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## **Beyond carbon and energy: The challenge in setting guidelines for life cycle assessment of biofuel systems**

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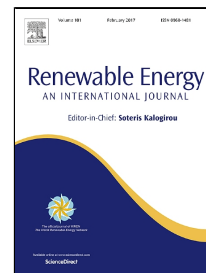
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# Accepted Manuscript

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**Highlights:**

1. More than one functional unit should be used whenever possible
2. System boundaries should be expanded to include co- and by-products.
3. Climate, acidification, eutrophication, land use, and energy are the key indicators
4. Existing frameworks for biofuels LCA focus on greenhouse gases emissions

1 **Beyond carbon and energy: the challenge in setting guidelines for life cycle assessment**  
2 **of biofuel systems**

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7  
8  
9 **Abstract:**

10 Life cycle assessment (LCA) is a most suitable tool for a uniform assessment methodology of  
11 sustainability of biofuel systems. However, there are no binding guidelines for LCA of biofuel  
12 systems. Published LCAs use a range of methodologies, different system boundaries, impact  
13 categories and functional units, various allocation approaches, and assumptions regarding by- and co-  
14 products, as well as different reference systems to which the biofuel system is compared. The  
15 European Renewable Energy Directive and the US Renewable Fuel Standard focus on greenhouse gas  
16 (GHG) emissions. However, previous LCAs of biofuel systems have shown that a reduction of GHG  
17 emissions does not lead automatically to a decrease in other environmental impacts, and might in fact  
18 be associated with an increase in impacts such as acidification, eutrophication, and land use change.  
19 In order to enable effective comparison of biofuel systems, the authors propose a framework for  
20 biofuel LCA. System boundaries should be expanded to include the life cycle of by- and co-products.  
21 Results should be reported using more than one functional unit. Burden shifting can be avoided by  
22 considering an array of impact categories including global warming potential and energy balance,  
23 along with eutrophication and acidification potential, and a land use indicator.

24  
25 **Keywords:** life cycle assessment; sustainability guidelines; system boundaries, impact categories,  
26 biofuels.

27  
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## 29 **1. Introduction**

### 30 **1.1 Background**

31 Total world energy consumption is predicted to increase by 56% between 2010 and 2040 [1]. The EU  
32 has set an overall mandatory target of 20% share of renewable energy in gross domestic consumption,  
33 and a target of 10% share of renewables in transport (RES-T), by 2020 [2]. The US Energy  
34 Independence and Security Act of 2007 requires 18% of transport fuel consumption to come from  
35 renewables by 2022 [3]. The use of biofuels, biogas or bioliquids as a sustainable and efficient energy  
36 source has been explored for decades [4,5]. For some time, especially in the context of peak oil,  
37 biofuels were considered a reliable renewable energy source, but their overall sustainability has been  
38 questioned in recent years [6], leading to increased focus on methods for its evaluation. Life cycle  
39 assessment (LCA) is one of the most promising approaches for evaluating the sustainability of  
40 biofuels [7] and for comparing biofuel systems. LCA is a multistage approach, which covers the full  
41 life cycle of a product. While LCA is an important tool for understanding biofuel systems, and  
42 supporting decision making, LCA procedures could be considerably strengthened in a number of key  
43 areas [8]. These include: i) methodology with goal and scope definition, ii) data collection based on  
44 real-life rather than lab scale facilities, and iii) multi-criteria of LCA that considers all  
45 environmental impacts equally. These challenges are described in more detail below.

46  
47 Goal and scope of the study, like functional unit, are often not clearly defined; sensitivity analyses are  
48 rarely performed [9]. Thomassen et al. stated that there is a lack of a generic framework for  
49 sustainability assessment of algal biofuels [10]. LCA studies of the same biofuel by different authors  
50 can give significantly different results, which can lead to the introduction of contradictory policies  
51 [11]. Such discrepancies are largely due to different methodological approaches and, although the  
52 need for a standardised methodology has been identified, there are as yet no guidelines for achieving  
53 this [8,10–13]. Moreover, many LCA studies do not specify the methodological choices made, which  
54 makes it almost impossible to replicate the study [14]. LCA addresses the life cycle of produced and  
55 consumed product [15], and is largely used by industry to assess a real-life product system. Advanced  
56 biofuels are not yet deployed on a large scale and industry data might not be available. Therefore,  
57 often LCA is based on laboratory data which makes the results much less accurate [10,16].

58 A further weakness is that most existing LCA studies have focused only on energy and/or greenhouse  
59 gas (GHG) emissions. Reinhardt pointed out that majority of studies called LCA are in fact only GHG  
60 emissions and energy balances [9]. Lazarevic and Martin analysing LCA and sustainability studies in  
61 Sweden found that GHG emissions dominate other environmental impacts [17]. Ridley et al. assessed  
62 more than 1600 peer-reviewed papers on biofuels and found that the most discussed topics are  
63 production technologies, GHG emissions and agricultural production of substrates, whereas the  
64 impact of biofuels on biodiversity and human health was much less investigated [18]. This is in line  
65 with Raman et al. who argue that impact on human health and resources are understudied, when it

66 comes to biofuel sustainability assessment [16]. Also, water, land use and land use change are seldom  
67 found in the literature [14]. GHG emissions and Global Warming Potential (GWP) are very important  
68 for LCA studies [2]. However, there are concerns that, although the use of biofuels may decrease  
69 GHG emissions, other detrimental environmental impacts, such as acidification, human toxicity, or  
70 land use change, may increase and should therefore also be taken into account in the LCA [17,19,20].  
71 However, the inclusion of additional environmental impacts is hindered by the fact that some of the  
72 key parameters, such as indirect land use change or water use, are currently not well understood or  
73 lacking a transparent and mature calculation methodology [11,13,17,19].  
74

## 75 **1.2 Aims and Objectives**

76 Previous papers stressed a lack of guidelines for LCA of biofuels, and a need to propose a common  
77 framework [8–10,12,13,19]. With the overall goal of bridging this gap and assisting policy makers to  
78 make informed decisions based on solid scientific evidence, the aim of this paper is to investigate and  
79 make recommendations for improving the robustness and accuracy of biofuel LCA framework. The  
80 objectives of this paper are to:

- 81 • Discuss the existing frameworks providing guideline for LCA of biofuels
- 82 • Compare various approaches used in biofuel system LCA in terms of functional unit, system  
83 boundaries, reference scenario, allocation methodology and impact categories;
- 84 • Discuss how extensive the evaluations should be in order to produce sound results;
- 85 • Make recommendations for improving the robustness of the biofuel LCA framework.

## 87 **2. Methodology – construction of literature source database**

88 A literature search was conducted in Science Direct, Research Gate and Google Scholar, using both  
89 “biofuel” and “LCA” as keywords. This includes: life cycle assessment, analysis and approach.  
90 Initially, 54 papers were selected, these were published between 2006 and 2014 (Fig. 1). Out of this  
91 initial trawl, 16 papers were retained as these papers met the criteria under investigation in this paper.  
92 Papers were excluded based on repetitiveness (in terms of system boundaries and subject of studies),  
93 and scope and aim of the studies (e.g. studies focusing on infrastructure such as LCA of anaerobic  
94 digester; LCAs of biofuels different that biodiesel, bioethanol and biomethane; LCAs with an  
95 incomplete scope, missing one of the following: functional unit, scope, impact categories, allocation).  
96 Similarly, reviews, methodological papers and cost analyses were excluded. Thus the initial 54 papers  
97 were reduced to 16 papers for detailed investigation. These 16 papers were selected in such a way that  
98 three biofuel systems (biodiesel, bioethanol and biomethane) are represented, and also under a  
99 condition that each discussed at least four of the five headings in Table 1, namely: functional unit,  
100 system boundaries, reference system, allocation, and impact categories. Biogas was included as an  
101 energy vector that is very close to biomethane. A diverse range of substrates were selected from crops

102 (rapeseed, palm oil, maize and grass), and from residues (tallow, used cooking oil, food waste,  
103 manures and straw). Third generation substrates were also selected (micro and macroalgae).  
104 Additionally, to validate the focus of biofuel LCAs on GHG emissions a simple calculation was  
105 carried to evaluate the percentage of peer-reviewed LCAs according to impact categories assessed.  
106 These were classified into six groups: GHG or GWP, energy, both GHG and energy, other  
107 environmental impacts, land use and water. 39 papers were considered, which were published after  
108 2008. These were screened for impact categories assessed (Fig. 5).

109

### 110 **3. LCA of biofuels- results**

#### 111 **3.1 Existing LCA frameworks**

112 LCA norms ISO (International Standard Organization) 14040 and 14044 set a general framework for  
113 LCA of any goods or services [15,21]. Functional unit should reflect the function of the products;  
114 system boundaries and impact categories should stay consistent with the goal of the study; choices  
115 should be clearly stated and explained; in terms of allocation, ISO 14044 allows for any of the  
116 allocation methodologies, provided that the choice is explained and fits into the scope of study;  
117 however it is recommended to avoid allocation whenever possible by 1) dividing processes into sub-  
118 processes and collecting data related to sub-processes; or 2) by expanding the system (substitution)  
119 [15]. PAS (Publicly Available Specification for assessment of life cycle GHG of goods and services)  
120 follows ISO 14044 in terms of general guidelines, but concentrates only on GHG emissions [22]. The  
121 UNFCCC (United Nations Framework Convention on Climate Change) through its MethPanel  
122 discusses only the issue of allocation and agrees partially with ISO, postulating that all allocation  
123 procedures can be justified [3]. However, the International Energy Agency (IEA) in its BIOMITRE  
124 manual (BIOmass-based Climate Change MITigation through Renewable Energy) states that  
125 substitution may increase the complexity of the study, and therefore advocates in favour of allocation  
126 by market value or physical relationship, such as mass volume or calorific value [23]. BIOMITRE  
127 also gives more detailed recommendations suggesting that the FU should be mass or volume or  
128 energy content. It gives guidelines and examples for biofuels' system boundaries. The tool focusses  
129 only on carbon emissions as an indicator. This is in line with the EU European Renewable Energy  
130 Directive (RED), which limits the allocation possibilities to allocation by energy content (lower  
131 heating value) and sets a MJ of fuel as the FU [2]. RED is applied in the BioGrace Excel tool [24].  
132 BioGrace is a harmonised tool that can be used in a European context for life cycle GHG calculation  
133 of biofuels. The system boundaries are pre-set but there is a possibility to create new entries. The  
134 calculation includes also direct land use change emissions.

135

### 136 3.2 Functional unit

137 At the early stage of any LCA, the functional unit (FU) of product must be defined. The FU is a  
138 quantified description of the product system performance [15,25]. For biofuel, the product function  
139 might be the provision of fuel for transportation, or the processing of particular feedstock. Cherubini  
140 and Strømman [13] distinguished input- and output functional units. The FU can be expressed per  
141 mass of input substrate, such as one tonne of dry seaweed [26]. However, a majority of studies used  
142 an output-related FU, typically expressed in MJ of energy generated from a given feedstock [27,28],  
143 or in kilogram of produced fuel [29]. The EU RED recommends a FU of a MJ of fuel [2]. As the  
144 primary function of biofuel is to provide vehicle fuel, a commonly used FU is kilometres driven by a  
145 car or a truck transporting a given mass of freight [30,31]. For land-based biofuels, the most relevant  
146 FU was identified as land area under a crop [13].

147 The impact of FU was investigated by Lettens et al. [32]. Low-input energy crops were compared  
148 with traditional energy crops and it was found that if land surface was used as FU, conventional crops  
149 performed better in terms of GHG emissions, while if GJ of energy from crops was used as FU, then  
150 low-input crops performed better. Thamsiriroj and Murphy [33] [34] also explored the impact of  
151 choosing different FU, and came to contradictory conclusions when using GHG emissions per GJ or  
152 per ha as FU (Fig. 2). To better understand the findings of a LCA, Cherubini and Strømman [13]  
153 recommended using more than one FU. However LCA studies that present results using several FU  
154 are seldom found, and there is a lack of guidance on the appropriate selection of multiple FUs.

155

### 156 3.3 System boundaries and reference system

157 The boundaries of the system must be defined together with the goal and scope of the LCA. Processes  
158 and flows should be listed to consider which should be included in a LCA. Usually, there is more than  
159 one product delivered to the market from the same system. In such a case, the system boundaries  
160 might be expanded to include the life cycle of co-products, by-products, and residues [35]. Early  
161 LCAs of bioenergy systems did not include the life cycle of co- or by-products, thus giving a poor  
162 impression of biofuel systems environmental performance [31].

163

164 Some of the reviewed studies considered more than one system boundary [33,36–38]. However, only  
165 three take into account the use of biofuel in a vehicle, a boundary expansion, which can have a  
166 considerable impact on the results. Korres et al [39] found that when use of grass biomethane in a car  
167 is included (well to wheel), the GHG savings are 18% lower than for well to tank analysis. Luo et al.  
168 [30] reported that expanding boundaries to include car driving as well as food and fodder production,  
169 led to lower GWP but higher impacts in other environmental categories (Table 1).

170

171 The reference system for a biofuel is typically a fossil fuel system delivering the same service, thus  
172 having the same function. The EU RED sets a reference value based on fossil fuels. This value

173 represents the actual average GHG emissions from petrol and diesel within the EU that is set at 83.8 g  
174 CO<sub>2</sub> eq / MJ for both fuels. Several of the reviewed studies follow the RED recommendations to  
175 calculate GHG reduction. Thamsiriroj and Murphy [34] used diesel as a reference system for biodiesel  
176 and Kaufman et al. [40] used gasoline for bioethanol.

177 The choice of reference system is not always straightforward. If biomethane displaces natural gas (as  
178 might be the case in a country with a high penetration of NGVs), then the savings in displacing  
179 natural gas (at 50.3 g CO<sub>2</sub> eq / MJ natural gas) [41] appear significantly less than if displacing petrol  
180 or diesel (at 83.8 g CO<sub>2</sub> eq / MJ), as might be the case in a country with a low penetration of NGVs.  
181 Generally, the reduction in GHG emissions will be higher if biofuel systems are compared with  
182 carbon intensive fuels, such as coal, and lower if compared with ‘cleaner fuels’, such as natural gas. In  
183 the majority of studies, the boundaries of the reference system did not extend beyond the production  
184 and use of fossil fuel [33,34,39,40,42]. However, Börjesson and Berglund [43,44] defined boundaries  
185 that included also the production of mineral fertilizers and alternative uses of the raw materials or  
186 land. By expanding the system’s boundaries, Börjesson and Berglund [43,44] detected potential  
187 indirect benefits derived from biogas systems linked to changes in handling of feedstocks and  
188 digestate.

189

### 190 **3.4 Allocation methodology**

191 If process chains deliver more than one product, all system flows must be divided between different  
192 products delivered by the system. This division procedure is called allocation whereby all flows are  
193 weighted and divided between the products of the system in proportion to the products’ energy  
194 content, mass or market value (Fig. 3). Another approach is subdivision, in which multifunctional  
195 processes are sub-divided into sub-processes, and separate data are collected for each mono-functional  
196 process [45]. No-allocation approach is the most conservative; all burdens are assigned to the main  
197 product.

198 The third major approach is system expansion, in which the boundaries of the system are expanded to  
199 include the functions and life cycles of co-products. System expansion and substitution are often used  
200 as synonyms. However, while the former approach is only about expanding boundaries, the latter  
201 considers all products and/or functions that can be replaced by the co-products and by-products of the  
202 system under analysis. For example, digestate as a co-product of a biogas system can be used as  
203 fertilizer and therefore the system gets credits for reducing the use of mineral fertilizers [46].

204 Despite the fact that choosing an allocation methodology is a fundamental step in LCA, different  
205 organisations recommend different approaches (Table 2) and several studies found that different  
206 allocation approaches can lead to completely different results [33,40,42].

207

### 208 **3.4.1 Comparing biodiesel and grass biomethane**

209 Thamsiriroj and Murphy [33] used three different allocation approaches to analyse rapeseed biodiesel,  
210 tallow biodiesel, UCO (used cooking oil) biodiesel, and grass biomethane. They found the biggest  
211 difference when assessing tallow biodiesel, with GHG savings varying between 33% (no allocation)  
212 and 150% (substitution approach). In the case of grass biomethane, GHG savings increased from 54%  
213 (no allocation) to 129% (system expansion). The simplest system, UCO biodiesel, seemed the least  
214 affected by the choice of the allocation method.

### 215 **3.4.2 LCA comparison of rapeseed biodiesel system using various allocation approaches**

216 Stephenson et al. [47] considered that an allocation based on direct substitution was the most  
217 appropriate for rapeseed biodiesel system (scenario 2; Fig. 4). However, if the product being replaced  
218 is a by-product of another process, direct substitution becomes difficult to implement, and therefore  
219 Stephenson et al. applied allocation based on market prices (scenarios 1a and 1b; Fig. 4).

220 Thamsiriroj and Murphy [48] tested various scenarios, including no-allocation (scenario 0; Fig. 4),  
221 and substitution of various co-products (scenarios 3-6; Fig. 4). Compared to economic allocation (1a  
222 and 1b; Fig. 5), the direct substitution (scenario 2; Fig. 4) resulted in higher GHG savings and lower  
223 total energy consumption. If electricity and heat generated from rapeseed meal in CHP were used to  
224 substitute grid electricity and heat from coal or gas, the GWP decreased by 92% and total energy  
225 requirements by 216%. With the no-allocation method, however, the reduction in GHG emissions was  
226 only 28%. The highest GHG savings (135%) resulted from the use of rape cake for animal feed,  
227 glycerol for heat production, and straw for thermal energy (scenario 4; Fig. 4). Using rape cake as  
228 animal feed saved GHG emissions from the production and transport of soybean meal, usually  
229 imported to Ireland from South America. Stephenson et al. [47] did not consider the burden of fodder  
230 production when rape meal is used to generate energy.

231

### 232 **3.4.3 Assessing corn stover-based ethanol using system expansion, and allocation by mass,** 233 **energy and economic value**

234 In a case study of corn stover ethanol, Luo et al. [30] showed that using an economic allocation gave  
235 much higher results for GWP of bioethanol in comparison to mass/ energy allocation approaches. This  
236 is because the corn/ stover allocation ratio shifted from 1.7 to 7.5 when switching from mass/ energy  
237 allocation to economic allocation.

### 238 **3.4.4 Assessing bioethanol using six allocation methodologies**

239 Kraatz et al. [42] analyzed ethanol production from corn, with dried distillers grains and solubles  
240 (DDGS) as co-product. Using no-allocation approach and system expansion resulted in the highest  
241 energy intensity and highest GWP (see section 3.5). Conversely, mass, energy and economic  
242 allocation gave the lowest values for energy intensity and GWP of bioethanol produced.

### 243 **3.5 Impact categories**

244 Life cycle impact assessment (LCIA) methodologies model the pathway of substances and link them  
245 to effects. There is a large array of LCIA methodologies that propose diverse indicators or/and  
246 calculate the same indicator using different models. The International Reference Life Cycle Data  
247 System (ILCD) handbook reviews a wide range of methods for impact assessment, and provides LCA  
248 practitioners with recommendation on indicators and models used in LCIA [49,50]. The handbook  
249 was developed for LCAs in European context by the European Commission Joint Research Centre.  
250 EDIP 2003, ReCiPe or CML 2001 quoted in this study, are widely used methods [51]. The midpoint  
251 approach translates environmental impacts into mechanisms such as acidification, eutrophication, or  
252 climate change, while endpoint methodologies concentrate on damages and express impacts on the  
253 three following: human health, ecosystems and natural resources [52,53].

#### 255 **3.5.1 LCA based on energy and carbon balances**

256 The majority of biofuel LCAs in the literature look only at GHG emissions or GWP, and/or energy  
257 balance [13]. From 39 papers sampled in Science Direct that used LCA in the title, about half  
258 examined both carbon and energy, while only 26% considered also other environmental impacts (Fig.  
259 5). Stephenson et al. [47] reported on GWP and primary energy requirement (EDIP 2003) [54].  
260 Thamsiriroj and Murphy [34] calculated GHG emissions and reduction according to EU RED  
261 recommendations, as well as the gross and net energy of both rapeseed and palm oil biodiesel. A  
262 similar approach was employed by Smyth et al. [55] and Korres et al. [39] assessing the sustainability  
263 of grass biomethane. In a paper by Kraatz et al., corn grain ethanol was assessed using energy  
264 intensity and GHG (Fig. 6) [42]. When looking at energy intensity, electricity consumption, drying of  
265 DDGS and corn farming show the highest contribution to the overall impact. This changed when  
266 looking at GHG, where 70% of impact comes from electricity alone.

#### 268 **3.5.2 LCA based on impact factors beyond carbon and energy**

269 Measuring of sustainability in terms of impacts beyond carbon and/or energy often gave different  
270 results to when just carbon and/or energy were assessed. Aguirre-Villegas et al. [56] used four  
271 sustainability indicators: GWP; ammonia emissions; depletion of fossil fuel (DFF); and nutrient form  
272 and fate. They found that the anaerobic digestion (AD) pathway in comparison to other manure  
273 utilisations had the lowest values for GWP and DFF, but had the highest NH<sub>3</sub> emissions. Tufvesson et  
274 al. [36] in assessing biogas from industrial residues looked at GHG emissions, eutrophication,  
275 acidification, and energy balance. While GHG emissions were reduced whatever the substrate,  
276 impacts of both eutrophication and acidification were higher for biogas than fossil fuel systems. Only

277 Poeschl et al. [37,38] included land use change indicators (land transformation and occupation) as a  
278 part of the ReCiPe method [51].

279

## 280 **4. Discussion: overcoming the challenges**

### 281 **4.1 Existing LCA frameworks**

282 From the frameworks listed in Table 2, BIOMITRE and RED are the only one specifically developed  
283 for biofuels and biomass. RED also provides the most detailed recommendations on LCA of biofuels.  
284 BioGrace is currently the only integrated tool that complies with RED and can be used by farmers,  
285 policy makers and consultants within Europe [57]. It is an intuitive tool with simple interface that  
286 allows even an unexperienced LCA analyst to get a quick GHG calculation [24]. The tool provides  
287 also a liberty to change parameters and introduce more specific data. However, BioGrace can be  
288 applied only for GHG calculations. Moreover, the RED does not give any recommendation on  
289 extending the impact assessment beyond carbon. Also, despite the harmonisation, the RED still  
290 permits methodological choices that can lead to different results for same biofuel pathway. [58]

291

### 292 **4.2 Functional unit**

293 The choice of FUs should reflect biofuel life cycle stages (Fig. 7). Thus, if feedstock requires  
294 agricultural land, then LCA results should be reported on a per ha basis, and if biofuel is produced for  
295 transportation, then results should be reported on a per km basis. LCA results as per each life cycle  
296 stage should be available using different FU.

297

### 298 **4.3 System boundaries and reference system**

299 The definition of the system's boundaries and choice of the reference system are crucial, as the results  
300 of LCA vary according to the reference system chosen. In order to present a comprehensive  
301 understanding of the system, boundaries should be expanded to include co-products, by-products and  
302 residues (Fig. 7). The boundaries of the reference system should be the same as those of the primary  
303 system under analysis, and the choice of reference system should be informed by the goal of the  
304 analysis. It may be appropriate to define a reference system for each stage of the life cycle process,  
305 such as alternative land use and LCA of products that are being replaced by co- and by-products.  
306 However, the authors recognise that expanding the reference system boundaries and using multiple  
307 reference systems can considerably increase analysis complexity, and therefore recommend that the  
308 primary focus is on defining a reference system according to the analysis goal.

309

### 310 **4.4 Allocation methodology**

311 From the assessed studies, LCA results depend heavily on type of allocation chosen. Van der Voet et  
312 al. stressed that substitution implies higher variability in the results [14]. However, the authors believe



313 that co- and by-products should be included in biofuel LCA. Complex processes should be sub-  
314 divided and data collected for each sub-process by linking of inputs and outputs to products, and co-  
315 and by-products. If this is not possible due to lack of specific data and/or multiplicity of co-/ by-  
316 products, then substitution should be applied (Fig. 7).

317

#### 318 **4.5 Impact categories**

319 Since the majority of LCAs look only at GHG and energy balance, this can lead to the problem of  
320 burden shifting; where a biofuel system might achieve a high level of GHG reduction but could also  
321 impact the environment in other ways, for example through acidification and eutrophication [30]. In  
322 line with previous studies [14,17,36–38,56], the authors recommend broadening LCAs to include  
323 impacts other than just carbon and energy. The ISO does not set a list of recommended impact  
324 categories for life cycle impact assessment, but highlights the importance of choosing these in line  
325 with the goal and scope of the study [15].

326 A comprehensive LCA study would investigate a range of environmental impacts such as climate  
327 change, impact on human health, ecotoxicity, acidification and eutrophication of environment, ozone  
328 layer depletion. While assessment of all these factors would certainly be desirable to gain a full  
329 understanding of the system, it may not be practical due to time and resource constraints. A further  
330 difficulty is the current lack of knowledge of some parameters and/or their poor integration in LCA  
331 studies, particularly indirect effects such as land use change and nitrogen emissions [20,59].

332 The authors limited the number of recommended indicators to the five listed in Table 3. The ILCD  
333 handbook was employed to assess the existing indicators for climate change, acidification,  
334 eutrophication and land use. Only the midpoint categories were considered. Both mid- and endpoint  
335 approaches have their advantages and disadvantages, but midpoint categories are much more accurate  
336 and precise, and bring less uncertainty to the model, unlike the endpoint approach that requires  
337 weighting of the categories [52].

338 To assess climate change, all LCIA methodologies use the GWP midpoint indicator developed by the  
339 Intergovernmental Panel on Climate Change (IPCC) [50]. The GWP should be always based on the  
340 latest IPCC report, in this case the Fifth Assessment Report [60]. GWP can be calculated over a 20, 50  
341 and 100-year timeframe. Well-mixed GHGs, such as CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O (including direct and  
342 indirect emissions from NH<sub>3</sub> and NO) are included. The GWP unit is kg CO<sub>2</sub> eq.

343 Acidification is mainly caused by the airborne acidifying substances, such as ammonia (NH<sub>3</sub>) (after  
344 nitrification in the soil when nitrite is produced), nitrogen oxides (NO<sub>x</sub>) and sulphur dioxide (SO<sub>2</sub>)  
345 (largely from fossil fuels combustion) [61]. The ILCD evaluated the accumulated exceedance (AE)  
346 model as the most suitable [52,53,62]. This method is widely accepted and used by the European  
347 Commission and the United Nation Economic Commission for Europe for policy purposes. It uses  
348 critical load of nutrients to quantify the sensitivity of the ecosystem. It also provides characterization  
349 factors that are country specific. It is expressed in moles of hydrogen ion (H<sup>+</sup>) eq.

350 Eutrophication potential examines the impacts of the surplus of nitrogen and phosphorus on the  
351 terrestrial and aquatic ecosystems (marine and freshwater). Terrestrial eutrophication is caused by  
352 deposition of airborne N emissions, such as  $\text{NO}_x$  (combustion processes), and  $\text{NH}_3$  (agriculture). For  
353 terrestrial eutrophication, ILCD recommends the AE model [52,53,62]. The indicator is expressed in  
354  $\text{kg N eq}$  [52]. Freshwater and marine eutrophication is induced by waterborne emissions, such as  
355 nitrate, phosphate and other N and P compounds [52]. It is recommended to use the ReCiPe method,  
356 as it models best the aquatic fate of emissions; however it is restricted only to European countries.  
357 The indicator is expressed in  $\text{kg P eq}$ .

358 Land use indicators reflect the changes to ecosystems due to the effects of land occupation and  
359 transformation. To assess the impact of land use the ILCD handbook recommended the method by  
360 Milà i Canals et al., based on soil organic matter (SOM) [63,64]; however the level of  
361 recommendation is a grade lower than for the other impact categories discussed above [50]. The  
362 drawback of this method is that the LCA practitioner must calculate the case-specific characterisation  
363 factors based on collected data, such as SOM value before and after the land occupation and SOM  
364 value of the reference system. Moreover, the method presents a limited impact indicator based on  
365 SOM that considers soil quality but does not include the impact on soil biodiversity [52].  
366 Alternatively, the midpoint ReCiPe method can be applied, but it accounts only for surface area of  
367 transformed and occupied land (expressed in  $\text{m}^2$ ).

368 An energy indicator is typically associated with the assessment of energy vectors such as biofuels. It  
369 requires thorough data collection on energy inputs and outputs. It can take various forms, such as: 1)  
370 energy balance (energy output to input ratio), 2) net energy (gross energy of the product minus  
371 parasitic energy demand of the processes), 3) land use energy efficiency (for land-based biofuels;  
372 energy produced per unit area).

373 Traditionally, LCA is a tool for assessment of global impacts [65]. This is still valid for climate  
374 change, but for impacts such as acidification and eutrophication that occur locally, there is a need for  
375 country- or site-specific characterisation factors. ReCiPe and CML methods include the European  
376 average factor [17,50], while Seppala et al [53] and Posch et al [62] went even further to include EU  
377 country-specific factors for acidification and terrestrial eutrophication. Finnveden and Nilsson argue  
378 that the site-specific factors are needed to allow including the local conditions into the model [17].

379 To conclude, there is a need for a framework on impact categories for biofuels assessment. These  
380 should include at least the categories described above: climate change, acidification, eutrophication,  
381 land use, and energy. Further research should provide LCA practitioners with site-specific factors for  
382 acidification, eutrophication and land use.

383

## 384 **5. Concluding remarks**

385 A sound evaluation of biofuel systems requires conducting a full cradle to grave LCA. Valid LCA  
386 studies should consider other sustainability indicators in addition to GHG emissions and energy

387 balances. Standard requirements should include functional unit, system boundaries, allocation  
388 methodology, and environmental indicators. This unified methodology will allow comparing biofuel  
389 LCA studies, which currently is not possible as different studies follow different rules. Whereas it is  
390 preferable that full LCA is carried out in academia, industry should have access to a simplified and  
391 cost-effective version of LCA. LCA is a powerful tool but needs to continue to be refined as  
392 knowledge of the science grows. The recommendations outlined in this paper should go some way  
393 towards achieving this.

394

395

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- 590
- 591



592 **Figures**

593 Fig. 1. Methodology for selecting reviewed papers.

594

595 Fig. 2. Cradle to gate energy balance (GJ) and GHG balance (kg CO<sub>2</sub>) of three biofuels. Data were  
596 expressed using in each case two FU: ha of land and GJ of biofuel. In [33] grass biomethane  
597 performed better regardless the FU, but there were bigger discrepancies between rapeseed biodiesel  
598 and grass biomethane results if FU was set as GJ of fuel (a.). In [34], if results were expressed using  
599 ha of land, rapeseed biodiesel performed better (less kg CO<sub>2</sub> ha<sup>-1</sup>), while if FU was switched to GJ of  
600 fuel, palm oil biodiesel was better (less kg CO<sub>2</sub> GJ<sup>-1</sup>) (b.).

601

602 Fig. 3. Allocation approaches.

603

604 Fig. 4. Scenarios for LCA comparison of rapeseed biodiesel based on Stephenson et al. [47] and  
605 Thamsiriroj and Murphy [48].

606

607 Fig. 5. Percentage of peer-reviewed LCA studies of biofuels by impact categories assessed. 39 peer-  
608 reviewed papers were sampled using Science Direct. All papers were published between 2008 and  
609 2015 using “LCA” and “biofuel” in title.

610

611 Fig. 6. Contribution of the three most impacting life cycle stages of corn grain ethanol to the total  
612 environmental impact measured in energy intensity (blue) and GWP (red) [42]. Electricity was always  
613 the highest contributor to both GWP and energy intensity, but the discrepancies between various life  
614 cycle stages were much lower if energy intensity is used as measure. Drying of DDGS has a much  
615 higher impact on energy intensity than on GWP.

616

617 Fig. 7. Flowchart with recommendations for biofuels LCA.

618

619 **Tables**

620 Table 1. Overview of sixteen selected papers on LCA.

| Category            | References                  | Product studied   | Functional unit   | System boundaries   | Reference system  | Allocation methodology  | Impact categories   |
|---------------------|-----------------------------|---|---|---|---|---|---|
| <b>• Biodiesel</b>  |                             |   |   |   |   |   |   |
|                     | Stephenson et al. [47]      | Rapeseed biodiesel  | 1 tonne of biodiesel  | 1. Crop production<br>2. Transport<br>3. Oil extraction<br>4. Biodiesel production<br>5. Distribution   | Fallow set-aside (alternative land use)                             | 1. Economic<br>2. Substitution  | GWP and primary energy requirement based on EDIP 2003*  |
|                     | Thamsiriroj and Murphy [34] | Biodiesel from 1. palm oil and 2. rapeseed  | 1 GJ biodiesel or 1 ha per year                                   | 1. Crop production<br>2. Transport<br>3. Oil extraction<br>4. Biodiesel production<br>5. Distribution   | Diesel reference system   | No allocation (all burden to biodiesel)   | GHG and energy balance  |
|                     | Thamsiriroj and Murphy [33] | Biodiesel from 1. rape seed, 2. tallow and 3. Used cooking oil, and 4. grass biomethane | 1 GJ biodiesel or 1 ha per year                                   | Variable  | Fossil fuel reference system  | 1. No allocation<br>2. Energy<br>3. Substitution  | GHG and energy balance  |
|                     | Yang et al. [29]            | Microalgae biodiesel  | 1 kg of biodiesel   | 1. Growing and harvesting<br>2. Drying<br>3. Oil extraction<br>4. Esterification  | Alternative feedstocks  | No allocation   | Water footprint and nutrients balance   |
| <b>• Bioethanol</b> |                             |   |   |   |   |   |   |
|                     | Kaufman et al. [40]         | Ethanol from 1. corn-grain and 2. from corn stover                                      | 1 MJ of ethanol   | 1. Corn production<br>2. Transport<br>3. Ethanol refining (co-product of corn-grain ethanol can be fed to cattle; co-product of corn-stover ethanol, electricity is injected into the grid) | Gasoline reference system   | 1. Mass<br>2. Economic<br>3. Energy<br>4. System expansion<br>5. Subdivision                      | GHG   |
|                     | Kraatz et al. [42]          | Corn grain ethanol  | 1 kg of ethanol or 1 MJ of ethanol                                | 1. Corn grain production<br>2. Transport<br>3. Ethanol refining<br>4. Use of stillage for biogas production or for animal feed production   | Gasoline reference system   | 1. No allocation,<br>2. Mass<br>3. Economic<br>4. Energy<br>5. Subdivision<br>6. System expansion | GWP, energy intensity, and net energy value   |
|                     | Luo et al. [30]             | Corn stover ethanol   | 1 km driven in a midsize car                                      | 1. Agricultural production<br>2. Transport<br>3. Pretreatment<br>4. Fermentation<br>5. Distillation<br>6. Refining<br>7. Blending<br>8. Car driving   | Gasoline reference system   | 1. Mass<br>2. Economic<br>3. Energy<br>4. System expansion  | GWP, abiotic depletion, ozone layer depletion, photochemical oxidation, human and ecotoxicity, acidification and eutrophication based on CML 2001** |
| <b>• Biomethane</b> |                             |   |   |   |   |   |   |
|                     | Korres et al. [39]          | Grass biomethane  | 1 m <sup>3</sup> of biomethane per year, and 1 MJ energy replaced | 1. Crop production<br>2. Biogas production<br>3. Upgrading and compressing of biomethane<br>4. Use in a bi-fuel car<br>5. Digestate use   | Diesel reference system   | No allocation   | GHG and energy balance  |
|                     | Smyth et al. [55]           | Grass biomethane  | 1 ha per year   | 1. Crop production<br>2. Biogas production<br>3. Upgrading and compressing of biomethane<br>4. Digestate use  | 1 <sup>st</sup> generation biofuels (palm oil biodiesel and others) | No allocation   | Energy balance  |
|                     | Wang et al. [28]            | Microalgal biomethane   | 1 GJ of biomethane  | 1. Algae cultivation (incl. production of photobioreactor)<br>2. Biogas production<br>3. Upgrading of biogas  | Alternative biomethane production from ley crop                     | Substitution (avoided fertilizer)   | GHG and energy balance  |
| <b>• Biogas</b>     |                             |   |   |   |   |   |   |

|  |  |   |   |   |  |  |
|--|--|---|---|---|--|--|
| Alvarado-Morales et al. [26]                                 | Biogas from brown seaweed (and bioethanol + biogas)  | 1 tonne of dry seaweed                  | 1 <sup>st</sup> scenario:<br>1. Seaweed production<br>2. Mechanical pretreatment<br>3. Biogas production<br>4. Energy production<br>2nd scenario:<br>3. Bioethanol production<br>4. Blending<br>5. Car driving<br>6. Use of stillage for biogas production  | Coal-based electricity for biogas, and fossil gasoline for bioethanol | System expansion (avoided fertilizer and energy production, and fuel production)   | GWP, acidification and terrestrial eutrophication based on EDIP 2003* method, and energy consumption |
| Börjesson and Berglund [43], and Börjesson and Berglund [44] | Biogas systems (ley crops, straw, sugar beet, manure, food waste, municipal organic waste)   | 1 MJ of biogas                          | 1. Handling of raw materials (incl. crop cultivation)<br>2. Biogas production (farm-scale and large-scale biogas plants)<br>3. Biogas and digestate use   | Various (fossil fuels and other bioenergy systems)                    | Based on dry matter content of the substrates  | GWP, acidification, eutrophication, and photochemical oxidant creation potential                     |
| Poeschl et. al. [37] and Poeschl et. al. [38]                | Biogas system (various scenarios incl. single feedstock and co-digestion of manure, straw, corn, grass, whole wheat plant, MSW, food waste, pomace, slaughterhouse waste, grease sludge) | 1 tonne of organic material (feedstock) | 1. Feedstock supply<br>2. Biogas production<br>3. Biogas utilization<br>4. Digestate processing and handling; (system expansion: production of chemical fertilizers, electricity generation with Germany's fuel mix, heat generation using natural gas, and imports of natural gas and transportation fuel) | Typical biogas production and utilization pathways for Germany        | System expansion   | 18 midpoint and 3 endpoint indicators based on ReCiPe*** method                                      |
| Tufvesson et al. [36]  | Biogas from industrial residues (distiller's waste, rapeseed cake, whey permeate, fodder milk, and bakery residues)  | 1 MJ upgraded and compressed biogas     | 1. Transport of feedstocks<br>2. Biogas production including upgrading and compression<br>3. Digestate use (no allocation and system expansion)<br>4. Replacement of mineral fertilizer<br>5. Animal feed production (system expansion)   | Petrol and diesel reference system                                    | 1. No allocation, 2. System expansion and 3. allocation rules based on the sustainability criteria defined by the EU RED | GHG, energy balance, eutrophication and acidification potential                                      |

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\* EDIP 2003: Impact assessment method developed by the Institute for Product Development at the Technical University of Denmark [54].

\*\* CML 2001: Impact assessment method developed by the Institute of Environmental Sciences at Leiden University in Netherlands [54].

\*\*\* ReCiPe : Impact assessment method developed by various actors: PRé Consultants, CML Leiden University, Radboud University Nijmegen and RIVM Bilthoven, [51].

627 Table 2. Allocation methodologies according to different sources.

| Institution  | Recommended Approach  |
|--|---|
| <ul style="list-style-type: none"> <li>• <b>Norm ISO 14044</b> (International Standard Organization)</li> </ul>  | whenever possible avoid allocation and instead use subdivision or system expansion [15]   |
| <ul style="list-style-type: none"> <li>• <b>RED and BioGrace tool</b></li> </ul>   | allocation based on energy content (lower heating value) [2]  |
| <ul style="list-style-type: none"> <li>• <b>BIOMITRE</b> (BIOmass-based Climate Change MITigation through Renewable Energy)</li> </ul>                   | allocation by economic value (although not ideal since market prices often fluctuate) [23]  |
| <ul style="list-style-type: none"> <li>• <b>PAS</b> (Publicly Available Specification for assessment of life cycle GHG of goods and services)</li> </ul> | dividing processes into sub-processes or system expansion to include co-products, by-products, and waste; when neither of these is feasible, then allocation based on economic value should be applied [22] |
| <ul style="list-style-type: none"> <li>• <b>UNFCCC through Meth Panel</b> (Methodologies Panel)</li> </ul>   | all available approaches [66]   |

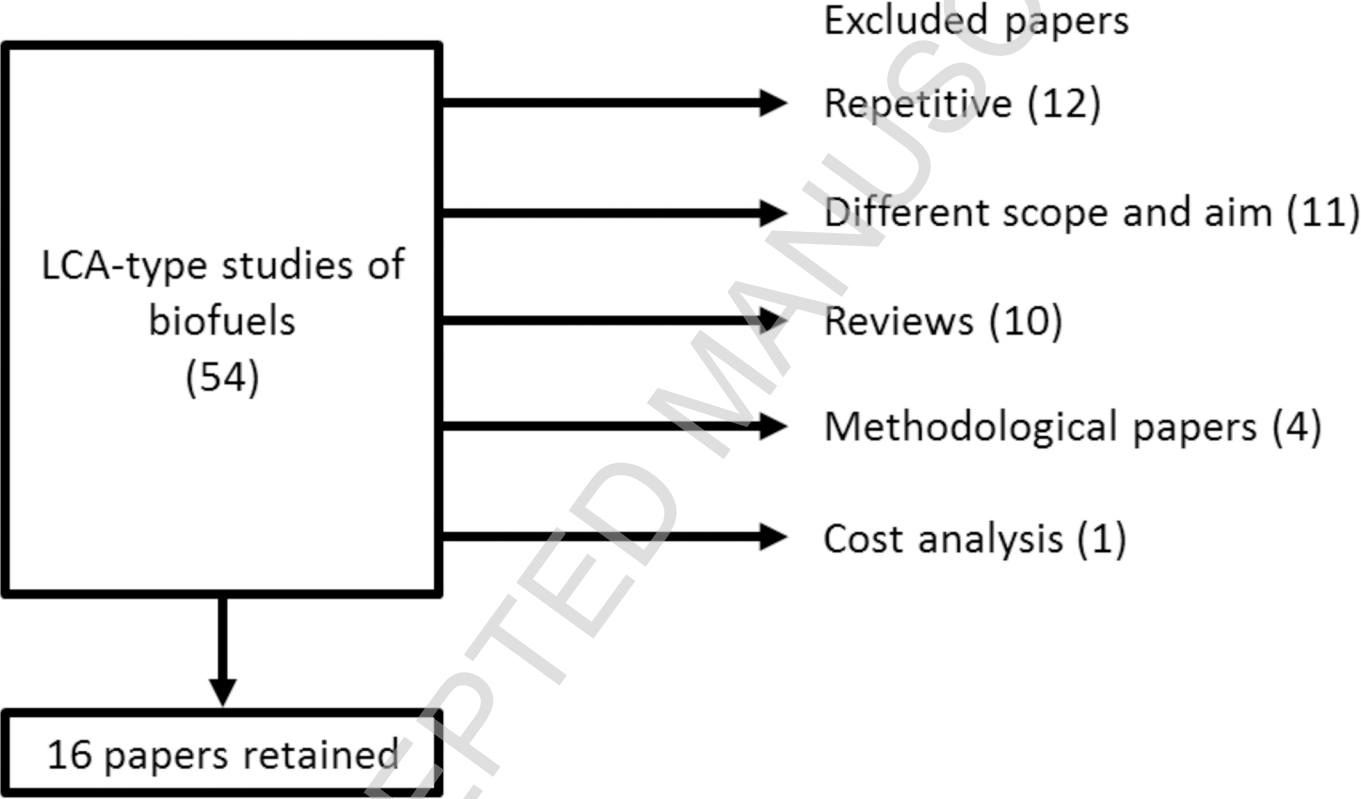
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629 Table 3. Recommendation for LCIA.

| Impact categories  | Indicators   | Calculation and unit  | Reasons  |
|--|--|---|--|
| <ul style="list-style-type: none"> <li>• Climate change</li> </ul>                           | GWP or GHG   | g or kg CO <sub>2</sub> -equivalent   | Following legislative requirements to calculate impact of biofuels on global warming and potential savings                           |
| <ul style="list-style-type: none"> <li>• Energy</li> </ul>                                   | Energy balance, net energy, land use energy efficiency | Energy balance (output/input ratio), net energy (gross energy minus parasitic energy demand) land use efficiency (energy production per unit of land); kwh or MJ (per ha) | Traditional indicator related to biofuel energy efficiency   |
| <ul style="list-style-type: none"> <li>• Eutrophication (terrestrial and aquatic)</li> </ul> | Accumulated exceedance (terrestrial), ReCiPe (aquatic) | Terrestrial: modelling following Seppala et al. [53,62]; kg N eq. Aquatic: ReCiPe; kg P eq. [51]  | Terrestrial strongly correlated with agriculture and combustion (N compounds); aquatic with waterborne emissions (N and P compounds) |
| <ul style="list-style-type: none"> <li>• Acidification</li> </ul>                            | Accumulated exceedance                                 | Modelling following Seppala et al. [53,62]; moles of hydrogen ion (H <sup>+</sup> ) eq.   | Strongly correlated with transport and agriculture (N and S compounds)   |
| <ul style="list-style-type: none"> <li>• Land use</li> </ul>                                 | SOM or surface area of transformed and occupied land   | mg SOM per year (deficit of SOM) [63,64] and m <sup>2</sup> of transformed and occupied land [51]   | Especially relevant for land-based biofuels  |

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LCA-type studies of  
biofuels  
(54)

Excluded papers

Repetitive (12)

Different scope and aim (11)

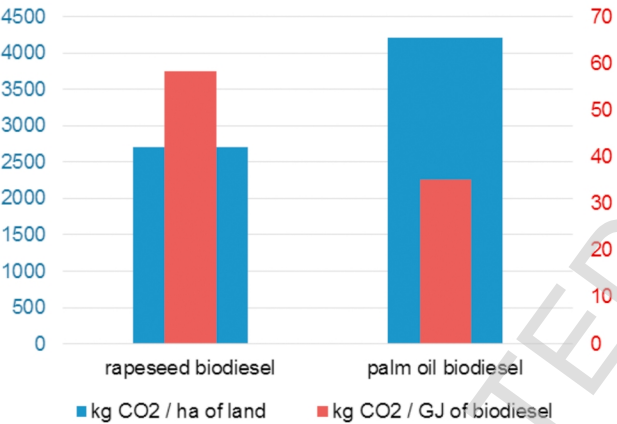
Reviews (10)

Methodological papers (4)

Cost analysis (1)

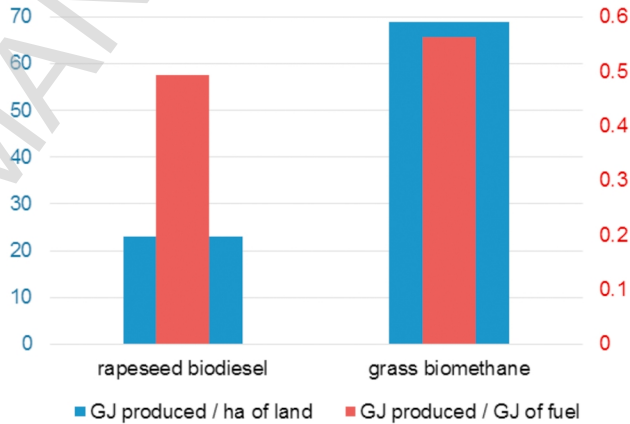
16 papers retained

a.

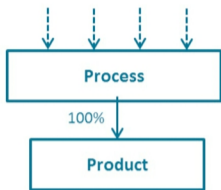
GHG [kg CO<sub>2</sub>]

b.

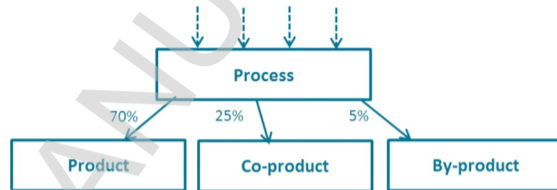
Energy balance [GJ]



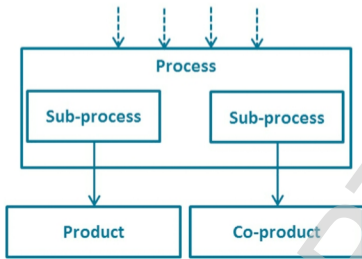
No allocation



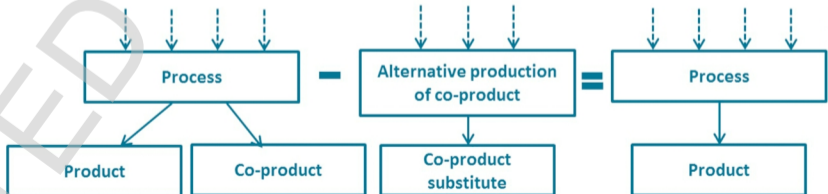
Mass/ energy/ economic allocation

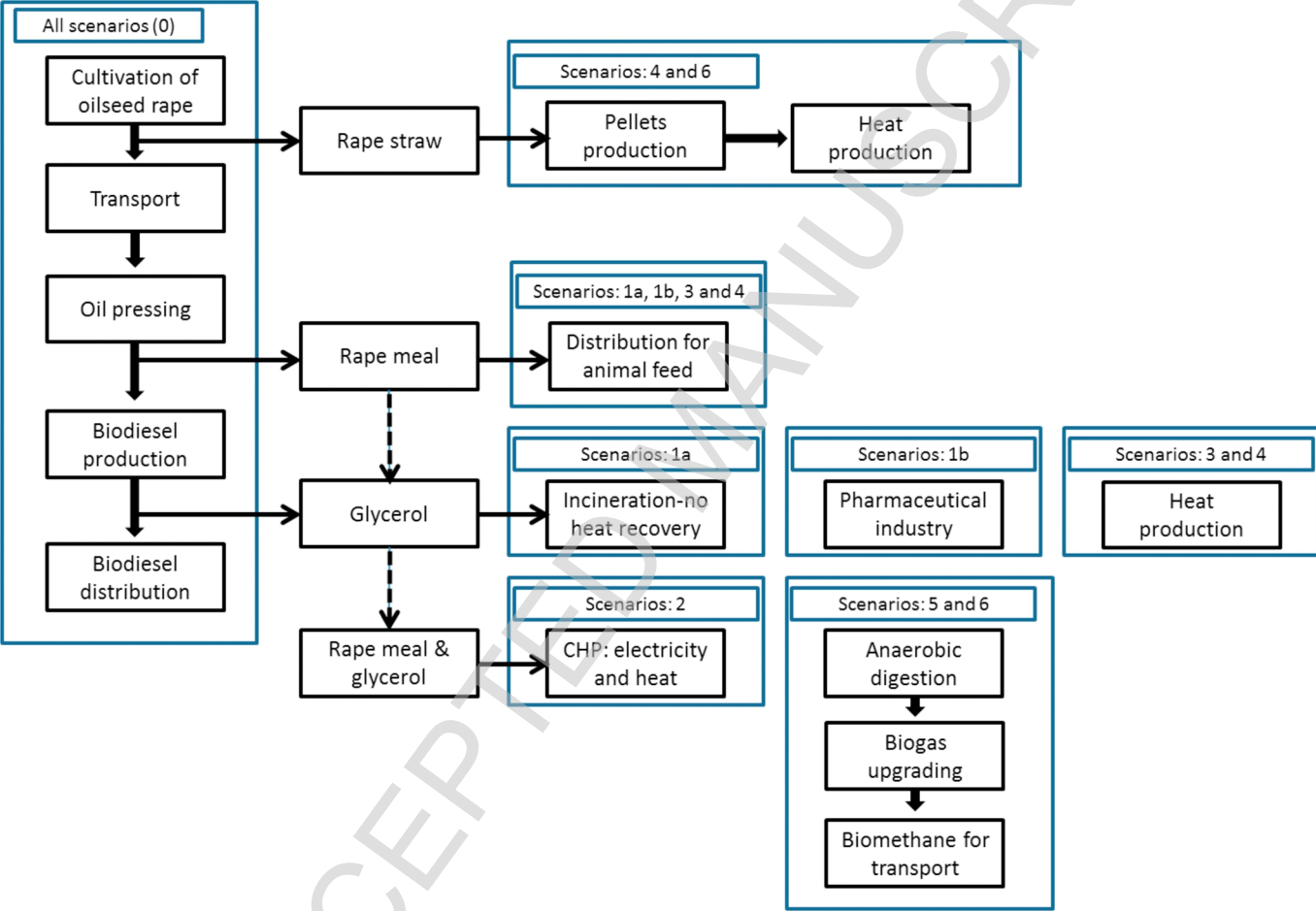


Subdivision



System expansion/ substitution







# Impact categories

