignocellulosic ethanol is higher given the doses of the enzymes used: 25–250 times more doses of cellulase compared to amylase and glucoamylase [29]. González-García et al. reported that 20% GHG emission produced all over the life cycle of bio-ethanol produced in a brewery waste-based biorefinery are attributed to enzymes and chemicals required [19]. Papadaskalopoulou et al., reported that enzyme contribution in the global warming impact (GWI) of biorefineries producing ethanol range from 11 to 62% due to high variation of the reported GWI of enzymes, different enzyme loadings and ethanol yield [9]. Additionally, cost of enzymes is an important element of the operational costs. A possible strategy to reduce this cost and make lignocellulosic ethanol cost effective is the production of the required enzymes on-site instead of using those commercially available [25,31–33]. Hong et al. estimated that enzyme GHG emissions are

258 g CO₂ eq. L⁻¹ of ethanol for on-site production, while for off-site production the relevant value is 403 g CO₂ eq. L⁻¹ [34].

The fermentative conversion process to ethanol can be carried out with two different process configurations, namely, separate hydrolysis and fermentation (SHF) and simultaneous saccharification and fermentation (SSF). SHF is a traditional strategy in which the feedstock is subjected to hydrolysis followed by fermentation of the monosaccharides to ethanol in a different bioreactor. On the contrary, hydrolysis and fermentation steps can be combined in a process known as SSF. An advantage of SHF is that enzymes and microorganism can operate at their optimal conditions, for example temperature. However, the main disadvantage of SHF is the accumulation of hydrolysis products which inhibit enzyme action resulting in decreased reaction rates. In SSF, temperature is not optimal for enzymes and, therefore, the rate of hydrolysis is slow but SSF presents the following advantages compared to SHF: low production cost, shorter processing time, limited reactor volume, higher ethanol productivity, lower enzyme load, limited enzymatic inhibition due to simultaneous end product removal [35]. According to González-García et al., the environmental hotspot in the profile of bio-ethanol production by brewery waste is associated with the SSF stage, mostly due to the use enzymes production of which have the key role in all the categories evaluated [19]. Ethanol concentration in the fermentation broth should be higher than 4% (*w/w*) in order to decrease the energy demand during distillation [36,37]. In order to reach this ethanol benchmark, the amount of sugars released in the hydrolysis step should be at least 80 g/L. In consequence, loadings of more than 15% solids are needed during enzymatic hydrolysis [38]. High solid loading result in increased viscosity due to difficulties in mixing, heat and mass transfer limitations. Fed-batch process could potentially limit those problems and lead to increased ethanol yields in high solids SSF. Another way to overcome high solids associated problem is pre-hydrolysis and Simultaneous Saccharification and Fermentation (PSSF) process, which is a variation of the SSF process, in which the substrate passes through a brief pre-hydrolysis step at optimum temperature for enzymes followed by fermentation [39]. During fermentation CO₂ is produced which is considered biogenic and according to the IPCC (2006) the biogenic CO₂ emissions have a global warming potential of zero [9].

Downstream processing of ethanol requires solid separation and purification which can be analysed in distillation and dehydration. Distillation is considered as an energy-intensive process and accounts for approximately 40% of the energy demand of the process [40,41]. Normally, two different columns are used in the distillation process: a stripper column, in which the ethanol is separated from solids and non-volatile compounds and a rectification column, where ethanol is concentrated near to the azeotropic point. The following methods can be applied for the production of anhydrous ethanol: adsorption, azeotropic distillation, chemical dehydration, diffusion distillation, extractive distillation, membrane process and vacuum distillation [35]. As mentioned above, obtaining a high ethanol concentration in the distillation feed is crucial. In a starch-based process the ethanol concentration is normally above 8% (*w/w*) while in a lignocellulose-based process the aim has been to reach at least 4–5% (*w/w*) ethanol in the stream entering the distillation step [37]. Ethanol yield of 30 g/L was achieved using thermally pretreated MSW (at 160°C for 30 min) in a fed-batch SSF process using amylases, at 25% (*w/w*) substrate loading [42]. On the other hand average ethanol concentrations of 16.5–22 g/L for enzyme loadings of 5 and 25 FPU/g dry solid have been reported when wet oxidized organic MSW (mainly source-sorted kitchen waste) enriched with wheat straw was used as substrate [43]. Much higher ethanol production (53.90 g/L) was achieved with two horizontal rotating bioreactors (HRRs) operating in fed batch mode at high solids content. In this case, cellulases and glycoamylase were supplemented for the saccharification [44]. Moreover, an ethanol concentration of 43 g/L has been reported via the application of SHF in FW [45]. On the other hand, concentrated FW hydrolysate fermented by immobilized yeast cells resulted in 89.28 g/L ethanol [46].

Solids after ethanol distillation can be transformed into various coproducts. The byproduct of bioethanol fermentation starch-rich grains such as corn, wheat and barley is known as distillers' dried grain with solubles (DDGS). DDGS because of its rich composition is used as aquaculture, livestock

and poultry feeds [47]. In lignocellulosic based ethanol, the excess solids can be used for heat and electricity cogeneration or pellets' formation improving the economic output of the process [37]. FW due to low lignin content offer less opportunity for energy coproduction. A possible way of using the solids after fermentation is anaerobic digestion. Antonopoulou et al. reported that direct anaerobic digestion (AD) of FW (FORBI) led to lower overall energy recovery, compared to that of AD following fermentation [48].

In general, the use of efficient cogeneration systems combined with optimized bioethanol production process may lead to energy savings. Cogeneration systems, such as the Rankine Cycle with condensing steam turbines and those based on gasification technologies, may increase the electricity produced, thus allowing its sell to the grid [49]. According to Ensinas et al., the electricity production could be significantly increased using sugarcane bagasse and trash as fuels, mainly with steam demand reduction [50]. With the international goal to decrease CO₂ emissions, this fact is gaining importance.

3.3. Impact Categories Analysis

Concern on global warming and climate change as well as the relevant policy regulations defining the limits of acceptable emissions dictate the impacts categories that analysts usually consider. As expected almost all the analyzed studies consider GHG emissions and related impact categories such as climate change or GWP depending on the software used for the analysis. It should be noted here that remarkable differences concerning the calculation of the GHG emissions are observed among the studies. These differences may be attributed to some extent to the approach selected which in some cases is dictated from the geographical origin of the study [5]. The EU in general is in favor of the allocation approach which has been applied in the frame of the RED and the recast RED in order to establish the GHG limits [3,4]. According to ISO 14040:2006 allocation provides a solution to multi-functionality problems by "partitioning the input or output 4 flows of a process or a product system between the product system under study and one or more other 5 product systems." On the contrary, via system expansion the system is expanded in order to include additional functions related to the co-products. This approach (concerning the production of ethanol from waste) is mostly used in the USA. The difference is evident in a case study which uses mixed MSW for the production of butanol and ethanol: GHG emissions estimations vary from -566 gCO₂ eq/MJ biofuel (under US policies that follow the approach of system expansion) to +86 gCO₂ eq/MJ biofuel and +23 gCO₂ eq/MJ biofuel (under former and current EU policies based on energy allocation), relative to the fossil fuel comparator of gasoline (equal to +94 gCO₂ eq) [5]. However, even in the case that allocation is selected over system expansion, the different possible types of allocation of the environmental burdens between the co-products may have a strong impact on the results obtained. For example, economic allocation is usually selected in LCA studies applied in biorefineries [19,20]. The same approach was followed by Mandegari et al. [14] but in this case a sensitivity analysis was performed and showed that choosing energy over economic allocation decreases the environmental impact for most of the categories assessed. Moreover, lower GHG emissions for the production of ethanol from citrus waste were reported by Pourbafrani et al. when energy allocation was chosen over economic allocation [15].

Regardless of the approach used, production of bioethanol from food waste seems to have environmental benefits in terms of GHG cuts. For example, Tonini et al. estimated a reduction in GHG emissions of -639 CO₂ eq./MJ bioethanol produced from household FW compared to fossil fuels [10]. According to Papadaskalopoulou et al. The production of bioethanol from biowaste results in -15 kg CO₂ eq/ton biowaste compared to the current waste treatment methods [9]. It is interesting to mention here an assumption commonly found in LCA works that study the production of energy from waste. A characterization factor of 'zero' is assigned to biogenic emissions meaning that the CO₂ which is released during the combustion of biofuels does not count in the GHG emissions. It is obvious that the system boundaries of the analysis are highly important in order to assess the potential impact of any process. For example, in a cradle-to-gate approach the use phase is excluded from the analysis [10,14] whereas in a cradle-to-grave approach all the steps of the life cycle of the analysis and

thus the selected characterization factor becomes relevant [17,23,24]. Another aspect of LCA studies that include the End of Life step is associated with the carbon incorporated into a landfill or soil and can be treated as sequestered leading to negative GWP impacts. For example, Stichnothe et al. mention that bioethanol with 100% biogenic carbon would save 107% CO₂ equiv. compared to petrol [17]. In any case any attempt to compare the GHG emissions results of different studies must begin with ensuring the consistent accounting of CO₂.

However, there are cases that present negative environmental impacts regarding the production of bioethanol from food waste. According to Yusof et al. who studied the possibility of producing bioethanol from oil palm frond sugar juice in a theoretical biorefinery, the bottleneck of the process was the fermentation step because of the nutrients used [13]. This finding highlights the need for a careful selection of the feedstock used as well as for optimization in all the steps of a process.

Apart from the GHG emissions and the relevant impact categories, 50% of the studies assess midpoint impact categories such as AP, EP, TEP, MEP which allow a more detailed assessment of the system [9,12–14,18–20]. These studies show strong evidence regarding the environmental benefits of the production of bioethanol from food waste. Only one study is based on endpoint categories assessing the impact on human health and ecosystem diversity providing in this way an overview of the system [16]. According to this study ethanol production is the environmentally the best option among the possible alternatives for the management of waste.

The studies that have been analysed show results with different level of aggregation. Only in few case studies the relevant stages of the life cycle—the ones carrying the biggest environmental burden— are identified. According to Schmit et al., chemicals production and waste collection are highly contributing to the overall environmental performance of the system [12]. In particular, the production of the relevant enzymes as an energy and steam intensive process has a significant impact [9,19].

4. Conclusions

LCA studies concerning the production of bioethanol from food waste present a high heterogeneity in terms of the composition of the feedstock used as well as in terms of the elements of the LCA making a direct comparison extremely difficult. Significant differences are observed concerning the functional unit of each study, the system boundaries and the assumptions made. Regarding the impact categories assessed, most studies focus on the GHG emissions in line with the general concern on climate change and the legislative provisions for biofuels without excluding other impact categories. As a general conclusion, it could be said that food waste is a promising feedstock for the production of bioethanol with significant environmental benefits.

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