The International Journal of Life Cycle Assessment (2018) 23:2055–2070
https://doi.org/10.1007/s11367-017-1432-6

UNCERTAINTIES IN LCA

Uncertainty in LCA case study due to allocation approaches and life cycle impact assessment methods

Edivan Cherubini[1,2] · Davide Franco[3] · Guilherme Marcelo Zanghelini[1,2] · Sebastião Roberto Soares[1]

Received: 28 March 2016 / Accepted: 20 December 2017 / Published online: 8 January 2018

Abstract

Purpose Uncertainty is present in many forms in life cycle assessment (LCA). However, little attention has been paid to analyze the variability that methodological choices have on LCA outcomes. To address this variability, common practice is to conduct a sensitivity analysis, which is sometimes treated only at a qualitative level. Hence, the purpose of this paper was to evaluate the uncertainty and the sensitivity in the LCA of swine production due to two methodological choices: the allocation approach and the life cycle impact assessment (LCIA) method.

Methods We used a comparative case study of swine production to address uncertainty due to methodological choices. First, scenario variation through a sensitivity analysis of the approaches used to address the multi-functionality problem was conducted for the main processes of the system product, followed by an impact assessment using five LCIA methods at the midpoint level. The results from the sensitivity analysis were used to generate 10,000 independent simulations using the Monte Carlo method and then compared using comparison indicators in histogram graphics.

Results and discussion Regardless of the differences between the absolute values of the LCA obtained due to the allocation approach and LCIA methods used, the overall ranking of scenarios did not change. The use of the substitution method to address the multi-functional processes in swine production showed the highest values for almost all of the impact categories, except for freshwater ecotoxicity; therefore, this method introduced the greater variations into our analysis. Regarding the variation of the LCIA method, for acidification, eutrophication, and freshwater ecotoxicity, the results were very sensitive. The uncertainty analysis with the Monte Carlo simulations showed a wide range of results and an almost equal probability of all the scenarios be the preferable option to decrease the impacts on acidification, eutrophication, and freshwater ecotoxicity. Considering the aggregate result variation across allocation approaches and LCIA methods, the uncertainty is too high to identify a statistically significant alternative.

Conclusions The uncertainty analysis showed that performing only a sensitivity analysis could mislead the decision-maker with respect to LCA results; our analysis with the Monte Carlo simulation indicates no significant difference between the alternatives compared. Although the uncertainty in the LCA outcomes could not be decreased due to the wide range of possible results, to some extent, the uncertainty analysis can lead to a less uncertain decision-making by demonstrating the uncertainties between the compared alternatives.

Keywords Life cycle assessment · Methodological choices · Sensitivity analysis · Uncertainty analysis · Allocation approach · LCIA method

Responsible editor: Nydia Suppen-Reynaga

Electronic supplementary material The online version of this article (https://doi.org/10.1007/s11367-017-1432-6) contains supplementary material, which is available to authorized users.

✉ Edivan Cherubini
edivan.cherubini@posgrad.ufsc.br; edivan@enciclo.com.br

[1] Life Cycle Assessment Research Group (CICLOG—www.ciclodevida.ufsc.br), Department of Environmental and Sanitary Engineering, Federal University of Santa Catarina, Campus Universitário, Mail Box 476, Florianópolis 88040-970, Brazil

[2] EnCiclo Soluções Sustentáveis Ltda., Florianópolis, Brazil

[3] Maritime Hydraulics Laboratory (LAHIMAR), Department of Environmental and Sanitary Engineering, Federal University of Santa Catarina, Campus Universitário, Mail Box 476, Florianópolis 88040-970, Brazil

1 Introduction

Life cycle assessment (LCA) methodology (ISO 2006a; ISO 2006b) has been widely used to evaluate and compare environmental profiles of products and services. LCA is a model that is mainly used to support decision making (Huijbregts 1998; Soares et al. 2013). However, certain issues in the methodology require the LCA practitioner to decide between different possibilities to conduct their study. This freedom of choice can sometimes lead to widely varying results, thus creating uncertainty. Uncertainty is present in many forms in all stages of an LCA and is generated from the sparse and imprecise nature of the available information and simplified model assumptions (Soares et al. 2013; Heijungs and Lenzen 2014). Therefore, the uncertainty in LCA outcomes can mislead decision makers in a scenario comparison (Huijbregts 1998; Geisler et al. 2005).

Finnveden et al. (2009) define uncertainty as "*the discrepancy between a measured or calculate quantity and the true value of that quantity.*" The authors distinguish between the sources of uncertainty in LCA (e.g., data, choices, and relations) and types of uncertainty, such as data variability, inconsistent choices across alternatives, and incorrect relations to reflect the relationship between a pollutant emission and its environmental impact. Several of these uncertainties in the four LCA phases are related to the definitions of system boundaries, cutoff rules, functional unit (FU), data quality, allocation approach, identification of representative impact categories, characterization models, normalization, weighting, and interpretation of trade-offs for decision making.

Perhaps two of the most remarkable causes of uncertainty are the approach used to address multi-functional problems and the LCIA method selection. The allocation approach (hereafter, "allocation approach" means both allocation and system expansion/substitution method) selection is one of the most discussed and controversial methodological issues because of the profound effect on the results in LCA studies (Curran 2007; Finnveden et al. 2009; Weidema and Schmidt 2010; Cherubini et al. 2011). This problem is very common in LCA because almost all product systems will present at least one multi-functional process, i.e. how to fairly allocate the environmental impacts between the different coproducts generated by the same process?

Galindro (2012), in a LCA of biodiesel production, performed a sensitivity analysis in the allocation approach and concluded that depending on the approach used, the results for eutrophication potential in a scenario comparison can change the ranking of the most favorable system (scenario with chemical fertilizer vs. Bioflocs technology). Schmidt and Dalgaard (2012) reached the same conclusions using different allocation approaches in a comparison of two different systems of milk production.

Conversely, in a case study conducted by Curran (2007), the allocation procedure selection had no impact on the relative ranking in a comparative LCA. Similarly, Huijbregts (1998), in a comparative LCA of two roof gutters, evaluated the uncertainty due to choices on the LCA results through a scenario analysis and a Latin hypercube simulation. This study found that the sensitivity analysis did not change the relative ranking of the scenarios. However, when a statistical analysis was applied, the results showed no significant differences in acidification between the two production systems considering different allocation procedures, i.e., statistically, it was not possible to state which scenario further decreased the acidification impacts. In this sense, an interpretation of the LCA results without an uncertainty analysis could indicate that one of the scenarios is the best environmental choice to decrease acidification potential. Therefore, it is not possible to state an uncertainty factor of all LCA studies based on previous sensitivity analyses.

In the life cycle impact assessment (LCIA) phase, uncertainties are due to the different pollutant substances and characterization factors adopted by each method (Cellura et al. 2011). Several studies (Dreyer et al. 2003; Bovea and Gallardo 2006; Renou et al. 2008; Hung and Ma 2009; Pizzol et al. 2011; Alvarenga et al. 2012; Cavalett et al. 2013; Owsianiak et al. 2014) have evaluated the sensitivity in LCA results due to different LCIA methods. Dreyer et al. (2003), comparing three different LCIA methods, concluded that in some cases, the selected impact assessment method does matter, especially when toxicity-related impacts are evaluated. Owsianiak et al. (2014) also evaluated the consequences of selecting different LCIA methods (ILCD 2009, IMPACT2002+ and ReCiPe 2008) on the impacts of four window designs. The results did not change the ranking of the environmental profiles between the best and worst alternative, although some differences in the absolute values were observed. However, for the toxicity impacts, the authors observed differences in scenario ranking. In addition, the authors only focused on sensitivity analysis without an uncertainty evaluation of the variations caused by the sensitivity; therefore, even though the scenario ranking did not change for some impact categories, the compared alternatives may not be statistically different. The results from the toxicity impacts are in line with the statement from Hung and Ma (2009) that differences across the LCIA methods can introduce a large degree of uncertainty into LCA outcomes.

Despite this recognized problem, no correct procedure currently exists to address the choice between the existing methods. However, the number of characterization models continues to increase. Thus, the selection of the LCIA method depends on the LCA practitioner's experience and interpretation of the evaluated product system and the available methods, which is a subjective choice (Weidema and Wesnæs 1996; Cellura et al. 2011; Hauschild et al. 2013). One of the recently published International Reference Life

Cycle Data System (ILCD) handbook guides (EC-JRC 2011) attempted to standardize the characterization model selection. The ILCD handbook provides the basis for greater consistency and quality in life cycle data, methods and LCA studies (Hauschild et al. 2013). Specifically, for the LCIA phase, a group of experts and stakeholders reviewed the existing models and provided individual recommendations per impact category (Hauschild et al. 2013). However, even though ILCD recommends some of the existing models due to its satisfactory level of scientific quality, the choice of the LCIA method remains a value judgment, and therefore there is no correct procedure for choosing between the existing methods.

To address the aforementioned sources of uncertainty, several statistical methods have been used in an attempt to increase the reliability of LCA results. Some statistical theories used in LCA are parameter variation and scenario analysis, classical statistical theory (e.g., probability distributions and tests of hypothesis); Monte Carlo simulations, bootstrapping and other sampling approaches; analytical methods based on first-order error propagation; non-parametric statistics, Bayesian analysis, fuzzy theory; and the use of qualitative uncertainty methods (e.g., based on data quality indicators) (Finnveden et al. 2009).

Although uncertainty is often not considered in LCA studies, it can be very high; thus, quantifying the uncertainty is an important step to provide support for the interpretation of LCA results to reach trustworthy and transparent decisions (Geisler et al. 2005; Finnveden et al. 2009). For a reliable judgment of the environmental gains in a comparison between products or options for improvement, the uncertainty analysis is very helpful in understanding to what extent the LCA results are in fact different between different scenarios (Huijbregts et al. 2001).

Uncertainty due to methodological choices has already been studied (Huijbregts 1998; Hung and Ma 2009), but only in the past few years, more attention has been paid to this topic. According to Zamagni et al. (2009), scenario uncertainty is the least addressed in studies and is generally treated at a qualitative level. As noted by the authors, major efforts should be made to address this source of uncertainty because of the significant consequences on the LCA results.

Although uncertainty due to methodological choices have already been studied for allocation approach (Mendoza Beltran et al. 2016; AzariJafari et al. 2017; Mendoza Beltran et al. 2017) and allocation along with LCIA methods (Gregory et al. 2016), treating uncertainty due to choices with a stochastic modeling framework is still poorly applied (Baustert and Benetto 2017) and the topic remains as an important issue for research. In addition, the uncertainty analyses due to allocation were only applied to partitioning methods, which can have small differences in the allocation factors. The inclusion of the substitution method can introduce a broad range of scenario space since the avoided product can bring a positive impact to the product system under study (e.g. climate change impacts in Luo et al. 2009). Furthermore, the uncertainty of different methodological choices in the swine production life cycle and its byproducts has not been evaluated by a probabilistic analysis. Therefore, two main questions can be raised: Considering the uncertainties due to methodological choices is it possible to differentiate four alternatives for manure management in the swine supply chain? Moreover, which methodological choice leads to greater uncertainty in the LCA results? To answer these questions, we evaluated the LCA uncertainty and sensitivity due to two methodological choices: the allocation approach and the LCIA method.

2 Material and methods

To address the uncertainty in LCA, we used a scenario variation with sensitivity analysis followed by uncertainty analysis. For the scenario variation, a comparative case study of four manure management systems (MMS) in swine production was used to evaluate whether the uncertainty due to methodological choices could change the scenario ranking. The functional unit of the case study was 1000 kg of swine carcasses (deadweight) in the equalization chamber for cutting or further distribution. The scenarios for the MMS are liquid manure storage in slurry tanks (Sce.Ref), the biodigestor by flare (Sce.Flare), the biodigestor for energy purposes (Sce.CHP), and composting (Sce.Comp). A detailed description of the scenarios is given in Cherubini et al. (2015a). The system boundaries with the main unit processes of swine production are displayed in Fig. 1.

Figure 2 shows an overview of the schematic framework of the statistical method used to evaluate the uncertainty. The input data are based on the life cycle inventory (LCI) of the case study, through a sensitivity analysis based on a deterministic approach, that is a function of the allocation procedure and the LCIA method, and then the outcomes of this analysis were used as inputs in the probabilistic approach for the uncertainty analysis per impact category. Table S2 in the Electronic Supplementary Material shows the input data used in the Monte Carlo simulation. To compare the probability that one alternative has lower impact than another through the results of the Monte Carlo simulation we calculated a comparison indicator based on Huijbregts et al. (2003). The results of the comparison indicators were plotted in histograms.

2.1 Sensitivity analysis of the methodological choices in LCA

The sensitivity analysis was conducted and interpreted in two steps: (1) considering different approaches to address the multi-functionality problem in the main processes of the case

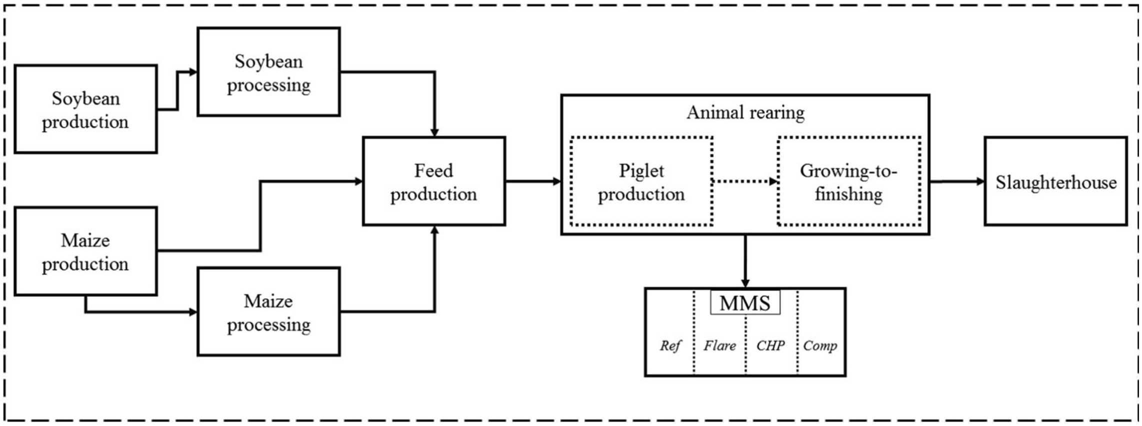


Fig. 1 Simplified system boundaries of swine production

study; and (2) the use of several life cycle impact assessment (LCIA) methods. The results from this analysis identify which choice (allocation or LCIA) more significantly influences the LCA outcomes and also generates the data used in the uncertainty analysis.

2.2 Multi-functionality problem in the case study

In swine production, the main multi-functional processes are grain processing (i.e., soybean and maize), animal rearing, the manure management system (MMS) and the slaughtering stage.

There are two main approaches to address the multi-functionality problem: the partitioning methods and the avoided burdens or substitution method, also called system expansion[1] (Heijungs and Suh 2002; Heijungs and Guinée 2007; Finnveden et al. 2009; Cherubini et al. 2011). The partitioning methods are commonly based on weight (mass), volume, market value, energy, exergy, and demand (Curran 2007; Cherubini et al. 2011). A detailed background of the mathematical procedures on the partitioning methodologies can be found in Curran (2007).

According to Schmidt and Dalgaard (2012), for recycling processes such as the MMS stage, allocation can also be divided into two types: type I, where the allocation occurs after the MMS, i.e., before manure application in soil (point of substitution), and type II, which occurs before sending the manure to the MMS, i.e., not at the point of displacement. In the second type, the impacts of treatment and/or recycling are attributed to the product system that will receive the byproducts. Following the authors' definition, we considered type I allocation.

[1] Some authors argue that system expansion and the substitution method are equivalent concepts (Tillman et al. 1994; Ekvall and Tillman 1997; Ekvall and Finnveden 2001). However, equivalent does not mean equal, and the two concepts will not generate the same results, so the concepts can be compatible (Wardenaar et al. 2012; Heijungs 2014).

Table 1 shows the scenario variation and the approaches used to address the multi-functionality problems in the main processes of swine production. The reference scenario considers the allocation of the environmental burdens on a case-by-case basis and was described in Cherubini et al. (2015a). Regarding scenario variation for economy allocation, all of the multi-functional processes in the case study were handled considering the market values. Similarly, in the remaining scenario variation, we applied the same procedure considering only the mass allocation and system expansion through the substitution method for all of the main processes with coproducts. Table 1also describes the allocation factors attributed to each coproduct using the partitioning methods and the avoided products for the substitution method. Further information about scenario assumptions and definitions are presented in the following sections.

2.2.1 Economic allocation assumptions

The market values for piglets, sows, boars, and swine in animal rearing and for swine carcasses and their coproducts in the slaughtering house stage were taken from the Brazilian agroindustry and are representative for the period of 2014–2015. To define a market value for the manure used as organic fertilizer in the economic allocation variation, we assume the same commercial price of urea, triple superphosphate, and potassium chloride, which are equivalent to the fertilizing potential of manure. One disadvantage of economic allocation is the simplified assumption that market value remains constant which in practice cannot be realistic. To address possible effects of price in our results, we conducted a sensitivity analysis assuming an arbitrarily price variation of ±50%. The outcomes of this sensitivity analysis showed little variations on the means and standard deviations of our results; therefore, even considering such a high variation, our results were not affected. A detailed description of estimating the fertilizing potential of manure is given in Cherubini et al. (2015b), and the allocation factors are displayed in Table 1.

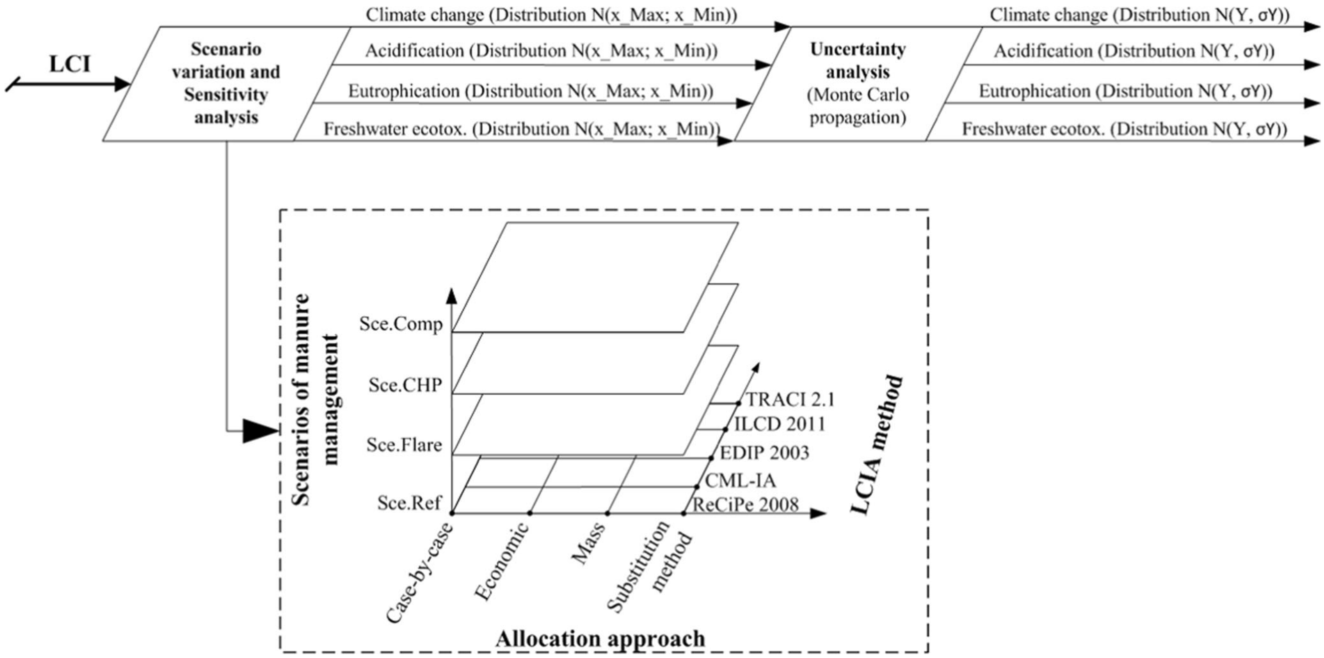


Fig. 2 Schematic framework of the method used to estimate the uncertainty in LCA

2.2.2 Mass allocation assumptions

The mass allocation in the MMS stage considered the mass of the fertilizer content in manure (e.g., 80.2 kg of chemical fertilizer in Sce.Ref) due to the greater amount of manure generated per ton of swine carcasses (i.e., 6.33 m^3). For Sce.CHP, we decided to not consider any burden attributed to electricity or heat production in the MMS because it is not possible to set a weight on energy (Table 1). To overcome this situation, we expanded the system boundaries of Sce.Ref, Sce.Flare, and Sce.Comp to also consider the impacts of electricity and heat production. Therefore, all the system products provide the same functions (i.e. meat, electricity, and heat).

2.2.3 Substitution method assumptions

The system expansion through the substitution method implies that the coproducts of swine production avoid other products in the market (sometimes also called replaced products). In this case, the most price competitive product of each coproduct needs to be identified along with in which proportion that product would be avoided or replaced by the coproduct of swine production.

However, as stated by Gac et al. (2014), finding an equivalent product is a weak point of this approach because this definition is often difficult and quite subjective and it is sometimes not possible to identify an avoided product. Heijungs and Guinée (2007) argue that the substitution method or system expansion introduces many "what-if" assumptions into LCA modeling. For these reasons, this type of question should preferably be left out of a primarily scientific tool. Therefore, it should further be considered that our results could be quite different depending on the assumptions made to identify an avoided product. Nevertheless, we believe that the assumptions made in this case study are adequate to illustrate the uncertainty in a comparative LCA.

For soybean processing, based on Weidema (1999) and Dalgaard et al. (2008), it was assumed that the rapeseed meal and oil are the marginal products avoided by the soybean meal and oil, respectively. Conversely, soybean hulls are most price competitive with maize grain in cattle feed; even though the former has lower total digestible nutrients, we considered a 1:1 ratio with maize because soybeans have a positive impact on feed intake and digestibility in cattle feed (Ipharraguerre and Clark 2003; Rankins 2015). Data on rapeseed coproducts were from the ecoinvent® database (Jungbluth et al. 2007), and the data for the maize production and processing were obtained from Alvarenga et al. (2012).

For maize processing, we consider that maize gluten meal feed and maize gluten meal (60) avoid the production of maize grain in a 1:1 ratio and urea used in animal feed in a 1:0.015 and 1:0.023 ratio, respectively (Santos 2004; Kim and Dale 2005; Pedroso et al. 2009). For maize oil, we assume that the avoided production of soybean oil is a 1:1 ratio (Kim and Dale 2005). Maize starch was assumed to replace wheat starch at the ratio of 1:1 because wheat is the second most used crop to produce starch and has similar compositions (i.e., fiber, lipids, protein, moisture, and starch) (European Commission 2002; International Starch Institute 2015). Data on wheat starch were based on the ecoinvent® database (Jungbluth et al. 2007) and Würdinger et al. (2002) for crop production and wheat processing, respectively.

Table 1 Multi-functional processes in swine production with allocation factors and avoided products used in the scenario variation

Stage/products	Case-by-case	Economic allocation	Mass allocation	Substitution method
Soybean processing	All scenarios			
Soybean meal	55.7	55.7	71.4	Rapeseed oil and maize grain
Soybean oil	41.7	41.7	19.4	Rapeseed meal and maize grain
Soybean hulls	2.6	2.6	9.3	Rapeseed meal and oil
Maize processing	All scenarios			
Maize starch	83.2	83.2	67.2	Wheat starch
Maize gluten feed	7.0	7.0	25.1	Maize grain and urea
Maize oil	3.1	3.1	2.7	Soybean oil
Maize gluten meal (gluten 60)	6.7	6.7	5.0	Maize grain and urea
Piglet production (PP)	Ref/flare/CHP/comp	Ref/flare/CHP/comp	Ref/flare/CHP/comp	Ref/flare/CHP/comp
Piglets	82.7	88.8/88.7/88.3/88.8	79.6/79.5/79.5/79.9	100.0
Sows	16.9	10.2/10.2/10.1/10.2	16.3/16.2/16.2/16.3	Poultry (live weight)
Boars	0.5	0.1/0.1/0.1/0.1	0.2/0.2/0.2/0.2	Poultry (live weight)
Organic fertilizer	Chemical fertilizer	0.9/1.0/1.0/0.9	4.0/4.1/4.1/3.6	Chemical fertilizer
Heat (only for Sce.CHP)	Wood heat in poultry production	0.5	n.a.[a]	Wood heat in poultry production
Electricity (only for Sce.CHP)	Brazilian electricity at grid	0.04	n.a.[a]	Brazilian electricity at grid
Growing to finishing (GF)	Ref/flare/CHP/comp	Ref/flare/CHP/comp	Ref/flare/CHP/comp	Ref/flare/CHP/comp
Swine (live weight)	100.0	98.0/98.0/96.9/98.2	94.4/94.2/94.2/95.0	100.0
Organic fertilizer	Chemical fertilizer	0.9/1.0/1.0/0.9	5.6/4.1/4.1/3.6	Chemical fertilizer
Heat (only for Sce.CHP)	Wood heat in poultry production	0.5	n.a.[a]	Wood heat in poultry production
Electricity (only for Sce.CHP)	Brazilian electricity at grid	0.04	n.a.[a]	Brazilian electricity at grid
Slaughtering house	All scenarios			
Swine carcass	86.9	96.7	86.9	100.0
Edible offal	6.8	3.1	6.8	Protein from poultry meat
Inedible offal	6.3	0.2	6.3	Poultry meal for PET feed production[b]

[a] Not applicable because it is not possible to state a mass for energy flows

[b] Some byproducts were not considered due to lack of data for the avoided product

In the animal production stage, it was assumed that sows and boars sent to slaughter replace poultry meat (live weight) at a 1:1 ratio. Data on poultry production were from Prudêncio da Silva et al. (2014). In the MMS, for all scenarios, manure avoids the production of chemical fertilizer. The Sce.CHP also considered the avoided production of electricity on the grid and wood-based heat.

Establishing an avoided product in the slaughtering stage was quite difficult because it is not clear that the edible offal could replace other products. However, if we consider that the main function of meat products is to provide protein, then one approach is to assume that the consumption of protein from edible offal can replace the protein consumption of other sources such as poultry meat, cattle beef or even vegetable sources. Therefore, due to the availability of data on poultry production in our database (Prudêncio da Silva et al. 2014), we assume that the protein provided by the coproducts of the slaughtering stage could avoid the production of a certain amount of protein from poultry carcasses (Table 1). We consider that swine blood and livers replace poultry meal in the production of PET feed on a protein-basis. No avoided production was considered for the intestinal mucous due to lack of data on the replaced product and because this byproduct has low environmental relevance for the product system. The intestinal mucous is often used to produce heparin and replaces mucosal tissues from bovine lungs.

2.3 Life cycle impact assessment (LCIA) methods

For the sensitivity analysis of the LCIA, five methods were compared at the midpoint level. The criteria used to select the methods were (a) the indicator at the midpoint level and (b) the possibility to assess at least four common impact categories across the methods. In this sense, the selected methods were the ReCiPe 2008 (H) midpoint, CML-IA, EDIP 2003, ILCD 2011, and TRACI 2.1. ReCiPe 2008 was used as the baseline method to compare the results of the sensitivity analysis because it was the method used in Cherubini et al. (2015a) to evaluate the environmental impacts of swine production.

To convert the impact scores into a common metric, we used the approach proposed by Dreyer et al. (2003) and recently used by Owsianiak et al. (2014). This method defines a new reference substance with a characterization factor for all compared methods and a reference substance in at least one of the compared methods. In methods that divide the eutrophication potential between freshwater and marine (e.g., ReCiPe 2008), the impact scores were aggregated using the Redfield conversion ratio between phosphate and nitrogen compounds (Goedkoop et al. 2013; Owsianiak et al. 2014). The conversion to a common unit is necessary to compare the impact categories across the methodologies. Table 2 shows the methods, the impact categories and the new reference substances.

2.4 Uncertainty analysis of the methodological choices in LCA

The uncertainties due to scenario variation were estimated using a Monte Carlo simulation that is a probabilistic modeling technique widely used to evaluate the uncertainties in input parameters and scenarios (Clavreul et al. 2012). This statistical method can run 10,000 independent repetitions representing the probability distribution of the scenarios results. Although some authors (Morgan and Henrion 1990; Gregory et al. 2016) argue that uncertainty generate by normative choices should not be treated probabilistically, it is technically feasible and has been done in the context of LCA (Huijbregts et al. 2003; de Koning et al. 2010; Mendoza Beltran et al. 2016; Gregory et al. 2016). Moreover, the effect of this uncertainty is similar to those generated due to data variability (parameter); in both cases we are not sure of the precise choice or value (Mendoza Beltran et al. 2016). Therefore, the use of probabilistic methods can be defendable (Mendoza Beltran et al. 2016). The input data used in this analysis were the minimum (x_Min) and the maximum (x_Max) impact scores for each impact category per scenario variation from the sensitivity analysis (see Fig. 2), assuming a uniform distribution (Mendoza Beltran et al. 2016; Gregory et al. 2016; AzariJafari et al. 2017) since we assume that all scenarios are equally likely due to lack of information on actual distributions.

3 Results

3.1 Sensitivity analysis of the allocation approaches

The summary of the LCA results due to the allocation approach selection is shown in Table 3. The percentages

Table 2 LCIA methods and impact categories used in the sensitivity analysis

Method	Impact categories	Original unit	New unit
ReCiPe 2008 (H) version 1.09	Climate change	kg CO_2 eq.	kg CO_2 eq.
(Goedkoop et al. 2013)	Terrestrial acidification	kg SO_2 eq.	kg SO_2 eq.
	Freshwater eutrophication Marine eutrophication	kg P eq. kg N eq.	kg PO_4 eq.
	Freshwater ecotoxicity	kg 1,4-dB eq.	kg 1,4-dB eq.
CML-IA version 3.01	Global warming potential	kg CO_2 eq.	kg CO_2 eq.
(Guinée et al. 2002)	Acidification potential	kg SO_2 eq.	kg SO_2 eq.
	Eutrophication potential	kg PO_4 eq.	kg PO_4 eq.
	Freshwater aquatic ecotoxicity	kg 1,4-dB eq.	kg 1,4-dB eq.
EDIP 2003 version 1.04	Global warming 100a	kg CO_2 eq.	kg CO_2 eq.
(Hauschild and Potting 2005)	Acidification	m^2	kg SO_2 eq.
	Aquatic eutrophication EP (N) Aquatic eutrophication EP (P)	kg N kg P	kg PO_4 eq.
	Ecotoxicity water chronic	m^3	kg 1,4-dB eq.
ILCD 2011 version 1.03	Climate change	kg CO_2 eq.	kg CO_2 eq.
(EC-JRC 2011)	Terrestrial acidification	molc H+ eq.	kg SO_2 eq.
	Freshwater eutrophication Marine eutrophication	kg P eq. kg N eq.	kg PO_4 eq.
	Freshwater ecotoxicity	CTUe	kg 1,4-dB eq.
TRACI 2.1 version 1.01	Global warming	kg CO_2 eq.	kg CO_2 eq.
(Bare et al. 2003)	Acidification	kg SO_2 eq.	kg SO_2 eq.
	Eutrophication	kg N eq.	kg PO_4 eq.
	Ecotoxicity	CTUe	kg 1,4-dB eq.

Table 3 Sensitivity analysis of scenario variation on the allocation procedures. The values highlighted in bold represent the lowest emissions per impact category for each scenario

Scenario variation	Climate change (kg CO_2 eq.)		Acidification (kg SO_2 eq.)		Eutrophication (kg PO_4 eq.)		Freshwater ecotoxicity (kg 1,4-dB eq.)	
Sce.Ref (ReCiPe method)		% of Case-by-case		% of Case-by-case		% of Case-by-case		% of Case-by-case
Case-by-case	**3503**		76		11.7		7.94	
Economic allocation	3896	+ 11	74	− 3	8.5	− 28	11.00	+ 39
Mass allocation	3634	+ 4	**63**	− 17	**8.0**	− 32	10.26	+ 29
Substitution method	4332	+ 24	84	+ 10	15.9	+ 35	**3.54**	− 55
Sce.Flare (ReCiPe method)		% of Case-by-case		% of Case-by-case		% of Case-by-case		% of Case-by-case
Case-by-case	**3389**		82		11.8		7.89	
Economic allocation	3842	+ 13	75	− 9	8.5	− 28	11.04	+ 40
Mass allocation	3581	+ 6	**65**	− 21	**8.0**	− 32	10.27	+ 30
Substitution method	4197	+ 24	91	+ 11	15.9	+ 35	**3.49**	− 56
Sce.CHP (ReCiPe method)		% of Case-by-case		% of Case-by-case		% of Case-by-case		% of Case-by-case
Case-by-case	**3114**		82		11.8		7.46	
Economic allocation	3548	+ 14	75	− 9	8.4	− 29	10.89	+ 46
Mass allocation	3323	+ 7	**65**	− 21	**8.0**	− 32	9.87	+ 32
Substitution method	3872	+ 24	91	+ 11	15.9	+ 35	**2.98**	− 60
Sce.Comp (ReCiPe method)		% of Case-by-case		% of Case-by-case		% of Case-by-case		% of Case-by-case
Case-by-case	**3552**		83		11.8		8.43	
Economic allocation	3962	+ 12	80	− 3	8.6	− 27	11.33	+ 34
Mass allocation	3705	+ 4	**69**	− 16	**8.2**	− 31	10.58	+ 25
Substitution method	4390	+ 24	92	+ 11	16.0	+ 35	**4.13**	− 51

demonstrate the impact variation compared to the case-by-case scenario of each alternative for manure management.

When considering only the substitution method for the product system, the results showed greater differences for the case-by-case scenarios. For instance, the ecotoxicity potential displayed a difference of up to 3.2.

In addition to the divergence in the absolute values, the scenario ranking did not show relevant changes independent of the approach adopted for the multi-functionality problems. Similar results were reported by other authors for climate change (Curran 2007; Kaufman et al. 2010; Cherubini et al. 2011), acidification, eutrophication, and freshwater ecotoxicity (Curran 2007; Luo et al. 2009).

However, these results should be interpreted carefully because minor changes in scenario ranking were noted for eutrophication. The rationale for neglecting these changes was that the impact scores were very similar for these impact categories with slight differences in the decimal values. For instance, Sce.Ref shows the lowest PO_4 eq. emissions compared to the other scenarios, except for the economic allocation (8.5 kg, Table 3). When the multi-functional processes were handled only by economic allocation, Sce.CHP showed the lowest eutrophication emissions, with 8.4 kg PO_4 eq. (difference of 0.9%). Therefore, the results are not conclusive and cannot be generalized for other product systems. Changes in scenario ranking due to different allocation approaches were reported in Luo et al. (2009) for climate change in a comparative LCA of fuels.

An interesting result is that the lowest emissions for each impact category varied according to the allocation approach used (the values highlighted in bold in Table 3), i.e., all case-by-case scenarios displayed lower values for climate change, while mass allocation had lower emissions for acidification and eutrophication. Conversely, the use of the substitution method to address the multi-functional processes in swine production showed the highest values for almost all of the impact categories, except freshwater ecotoxicity, in contrast to the results achieved by Cherubini et al. (2011), in which the substitution method represented the lower emissions for the main product.

For climate change, higher values in the scenarios with the substitution method are strongly related to soybean meal production. Soybean crops in west central Brazil are associated with deforestation impacts, resulting in higher CO_2 eq. emissions compared to rapeseed production, which is the avoided product production assumed for this scenario.

The decrease in the environmental impacts for acidification and eutrophication when considering only economic and mass allocation is partly due to the differences in the system boundaries. In the case-by-case and substitution method scenarios, the byproducts in the manure management system (MMS) stage avoid the production of chemical fertilizer, electricity and heat (the latter two are only in Sce.CHP). Hence, the boundaries are expanded to consider the manure application in soil and the consequentially avoided production of the aforementioned products. In economic and mass allocation scenarios, the system boundaries in the MMS stage end at the point of substitution, i.e., before manure application to the soil.

For freshwater ecotoxicity, the lower values from ReCiPe are due to the assumption that coproducts from soybean processing avoid the production of rapeseed meal and oil. Although rapeseed cultivation uses relatively low amounts of pesticide compared to the relatively high pesticide use for soybeans (Schmidt 2010), the type of pesticide used in each crop is one of reasons for this lower value. In rapeseed production, greater usage of pyrethroid insecticide cypermethrin is one of the causes of the ecotoxicity impacts (Schmidt 2010). For the ReCiPe method, the cypermethrin used in rapeseed has high ecotoxicity potential compared to diflubenzuron, which is used in soybean production (see Goedkoop et al. 2013 for more details). Thus, these results should be interpreted with caution once for rapeseed and soybean cultivation we use secondary data from ecoinvent® (Jungbluth et al. 2007) and Prudêncio da Silva et al. (2010), respectively. In either case, one can already note the influence of the LCIA method on the results and consequently on decision making (this issue is addressed in the next section).

The results presented in this section highlight the importance of a detailed explanation of the avoided product as well as the strong dependency of the LCA outcomes on the allocation approach selection. We also demonstrated that it is not possible to state that a specific allocation approach will always increase or decrease the impacts of the main product.

3.2 Sensitivity analysis of LCIA methods

The sensitivity analysis of the LCIA methods displayed almost no variations for climate change compared to the sensitivity generated by the allocation approach selection (Table 3). This behavior was expected because all LCIA methods follow the characterization model of IPCC with a time horizon of 100 years, except the IMPACT 2002+ method that considers a time horizon of 500 years (not evaluated here, see Jolliet et al. 2003). For acidification, eutrophication and freshwater ecotoxicity, the results were very sensitive to the LCIA method selection (Table 4).

Acidification and eutrophication potential are regional impact categories, i.e., very site-dependent. In addition, there are differences in the number of substances covered by each method, the inclusion of fate modeling and the characterization model and factors, e.g., only CML-IA also covers waterborne emissions for acidification (EC-JRC 2011; Hauschild et al. 2013). For eutrophication potential, CML-IA and TRACI 2.1 address both terrestrial and aquatic eutrophication (EC-JRC 2011).

Although Table 4 demonstrates differences in the final scores per impact category, using a substance contribution analysis, we observed that for all of the LCIA methods, the impacts on climate change and acidification were due to the carbon dioxide and ammonia emissions, respectively. For eutrophication, the phosphorus emissions were mainly responsible for the impacts in EDIP 2003, ILCD 2011, ReCiPe 2008, and TRACI 2.1, with nitrate being another main driver of the impacts in the latter LCIA method. For CML-IA, ammonia was the main contribution to the PO_4 eq. emissions. This divergence is likely because CML-IA did not separate the impacts of eutrophication in terrestrial and aquatic systems, as several of the LCIA methods (EC-JRC 2011).

For freshwater ecotoxicity, greater variations in the absolute values were observed. The EDIP 2003 method showed a result two orders of magnitude higher than our baseline method, ReCiPe 2008, i.e., an increase of up to 24,325% from the reference value in Sce.CHP. Rosenbaum et al. (2008), comparing seven methods for toxicity impact categories, observed differences in the characterization factor of up to 12 orders of magnitude, which can partly explain the high discrepancy for this impact category.

Specifically analyzing the results between ReCiPe 2008 and ILCD 2011, our findings disagreed with the outcomes obtained by Owsianiak et al. (2014), where minor differences for freshwater ecotoxicity were observed. The high dependency of the LCIA scores on the LCI associated with the very different environmental aspects per product category are not feasible for generalizing both results for other system products (e.g., the ones from this case study and from Owsianiak et al. 2014). In this sense, it is recommended to always conduct a sensitivity analysis in the LCIA method when the main purpose of an LCA is to decrease toxicity impacts.

The substance contribution analysis for freshwater ecotoxicity also demonstrates that it is not easy to see agreement across the LCIA methods. Different emissions were mainly responsible for the freshwater ecotoxicity impacts between the methods. Another remarkable difference was the small agreement among the pollutants with potential to impact freshwater ecotoxicity, i.e., only copper, nickel and zinc appear in all LCIA results with a contribution of higher than 1% of the total impacts. Similar results were found by Dreyer et al. (2003), evaluating the pollutant contribution on human toxicity for CML2001 (current CML-IA), EDIP97 and Eco-indicator 99.

Greater variations can be expected for freshwater ecotoxicity once toxicity impact categories can be modeled with a high variety of impact pathways. There are a large number of chemical substances used in industrial production that even the latest developments are sufficient to satisfactorily cover all the inventory flows (Geisler et al. 2005; Hauschild et al. 2013).

Table 4 Sensitivity analysis of LCIA method variation. The values highlighted in bold represent the lowest emission per impact category for each scenario

LCIA methods variation	Climate change (kg CO_2 eq.)		Acidification (kg SO_2 eq.)		Eutrophication (kg PO_4 eq.)		Freshwater ecotoxicity (kg 1,4-dB eq.)	
Sce.Ref (case-by-case)		% of ReCiPe		% of ReCiPe		% of ReCiPe		% of ReCiPe
ReCiPe 2008	3503		76		11.7		**7.94**	
CML-IA	3502	–	43	− 43	22.5	+ 92	201	+ 2432
EDIP 2003	3502	–	**43**	− 43	**11.5**	− 2	1915	+ 24,034
ILCD 2011	3503	–	72	− 5	12.7	+ 8	18.8	+ 137
TRACI 2.1	3502	–	61	− 20	12.4	+ 6	18.8	+ 137
Sce.Flare (case-by-case)		% of ReCiPe		% of ReCiPe		% of ReCiPe		% of ReCiPe
ReCiPe 2008	3389		82		11.8		**7.89**	
CML-IA	3388	–	47	− 43	23.3	+ 97	200	+ 2436
EDIP 2003	3388	–	**47**	− 44	**11.7**	− 1	1908	+ 24,071
ILCD 2011	3389	–	78	− 5	12.8	+ 8	18.8	+ 138
TRACI 2.1	3388	–	66	− 20	12.5	+ 6	18.8	+ 138
Sce.CHP (case-by-case)		% of ReCiPe		% of ReCiPe		% of ReCiPe		% of ReCiPe
ReCiPe 2008	3114		82		11.8		**7.46**	
CML-IA	3113	–	47	− 43	23.2	+ 98	189	+ 2438
EDIP 2003	3113	–	**47**	− 44	**11.7**	− 1	1823	+ 24,325
ILCD 2011	3114	–	78	− 5	12.8	+ 8	18.4	+ 147
TRACI 2.1	3113	–	66	− 20	12.5	+ 6	18.4	+ 147
Sce.Comp (case-by-case)		% of ReCiPe		% of ReCiPe		% of ReCiPe		% of ReCiPe
ReCiPe 2008	3552		83		11.8		**8.43**	
CML-IA	3551	–	**47**	− 43	26.3	+ 122	213	+ 2421
EDIP 2003	3551	–	47	− 43	**11.8**	–	2009	+ 23,717
ILCD 2011	3552	–	78	− 5	13.3	+ 12	19.2	+ 128
TRACI 2.1	3551	–	67	− 19	12.9	+ 9	19.2	+ 128

In addition to the greater differences between the methods, the overall ranking of scenarios did not change across the sensitivity analysis, i.e., Sce.Ref is the most favorable for decreasing the impacts on acidification and eutrophication, while Sce.CHP had lower emissions for climate change and freshwater ecotoxicity. Similar results were reported by Owsianiak et al. (2014). The agreement in scenario ranking can be expected when few processes dominate the impacts for all compared options (Huijbregts et al. 2010; Owsianiak et al. 2014). This was the case for swine production, where the impacts are driven by feed production (the same for the four options) and, to less extends, due to the manure management system (the compared options).

Nevertheless, this agreement in the scenario ranking is specific to our case study and to these impact categories. Another case study or even other variations of the LCIA methods at the endpoint level can generate contrasting results. For instance, Cavalett et al. (2013) found that ethanol presents lower potential environmental impacts with single-score results using ReCiPe endpoints, while gasoline is the most favorable when evaluated through IMPACT 2002+, Ecological Scarcity 2006 and Eco-indicator 99 (H).

It should further be considered that the uncertainty generated by the selected LCIA method is not due to an error in the models but comes from different assumptions and data used to model the environmental mechanism. In addition, occasional errors in the implementation of the characterization factors into software have been reported (Owsianiak et al. 2014), although we did not consider this issue in our evaluation.

3.3 Choice of allocation approach vs. LCIA method

To identify which methodological choice influences the results' variability more, we plotted the outcomes from the sensitivity analysis in scatter diagrams with the LCIA scores against the allocation approach categorized by the LCIA method. Our findings are displayed in Fig. 3 only for Sce.Ref because the methodological choices had the same behavior for all of the compared alternatives for MMS.

For climate change, the choice of allocation is mainly responsible for the uncertainty, while the LCIA method had no influence on the results, as already discussed (see section 3.2). Regarding acidification, both methodological choices led to uncertainties with major contribution from the LCIA methods,

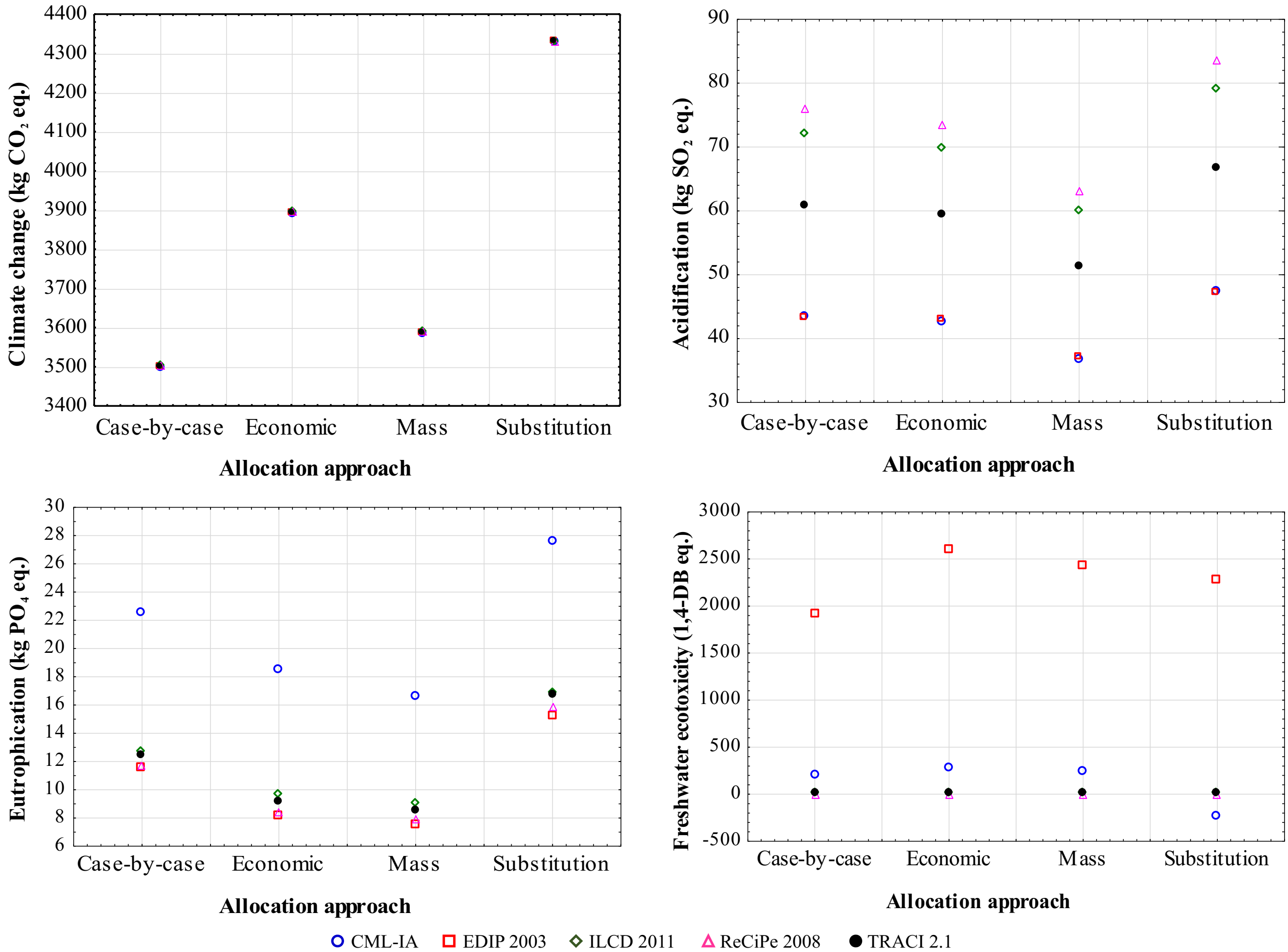
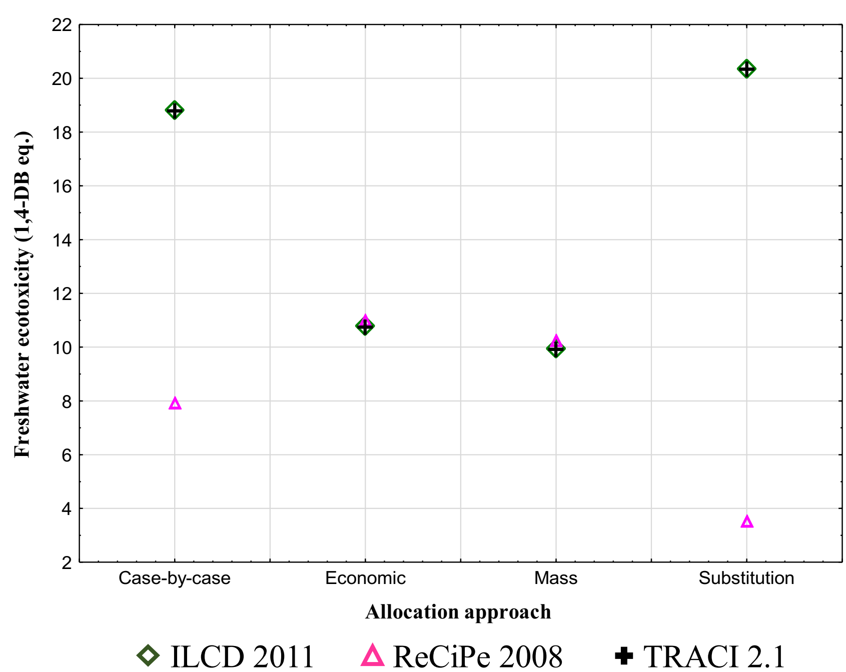


Fig. 3 Influence of the allocation approach selection and LCIA method on LCA outcomes

Fig. 4 Scatter plot of freshwater ecotoxicity (excluding CML-IA and EDIP 2003)



especially due to the characterization models from ReCiPe 2008, ILCD 2011 and TRACI 2.1. Similar behavior can be observed for eutrophication, differing only in the LCIA method, i.e., CML-IA.

For freshwater ecotoxicity, a first look at this figure could result in a misleading interpretation due to the high influence of CML-IA and (mainly) EDIP 2003. In Fig. 3, the LCI characterization through the ILCD 2011, ReCiPe 2008 and TRACI 2.1 appear to have minor effects on the allocation approach selection. However, analyzing the scatter diagram without CML-IA and EDIP 2003, we can observe that the allocation approach also contributes to the uncertainties in the LCA outcomes (Fig. 4). The graphics also demonstrate that CML-IA consider an environmental positive net benefit for freshwater ecotoxicity when the substitution method is used. The rationale for this behavior was explained in section 3.1 for the ReCiPe method but is also applied to CML-IA; it is related to the characterization factors for cypermethrin usage in rapeseed production. Therefore, both methodological choices introduce uncertainties for this impact category,

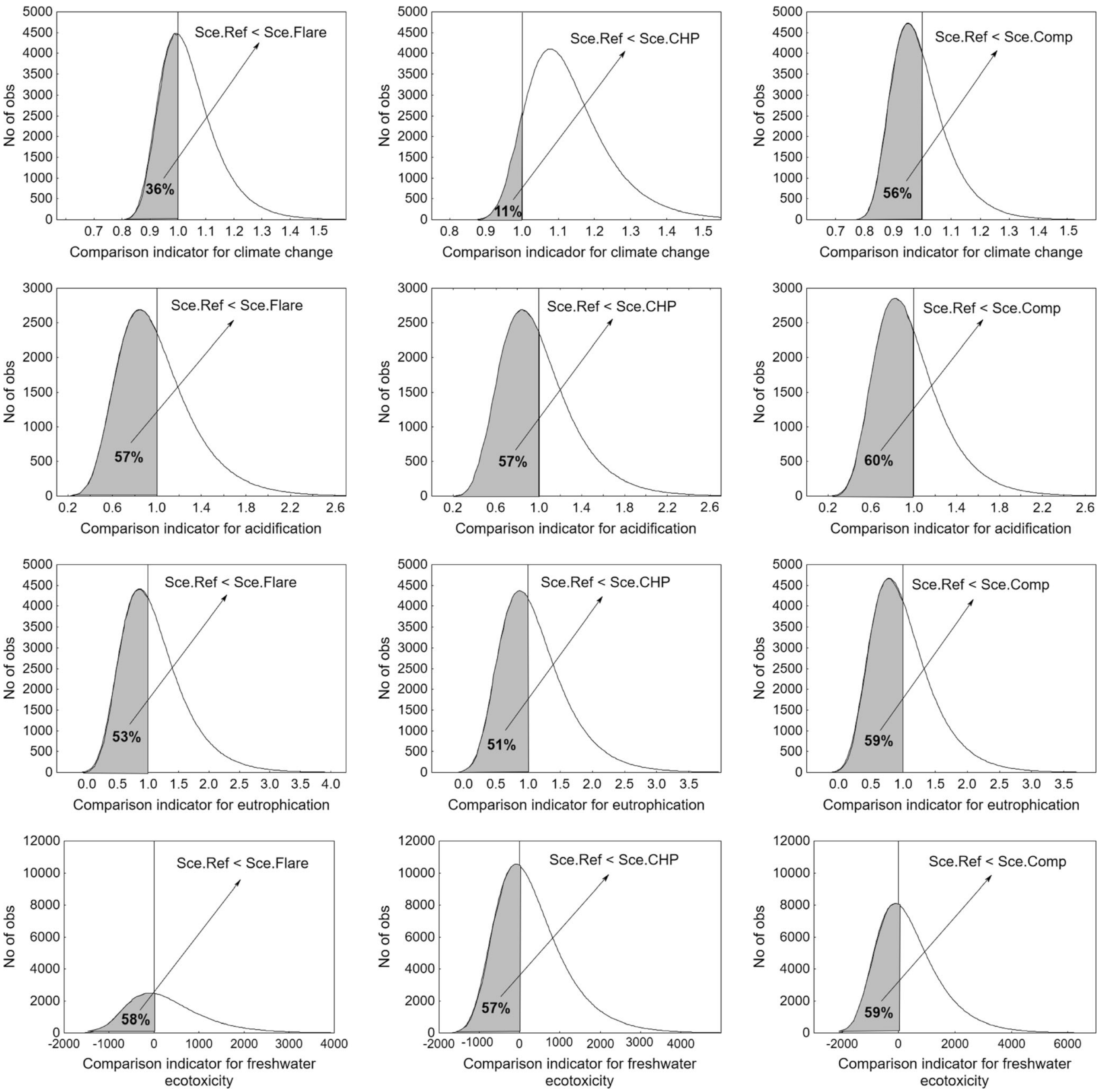


Fig. 5 Comparison indicators for the impact categories, including the scenario uncertainty. The shaded region corresponds to the likelihood that Sce.Ref has lower impact than the alternatives MMS

although major contributions can be expected from the LCIA methods, especially due to EDIP 2003.

3.4 Overall uncertainty in scenario comparison

The results of the comparison indicators are shown in Fig. 5, if the value is higher than 1, the MMS through the slurry tanks (Sce.Ref) has an impact score (e.g., kg CO_2 eq.) higher than the alternative MMS, and vice versa for a comparison indicator score lower than 1. Analyzing the results of the case study in Cherubini et al. (2015a) it seems, that Sce.Ref can decrease the impacts of acidification, eutrophication and freshwater ecotoxicity when compared to the alternative MMS. However, for almost all the comparison indicators from Fig. 5, the probability of Sce.Ref decrease in the environmental impacts is around 50%, i.e., in almost half of the cases, there is a probability that this choice will not be preferable, except to the comparison between Sce.Ref and Sce.CHP for climate change. These results mean that almost all the comparison indicators do not differ significantly and therefore it is not possible to indicate the best alternative to decrease the environmental impacts of our case study. When we consider the aggregate result variation across allocation approaches and LCIA methods, the uncertainty is too high to identify a statistically significant alternative. These conclusions are true even for climate change impact category that uses a more consolidated characterization model.

Although for our case study the choice of allocation procedure and LCIA method did not affected the scenario ranking, the results from Fig. 5 demonstrate that every important decision based on the LCA outcomes should be evaluated using an uncertainty or at least a sensitivity analysis due to the influence on the results that can be generated by the choices made by the LCA practitioner. One possible way to reduce the uncertainty in LCA is to search for scientific literature to validate the data used as well as the decisions made during the LCA (Soares et al. 2013; Gregory et al. 2016). For the environmental impact assessment, further developments in the models are expected, such as site-dependent models for regional impact categories (e.g., acidification and eutrophication) and the calculation of characterization factors for many new substances (Hauschild et al. 2013). Conversely, for the multifunctionality problem (i.e., the allocation approach), it seems to be more difficult to achieve improvements on the existing approaches, although it can be noticed some recommendations depending on the goals of the LCA or lifecycle stage (e.g., EC-JRC 2010; European Commission 2013).

4 Conclusions

The purpose of this paper was to evaluate the uncertainties in LCA of swine production due to two methodological choices; therefore, we addressed this issue with a deterministic model to infer the sensitivity in LCA methodological choices complemented by a Monte Carlo analysis. Our results have shown that climate change was the impact category most affected by the allocation approach, while acidification, eutrophication and freshwater ecotoxicity were most sensitive to the LCIA method used. The use of the substitution method to deal with the multi-functional processes introduced a greater variation in the absolute values of our analysis; this result is somewhat different than those found by Cherubini et al. (2011). Regarding the scenario comparison, the ranking of the best and worst alternatives did not change regardless of the allocation and LCIA method selection. However, some authors have found contrasting results for variations of these practitioner choices (e.g., Luo et al. 2009). Therefore, it is not a rule of thumb that the LCA outcomes will be consistent with different allocation approaches and LCIA methods, highlighting the importance of the sensitivity analysis in the interpretation phase.

Conversely, the uncertainty analysis shows that performing only a sensitivity analysis could mislead decision makers with respect to LCA results; our analysis using the Monte Carlo simulation indicates no significant difference between the alternatives compared for the impact categories. Therefore, a straightforward analysis with only a scenario variation might not detect this similarity between the options.

The method used on this paper was effective to demonstrate the uncertainty in the scenario comparison when considering the variation generated by different methodological choices. Although the uncertainty in the LCA outcomes could not be decreased due to the wide range of possible results, to some extent, the use of statistical methods can lead to a less uncertain decision making by showing the uncertainties between the compared alternatives. Furthermore, one must keep in mind that the LCI in our comparative case study were very similar. In this sense, another comparative case study in which the alternatives have more differences in LCI (e.g., plastic pallets vs wood pallets), the uncertainty analysis used in this paper, could point to the better alternative.

Despite the efforts made in the past years in an attempt to decrease the uncertainty in LCA outcomes, this topic still remains as a topic for further research. Some authors have already achieved interesting results dealing with one of the many methodological choices that a practitioner faces when conducting an LCA study. We decided to go further and aggregate the uncertainty due to two methodological choices and even though our uncertainty analysis used a robust statistical method such as Monte Carlo simulation, the uncertainty was too high to identify a statistically significant alternative.

Acknowledgments We would like to thank the National Council for Scientific and Technological Development (CNPq) for the financial support and to the anonymous reviewers for the important suggestions that considerably increased the quality of this paper.

References

Alvarenga RAF, Prudêncio da Silva V, Soares SR (2012) Comparison of the ecological footprint and a life cycle impact assessment method for a case study on Brazilian broiler feed production. J Clean Prod 28:25–32. https://doi.org/10.1016/j.jclepro.2011.06.023

AzariJafari H, Yahia A, Amor B (2017) Assessing the individual and combined effects of uncertainty and variability sources in comparative LCA of pavements. Int J Life Cycle Assess. https://doi.org/10.1007/s11367-017-1400-1

Bare JC, Norris GA, Pennington DW, McKone T (2003) TRACI: the tool for the reduction and assessment of chemical and other environmental impacts. J Ind Ecol 6:49–78

Baustert P, Benetto E (2017) Uncertainty analysis in agent-based modelling and consequential life cycle assessment coupled models: a critical review. J Clean Prod 156:378–394. https://doi.org/10.1016/j.jclepro.2017.03.193

Bovea MD, Gallardo A (2006) The influence of impact assessment methods on materials selection for eco-design. Mater Design 27(3):209–215. https://doi.org/10.1016/j.matdes.2004.10.015

Cavalett O, Chagas MF, Seabra JEA, Bonomi A (2013) Comparative LCA of ethanol versus gasoline in Brazil using different LCIA methods. Int J Life Cycle Assess 18(3):647–658. https://doi.org/10.1007/s11367-012-0465-0

Cellura M, Longo S, Mistretta M (2011) Sensitivity analysis to quantify uncertainty in life cycle assessment: the case study of an Italian tile. Renew Sust Energ Rev 15(9):4697–4705. https://doi.org/10.1016/j.rser.2011.07.082

Cherubini E, Zanghelini GM, Alvarenga RAF, Franco D, Soares SR (2015a) Life cycle assessment of swine production in Brazil: a comparison of four manure management systems. J Clean Prod 87:68–77. https://doi.org/10.1016/j.jclepro.2014.10.035

Cherubini E, Zanghelini GM, Tavares JMR, Belettini F, Soares SR (2015b) The finishing stage in swine production: influences of feed composition on carbon footprint. Environ Dev Sustain 17(6):1313–1328. https://doi.org/10.1007/s10668-014-9607-9

Cherubini F, Strømman AH, Ulgiati S (2011) Influence of allocation methods on the environmental performance of biorefinery products—a case study. Resour Conserv Recy 55(11):1070–1077. https://doi.org/10.1016/j.resconrec.2011.06.001

Clavreul J, Guyonnet D, Christensen TH (2012) Quantifying uncertainty in LCA-modelling of waste management systems. Waste Manag 32(12):2482–2495. https://doi.org/10.1016/j.wasman.2012.07.008

Curran MA (2007) Studying the effect on system preference by varying coproduct allocation in creating life-cycle inventory. Environ Sci Technol 41(20):7145–7151. https://doi.org/10.1021/es070033f

Dalgaard R, Schmidt J, Halberg N, Christensen P, Thrane M, Pengue WA (2008) LCA of soybean meal. Int J Life Cycle Assess 13(3):240–254. https://doi.org/10.1065/lca2007.06.342

de Koning A, Schowanek D, Dewaele J, Weisbrod A, Guinée J (2010) Uncertainties in a carbon footprint model for detergents; quantifying the confidence in a comparative result. Int J Life Cycle Assess 15(1):79–89. https://doi.org/10.1007/s11367-009-0123-3

Dreyer LC, Niemann AL, Hauschild MZ (2003) Comparison of three different LCIA methods: EDIP97, CML2001 and eco-indicator 99. Int J Life Cycle Assess 8(4):191–200. https://doi.org/10.1007/BF02978471

EC-JRC (2011) Recommendations based on existing environmental impact assessment models and factors for life cycle assessment in European context. ILCD Handbook—International Reference Life Cycle Data System, European Union EUR24571 EN. ISBN 978-92-79-17451-3. At http://publications.jrc.ec.europa.eu/repository/handle/JRC61049. Accessed 22 April 2015

EC-JRC (2010) General guide for life cycle assessment—detailed guidance. ILCD Handbook—International Reference Life Cycle Data System, European Union EUR24708 EN. ISBN 978-92-79-19092-6. At http://publications.jrc.ec.europa.eu/repository/handle/JRC48157. Accessed 13 January 2015

Ekvall T, Finnveden G (2001) Allocation in ISO 14041—a critical review. J Clean Prod 9(3):197–208. https://doi.org/10.1016/S0959-6526(00)00052-4

Ekvall T, Tillman A-M (1997) Open-loop recycling: criteria for allocation procedures. Int J Life Cycle Assess 2(3):155–162. https://doi.org/10.1007/BF02978810

European Commission (2002) Evaluation of the community policy for starch and starch products. European Commission—DG Agriculture. At http://ec.europa.eu/agriculture/eval/reports/amidon/full.pdf. Accessed 22 April 2015

European Commission (2013) 2013/179/EU: commission recommendation of 9 April 2013 on the use of common methods to measure and communicate the life cycle environmental performance of products and organisations. At http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A32013H0179. Accessed 29 April 2015

Finnveden G, Hauschild MZ, Ekvall T, Guinée JB, Heijungs R, Hellweg S, Koehler A, Pennington D, Suh S (2009) Recent developments in life cycle assessment. J Environ Manag 91(1):1–21. https://doi.org/10.1016/j.jenvman.2009.06.018

Gac A, Salou T, Espagnol S, Ponchant P, Dollé JB, van der Werf HMG (2014) An original way of handling co-products with a biophysical approach in LCAs of livestock systems. ACLCA, Vashon, WA, USA, San Francisco, USA

Galindro BM (2012) Análise técnica e avaliação do ciclo de vida de culturas de produção de microalgas para biodiesel. Master dissertation, Federal University of Santa Catarina. At http://www.ciclodevida.ufsc.br/publicacoes.php. Accessed 08 January 2015

Geisler G, Hellweg S, Hungerbühler K (2005) Uncertainty analysis in life cycle assessment (LCA): case study on plant-protection products and implications for decision making (3 pp). Int J Life Cycle Assess 10(3):184–192. https://doi.org/10.1065/lca2004.09.178

Goedkoop M, Heijungs R, Huijbregts MAJ, De Schryver A, Struijs J, van Zelm R (2013) ReCiPe 2008: a life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level. First edition report I: characterisation. RIVM, Bilthoven, May 2013. At http://www.lcia-recipe.net/file-cabinet Accessed 05 January 2015

Gregory JR, Noshadravan A, Olivetti EA, Kirchain RE (2016) A methodology for robust comparative life cycle assessments incorporating uncertainty. Environ Sci Technol 50(12):6397–6405. https://doi.org/10.1021/acs.est.5b04969

Guinée JB, Gorrée M, Heijungs R, Huppes G, Kleijn R, de Koning A, van Oers L, Wegener Sleeswijk A, Suh S, Udo de Haes HA, de Bruijn JA, van Duin R, Huijbregts MAJ (2002) Handbook on life cycle assessment: operational guide to the ISO standards. Series: eco-efficiency in industry and science. Kluwer Academic Publishers, Dordrecht (Hardbound, ISBN 1-4020-0228-9; Paperback, ISBN 1-4020-0557-1)

Hauschild M, Potting J (2005) Spatial differentiation in life cycle impact assessment—the EDIP2003 methodology. Environmental News no. 80. The Danish Ministry of the Environment, Environmental Protection Agency, Copenhagen

Hauschild MZ, Goedkoop M, Guinée JB, Heijungs R, Huijbregts MAJ, Jolliet O, Margni M, De Schryver A, Humbert S, Laurent A, Sala S, Pant R (2013) Identifying best existing practice for characterization modeling in life cycle impact assessment. Int J Life Cycle Assess 18(3):683–697. https://doi.org/10.1007/s11367-012-0489-5

Heijungs R (2014) Ten easy lessons for good communication of LCA. Int J Life Cycle Assess 19(3):473–476. https://doi.org/10.1007/s11367-013-0662-5

Heijungs R, Guinée JB (2007) Allocation and "what-if" scenarios in life cycle assessment of waste management systems. Waste Manag 27(8):997–1005. https://doi.org/10.1016/j.wasman.2007.02.013

Heijungs R, Lenzen M (2014) Error propagation methods for LCA—a comparison. Int J Life Cycle Assess 19(7):1445–1461. https://doi.org/10.1007/s11367-014-0751-0

Heijungs R, Suh S (2002) The computational structure of life cycle assessment. Springer Netherlands, Dordrecht. https://doi.org/10.1007/978-94-015-9900-9

Huijbregts MAJ (1998) Part II: dealing with parameter uncertainty and uncertainty due to choices in life cycle assessment. Int J Life Cycle Assess 3(6):343–351. https://doi.org/10.1007/BF02979345

Huijbregts MAJ, Gilijamse W, Ragas AMJ, Reijnders L (2003) Evaluating uncertainty in environmental life-cycle assessment. A case study comparing two insulation options for a Dutch one-family dwelling. Environ Sci Technol 37(11):2600–2608. https://doi.org/10.1021/es020971+

Huijbregts MAJ, Hellweg S, Frischknecht R, Hendriks HWM, Hungerbühler K, Hendriks AJ (2010) Cumulative energy demand as predictor for the environmental burden of commodity production. Environ Sci Technol 44(6):2189–2196. https://doi.org/10.1021/es902870s

Huijbregts MAJ, Norris G, Bretz R, Ciroth A, Maurice B, von Bahr B, Weidema B, Beaufort ASH (2001) Framework for modelling data uncertainty in life cycle inventories. Int J Life Cycle Assess 6(3):127–132. https://doi.org/10.1007/BF02978728

Hung M-L, Ma H (2009) Quantifying system uncertainty of life cycle assessment based on Monte Carlo simulation. Int J Life Cycle Assess 14(1):19–27. https://doi.org/10.1007/s11367-008-0034-8

International Starch Institute (2015) Statistic on starch raw materials, composition and worldwide use. At http://starch.dk/isi/stat/rawmaterial.asp. Accessed 22 April 2015

Ipharraguerre IR, Clark JH (2003) Soyhulls as an alternative feed for lactating dairy cows: a review. J Dairy Sci 86(4):1052–1073. https://doi.org/10.3168/jds.S0022-0302(03)73689-3

ISO (2006a) 14040:2006 Environmental management—life cycle assessment—principles and framework. International Standards Organization, Geneva (Switzerland)

ISO (2006b) ISO 14044:2006 Environmental management—life cycle assessment—requirements and guidelines. International Standards Organization, Geneva (Switzerland)

Jolliet O, Margni M, Charles R, Humbert S, Payet J, Rebitzer G, Rosenbaum R (2003) IMPACT 2002+: a new life cycle impact assessment methodology. Int J Life Cycle Assess 8(6):324–330. https://doi.org/10.1007/BF02978505

Jungbluth N, Chudacoff M, Dauriat A, Dinkel F, Doka G, Faist-Emmenegger M, Gnansounou E, Kljun N, Schleiss K, Spielmann M, Stettler C, Sutter J (2007) Life cycle inventories of bioenergy. Ecoinvent report no. 17. Swiss Centre for the Life Cycle Inventories, Dübendorf, Switzerland

Kaufman AS, Meier PJ, Sinistore JC, Reinemann DJ (2010) Applying life-cycle assessment to low carbon fuel standards—how allocation choices influence carbon intensity for renewable transportation fuels. Energy Policy 38(9):5229–5241. https://doi.org/10.1016/j.enpol.2010.05.008

Kim S, Dale B (2005) Life cycle assessment study of biopolymers (polyhydroxyalkanoates)—derived from no-tilled corn (11 pp). Int J Life Cycle Assess 10(3):200–210. https://doi.org/10.1065/lca2004.08.171

Luo L, van der Voet E, Huppes G, Udo de Haes HA (2009) Allocation issues in LCA methodology: a case study of corn stover-based fuel ethanol. Int J Life Cycle Assess 14(6):529–539. https://doi.org/10.1007/s11367-009-0112-6

Mendoza Beltran A, Heijungs R, Guinée J, Tukker A (2016) A pseudo-statistical approach to treat choice uncertainty: the example of partitioning allocation methods. Int J Life Cycle Assess 21(2):252–264. https://doi.org/10.1007/s11367-015-0994-4

Mendoza Beltran A, Chiantore M, Pecorino D, Corner RA, Ferreira JG, Cò R, Fanciulli L, Guinée JB (2017) Accounting for inventory data and methodological choice uncertainty in a comparative life cycle assessment: the case of integrated multi-trophic aquaculture in an offshore Mediterranean enterprise. Int J Life Cycle Assess. https://doi.org/10.1007/s11367-017-1363-2

Morgan MG, Henrion M (1990) Uncertainty: a guide to dealing with uncertainty in quantitative risk and policy analysis. Cambridge University Press, New York. https://doi.org/10.1017/CBO9780511840609

Owsianiak M, Laurent A, Bjørn A, Hauschild MZ (2014) IMPACT 2002+, ReCiPe 2008 and ILCD's recommended practice for characterization modelling in life cycle impact assessment: a case study-based comparison. Int J Life Cycle Assess 19(5):1007–1021. https://doi.org/10.1007/s11367-014-0708-3

Pedroso AM, Santos FAP, Bittar CMM (2009) Substituição do milho em grão por farelo de glúten de milho na ração de vacas em lactação em confinamento. Rev Bras Zootecn 38(8):1614–1619. https://doi.org/10.1590/S1516-35982009000800028

Pizzol M, Christensen P, Schmidt J, Thomsen M (2011) Eco-toxicological impact of "metals" on the aquatic and terrestrial eco-system: a comparison between eight different methodologies for life cycle impact assessment (LCIA). J Clean Prod 19(6-7):687–698. https://doi.org/10.1016/j.jclepro.2010.12.008

Prudêncio da Silva V, van der Werf HMG, Soares SR, Corson MS (2014) Environmental impacts of French and Brazilian broiler chicken production scenarios: an LCA approach. J Environ Manag 133:222–231. https://doi.org/10.1016/j.jenvman.2013.12.011

Prudêncio da Silva V, van der Werf HMG, Spies A, Soares SR (2010) Variability in environmental impacts of Brazilian soybean according to crop production and transport scenarios. J Environ Manag 91(9):1831–1839. https://doi.org/10.1016/j.jenvman.2010.04.001

Rankins D (2015) Feeding soybean hulls. Personal homepage, At http://www.auburn.edu/~rankidl/. Accessed 25 March 2015

Renou S, Thomas JS, Aoustin E, Pons MN (2008) Influence of impact assessment methods in wastewater treatment LCA. J Clean Prod 16(10):1098–1105. https://doi.org/10.1016/j.jclepro.2007.06.003

Rosenbaum RK, Bachmann TM, Gold LS, Huijbregts MAJ, Jolliet O, Juraske R, Köhler A, Larsen HF, MacLeod M, Margni M, McKone TE, Payet J, Schuhmacher M, van de Meent D, Hauschild MZ (2008) USEtox—the UNEP-SETAC toxicity model: recommended characterisation factors for human toxicity and freshwater ecotoxicity in life cycle impact assessment. Int J Life Cycle Assess 13(7):532–546. https://doi.org/10.1007/s11367-008-0038-4

Santos FA (2004) Glúten de milho na alimentação de aves e suínos. R Eletrônica Nutritime 1:79–100

Schmidt JH (2010) Comparative life cycle assessment of rapeseed oil and palm oil. Int J Life Cycle Assess 15(2):183–197. https://doi.org/10.1007/s11367-009-0142-0

Schmidt JH, Dalgaard R (2012) National and farm level carbon footprint of milk—methodology and results for Danish and Swedish milk 2005 at farm gate. Arla Foods, Aarhus, Denmark

Soares SR, Finotti AR, Prudêncio da Silva V, Alvarenga RAF (2013) Applications of life cycle assessment and cost analysis in health care waste management. Waste Manag 33(1):175–183. https://doi.org/10.1016/j.wasman.2012.09.021

Tillman A-M, Ekvall T, Baumann H, Rydberg T (1994) Choice of system boundaries in life cycle assessment. J Clean Prod 2(1):21–29. https://doi.org/10.1016/0959-6526(94)90021-3

Wardenaar T, van Ruijven T, Beltran A, Vad K, Guinée JB, Heijungs R (2012) Differences between LCA for analysis and LCA for policy: a case study on the consequences of allocation choices in bio-energy policies. Int J Life Cycle Assess 17(8):1059–1067. https://doi.org/10.1007/s11367-012-0431-x

Weidema BP (1999) System expansions to handle co-products of renewable materials. pp 45–48. At http://lca-net.com/files/casestudy99.pdf. Accessed 10 February 2015

Weidema BP, Schmidt JH (2010) Avoiding allocation in life cycle assessment revisited. J Ind Ecol 14(2):192–195. https://doi.org/10.1111/j.1530-9290.2010.00236.x

Weidema BP, Wesnæs MS (1996) Data quality management for life cycle inventories—an example of using data quality indicators. J Clean Prod 4(3-4):167–174. https://doi.org/10.1016/S0959-6526(96)00043-1

Würdinger E, Roth U, Wegener A, Peche R, Rommel W, Kreibe S, Nikolakis A, Rüdenauer I, Pürschel C, Ballarin P, Knebel T, Borken J, Detzel A, Fehrenbach H, Giegrich J, Möhler S, Patyk A, Reinhardt GA, Vogt R, Mühlberger D, Wante J (2002) Kunststoffe aus nachwachsenden Rohstoffen: Vergleichende Ökobilanz für Loose-fill-Packmittel aus Stärke bzw. Polystyrol, Umwelt Stiftung

Zamagni A, Buonamici R, Buttol P, Masoni P (2009) Main R&D lines to improve reliability, significance and usability of standardised LCA. ENEA, Italian National Agency on new technologies, energy and the environment