Review and assessment of published Life Cycle Assessment studies on bio-based fertilisers

Deliverable 6.1 – D25 – WP6

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OPTIMISING BIO-BASED FERTILISERS IN AGRICULTURE – PROVIDING A KNOWLEDGE BASIS FOR NEW POLICIES

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Deliverable 6.1 – D25

Work-package n°6

Nature of the deliverable		
R	Report	x
Dec	Websites, patents, filling etc.	
Dem	Demonstrator	
O	Other	

Dissemination Level		
PU	Public	x
CO	Confidential, only for members of the consortium (including the Commission Services)	

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LEX4BIO aims to reduce the dependence upon mineral/fossil fertilisers, benefiting the environment and the EU's economy. The project will focus on collecting and processing regional nutrient stock, flow, surplus and deficiency data, and reviewing and assessing the required technological solutions. Furthermore, socioeconomic benefits and limitations to increase substitution of mineral fertiliser for BBFs will be analysed. A key result of LEX4BIO will be a universal, science-based toolkit for optimising the use of BBFs in agriculture and to assess their environmental impact in terms of non-renewable energy use, greenhouse gas emissions and other LCA impact categories. LEX4BIO provides for the first-time connection between production technologies of BBFs and regional requirements for the safe use of BBFs.

The project runs from June 2019 to May 2023. It involves 20 partners and is coordinated by Luke (LUONNONVARAKESKUS - Natural Resources Institute Finland).

More information on the project can be found at: http://www.lex4bio.eu

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1. INTRODUCTION

The objective of WP6 is to perform a comparative Life Cycle Assessment (LCA) of BBFs, mineral fertilisers and traditional methods of using agricultural residues. The LCA will be based on a jointly established convention aiming at making future LCAs of fertilising products comparable, thus enabling policymakers, regulatory bodies and stakeholders at large to understand and compare the expected ecological impact of producing and using BBFs and mineral fertilisers.

For the purpose of designing such a convention, it is necessary to have a thorough overview of the way, fertilisers are currently treated in LCAs, analogies and differences between studies, as well as their main shortcomings. Therefore, a review of LCAs in the field has been conducted and is presented in the present deliverable D6.1.

2. METHODOLOGY

LCA studies included in the assessment were both transmitted to us by the LEX4BIO partners and network and collected by a query in Google Scholar.

In the latter case, query terms were adapted from a similar review by Brockmann et al. (2018)[1] as follows:

*OR (“*waste*”, “residu*”, “sludge*”, “sewage*”, “biosolid*”, “*compost*”, “digestate*”, “anaerobic”, “digest*”, “manure*”, “slurr*”, “effluent*”, “sediment”, “ash*”, “biochar”) AND (“Life Cycle Assessment”)*

The query was limited to publications from 2011-2021 and only open access articles published in peer-reviewed journals of the first 1500 results were considered.

[1] Brockmann, D.; Pradel, M. And Hélias, A., 2018. *Agricultural use of organic residues in life cycle assessment: Current practices and proposal for the computation of field emissions and of the nitrogen mineral fertilizer equivalent*. Resources, Conservation & Recycling, 133, 50-62. https://doi.org/10.1016/j.resconrec.2018.01.034

In a subsequent manual screening review studies, studies not conducting a LCA and studies not dealing with BBFs were excluded from the sample.

This left a total of 123 studies for the subsequent analysis (see Annex I). In order to analyse temporal trends, the sample was divided in two subsamples: those published in the period 2011-2015 (64 studies) and those from the period 2016-2021 (59 studies).

While we consider the sample large and variable enough to draw some general conclusions on the state of the art of BBFs in LCA, we do not claim for this review to be exhaustive, in terms that it would cover all LCA studies on BBFs covered in the respective periods.

3. RESULTS

3.1. Characteristics of collected LCA studies

3.1.1. Geographical background

Figure 1 shows the geographical background of the studies in the sample. Around half (64 out of 123) are conducted in Europe with most of them stemming from Denmark and Italy (12 studies each). Within Asia, most studies are conducted in China (13 out of 36 studies). 11 studies stem from the USA, whereas other regions only play a minor role in the sample. However, the number of European studies in the sample decreases between the periods 2011-2015 and 2016-2021 and here especially in countries other than Italy and Denmark. While five studies refer to several countries or a larger region (e.g. Scandinavia, North-Western Europe, etc.), there are no cross-continental studies in the sample. Two studies[2,3] do not refer the LCA to a specific region.

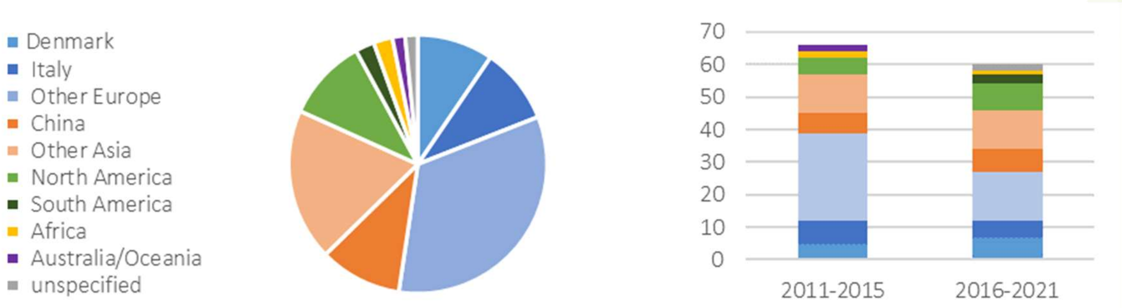


Figure 1: Geographical background of LCA studies in the sample and development over time.

3.1.2. Context

Three main thematic contexts could be identified in the sample, as shown in Figure 2. The majority of studies in the sample (69 of 123) deal with waste management, 36 studies have an agricultural background and 18 studies come from an anaerobic digestion (AD) context, although there are no sharp transitions between the categories. Farm-scale AD of agricultural waste (manure and crop residues) was generally considered in the agricultural context. While the number of studies with a waste management and agricultural context remains constant between the periods 2011-2015 and

[2] Amann et al. 2018 (full reference see Annex I)
[3] Eriksson et al. 2015 (full reference see Annex I)

2016-2021, the number of studies on AD decreases by 50% from 12 studies published between 2011-2015 to six studies in 2016-2021.

Studies on waste management are present in all geographical regions; however, while they make up 79% of studies conducted in Asian countries other than China, they constitute less than half of European studies and only 25% of Danish studies.

Studies with an agricultural context can be further divided into studies on crop production and studies on animal farming (mainly dealing with manure management), both being equally represented in the sample. Denmark (seven studies), China (six studies), Italy and Spain (five studies each) are the dominant regions, in which agricultural LCAs have been conducted. While in the former two countries more studies focus on animal production, all studies from Italy and Spain in the sample deal with crop production.

Two thirds of studies on AD have been conducted in a European context.

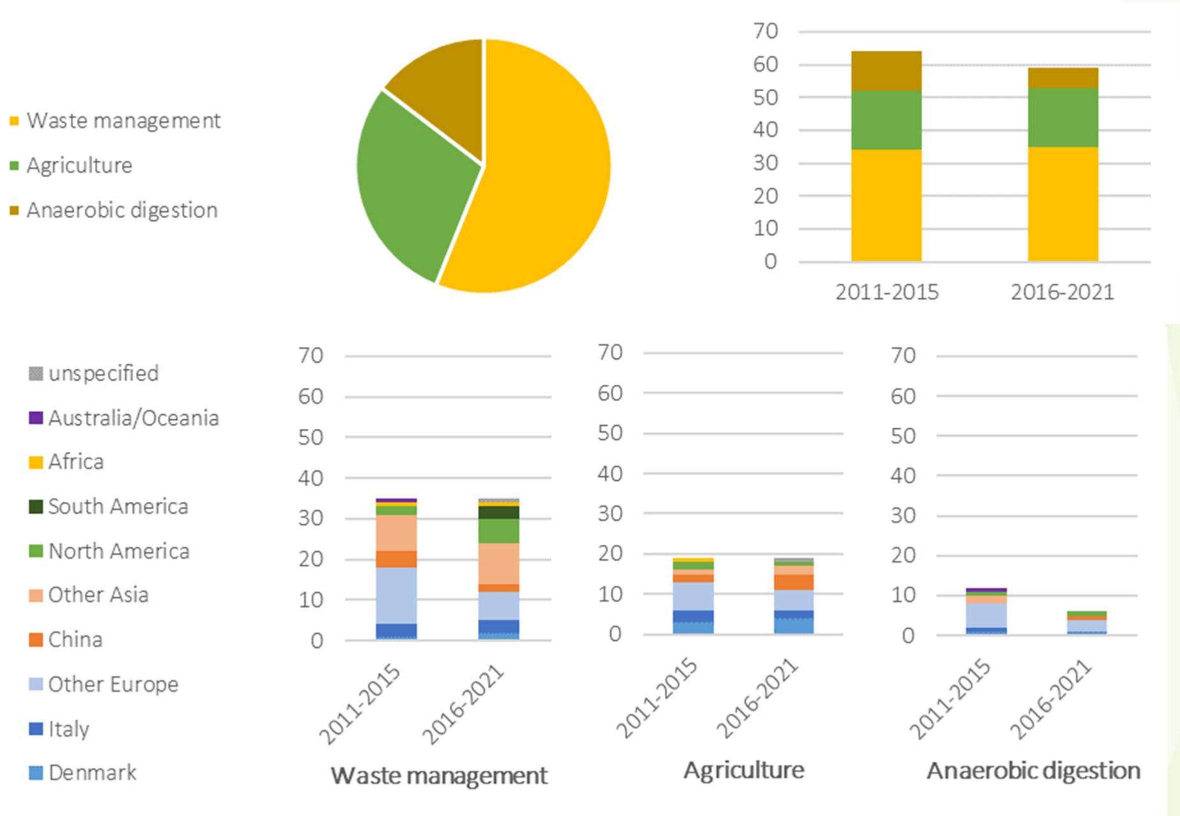


Figure 2: Context of LCA studies in the sample and geographical distribution within thematic background categories with development over time.

3.1.3. Substrates and treatments

The most frequently analysed substrate in the sample is municipal organic and food waste (49 out of 123 studies), followed by manure (45 studies) and sewage (24 studies). The former term includes separately collected municipal organic waste, as well as the compostable fraction of residual waste and food waste from households, canteens, restaurants and supermarkets. 37 studies analyse more than one type of substrate (e.g. because a comparison between the performance of different substrates in the LCA is made or because a combination of substrates is used as input to a treatment process), which is why the total number of substrate types analysed exceeds the number of studies in the sample in Figure 3. Although there are slightly less studies from 2016-2021 than from 2011-2015 in the sample, the number of substrate types analysed is higher in the latter period, meaning an increase in the number of substrate types analysed per study over time. Eight studies do not or only partly disclose the substrates considered for BBF production.

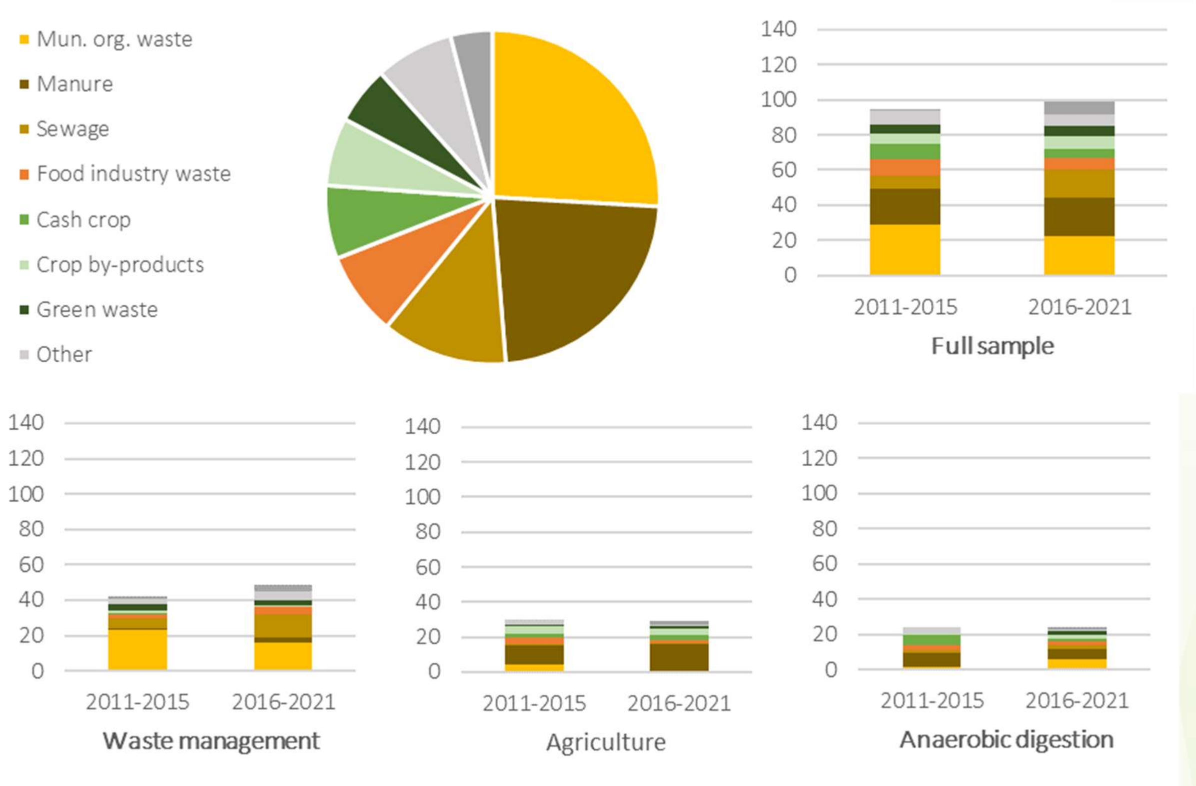


Figure 3: Substrates analysed in the sample including distribution among context categories and development over time.

Analysed treatment methods, on the other hand, are specified in all but one study[4,5] Like for substrates, studies frequently cover more than one treatment method, be it as a comparison of different treatments or through the application of a cascade of treatment technologies (e.g., AD followed by separation of liquid and solid digestate, manure separation followed by composting of the solid fraction, etc.). Composting and AD (72 and 67 studies out of 123, respectively) are the dominant forms of treatments analysed in the sample, followed by direct application (32 studies) and separation (25 studies), as shown in Figure 4. Between the period 2011-2015 and 2016-2021 the number of studies on composting and AD slightly decreases, whereas different forms of advanced treatments such as struvite precipitation, acidification or ash treatment appear more frequently. Furthermore, direct application is more frequently covered in the later period.

Reflecting the definition of context categories, manure is the most frequently analysed substrate type in agricultural and AD studies, whereas municipal organic and food waste as well as sewage dominate in studies from a waste management context. As may be expected, direct land application of BBFs is more common in studies from an agricultural context (16 out of 36 studies) than in studies from a waste management perspective (eight out of 69 studies) and in all but one study from an AD context

[4] Del Borghi et al. 2014 (full reference see Annex I)
[5] El Hannadeh et al. 2015 (full reference see Annex I)

the produced digestate is used as BBF. Over time, a shift from cash crops towards different forms of organic waste in studies from an AD context and from municipal organic and food waste to sewage and (to a lesser extent) industrial waste in studies from a waste management context can be observed. While especially in the waste management context advanced treatment technologies are increasingly studied, none of these technologies are covered in studies from an AD context. Regarding studies from an agricultural context almost all substrates analysed in the period 2016-2021 are generated within the sector. Simultaneously, direct land application is more frequently analysed.

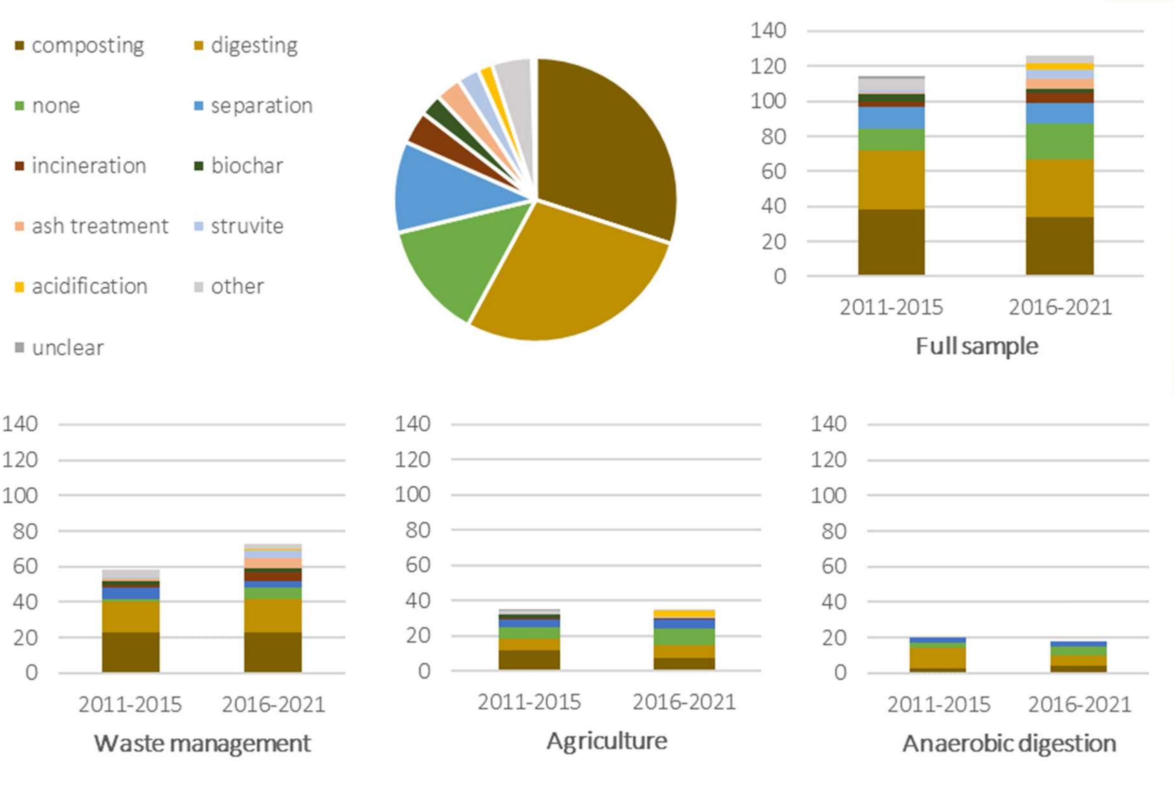


Figure 4: Treatment technologies analysed in the sample including distribution among context categories and development over time.

While there is a comparatively high correlation between studies on sewage sludge and studies on advanced treatment technologies (six out of 24 studies on sewage sludge deal with advanced treatment technologies), 40, respectively 36 out of 50 studies on municipal organic and food waste cover composting and AD. Together with direct land application, these treatment technologies are also the most frequently analysed in studies on manure and other agricultural products and by-products.

There are no major differences between regional backgrounds of the studies that cannot be explained by differences in other factors, except for a comparatively low representation of sewage in Asian studies from a waste management context (two out of 24 studies, compared to 12 out of 30 European studies and two out of seven North American studies).

3.1.4. ISO standard 14040 and 14044

ISO standard 14040 describes the basic principles and framework of LCA. Requirements and guidelines are further specified in ISO standard 14044. 90 of 123 studies in the sample claim compliance with the ISO standards, especially those with a European background (83%, compared to 69% of North

American and 61% of Asian studies). However, under a strict interpretation, none of the studies achieves full compliance with all requirements laid down in the standards. It should be noted though that the reports and articles collected often correspond to a summary report, while the full LCA is not disclosed. A detailed analysis of the most relevant criteria is given in the following chapters.

3.2. Study commissioner and audience

ISO standard 14040/14041 requires disclosure of both the study commissioner and the target audience. While 93 of 123 studies in the sample (76%) acknowledge the study commissioner, the opposite is true for the target audience: Only 25% of the studies in the sample specifically state to whom the analysis is addressed. Even of the studies claiming compliance with the ISO standard merely 28% meet the requirement for the target audience (compared to 16% of studies not claiming ISO compliance). However, for both criteria an increase in compliance can be observed between the periods 2011-2015 and 2016-2021 (from 70% to 83% for the disclosure of the study commissioner and from 20% to 29% for the disclosure of the target audience).

3.3. Goal definition

In all 123 studies the aim of the LCA is clearly defined making it the only requirement of the ISO standards met by all studies in the sample. Furthermore, in most studies the reasons for carrying out the analysis are elaborated (118 studies) and the conclusions drawn are in line with the goal definition (113 studies).

The vast majority of studies in the sample (108 out of 123) apply LCA in practices, while eight studies have a methodological approach and seven studies cover both aspects. All but one study[6] covering methodological issues have been conducted for Europe and of these most in an agricultural context (11 studies). The interest in methodological aspects of LCA increases slightly from the period 2011-2015 to 2016-2021.

Of the studies covering methodological issues five are developing or comparing models for nutrient uptake and emissions in the field. Two studies analyse spatial effects, one in life cycle inventory[7], one in impact assessment[8]. Another two studies propose new impact categories, one regarding odour emissions[9], one for soil quality[10].

In most studies a comparison between different scenarios is undertaken, most often between different treatment technologies for a substrate (81 out of 123 studies), especially in studies with a waste management context (55 out of 69 studies). Different substrates to a treatment technology are compared in 17 studies, eight of which stemming from an AD context, while 15 studies compare the application of BBFs to mineral fertiliser (MF), 10 of which in an agricultural context, and eight studies each compare different practices (e.g. conventional and organic agriculture) or technological configurations of a treatment method (see Figure 5). The former being more common in an agricultural context (10 out of 15 studies), the later in a waste management context (five out of eight studies). 13 studies pursue several aims, the most common combinations being with four studies each the simultaneous comparison of different treatment technologies and different substrates, as well as of

[6] Owsianiak et al. 2018 (full reference see Annex I)
[7] Hanserud et al. 2017 (full reference see Annex I)
[8] Owsianiak et al. 2018 (full reference see Annex I)
[9] Peters et al. 2014 (full reference see Annex I)
[10] Oberholzer et al. 2012 (full reference see Annex I)

different treatment technologies and MF and BBF. Between the periods 2011-2015 and 2016-2021 the share of studies comparing different treatments increases from 52% to 69%, mainly at the expense of studies comparing different substrates, whose share decreases from 17% to 8%. Notably, none of the studies in the sample on the comparison of different substrates has been conducted in an Asian context.

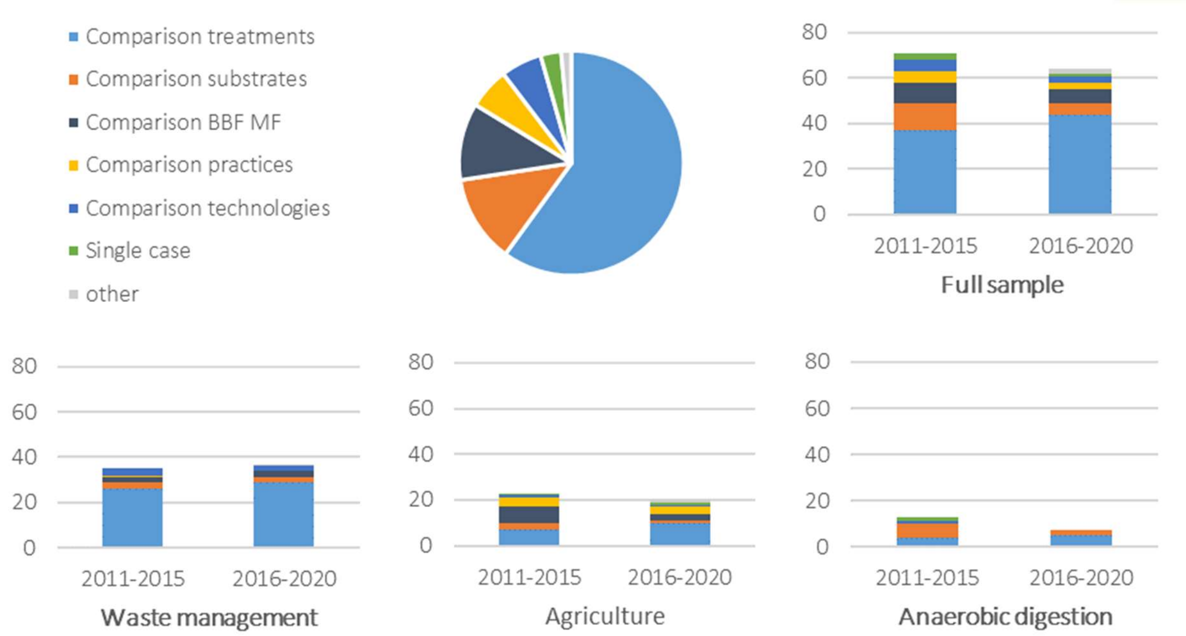


Figure 5: Study goals in the sample including distribution among context categories and development over time.

3.4. Functions and functional unit

According to the ISO standard 14040/14044 the functional unit (FU) constitutes the basic entity in LCA which refers to the main function of the system and to which all other flows are scaled. While in all but two studies in the sample[11,12] the system's main function is clear, only 16% (19 out of 123 studies) provide an explicit definition, although their share increases from 13% to 19% between the periods 2011-2015 and 2016-2021. Secondary functions included and/or omitted from the analysis are described in 77 out of 123 studies, with a slight increase in the more recent period.

106 out of the 123 studies in the sample provide a clear and measurable FU, with a slightly lower rate of compliance in studies not claiming to adhere to the ISO standards (24 of 32 studies). Moreover, in the large majority of cases the FU is defined in a way that ensures comparability between different scenarios and is consistent with the goal (98, respectively 104 studies). Of the 13 North American studies four studies each do not meet these criteria, though.

Almost twice as many studies use an input related FU, e.g., the quantity of substrates put into a treatment technology, as use an output related FU, e.g., the quantity of BBF produced (see Figure 6). Studies comparing treatment technologies particularly often use input related FUs (66 out of 83 studies) which is why they are particularly dominant in studies with a waste management context and

[11] Vázquez-Rowe et al. 2015 (full reference see Annex I)
[12] Ankathi et al. 2018 (full reference see Annex I)

show an increase over time. However, in the period 2016-2021 few studies also use both input- and an output related FUs.

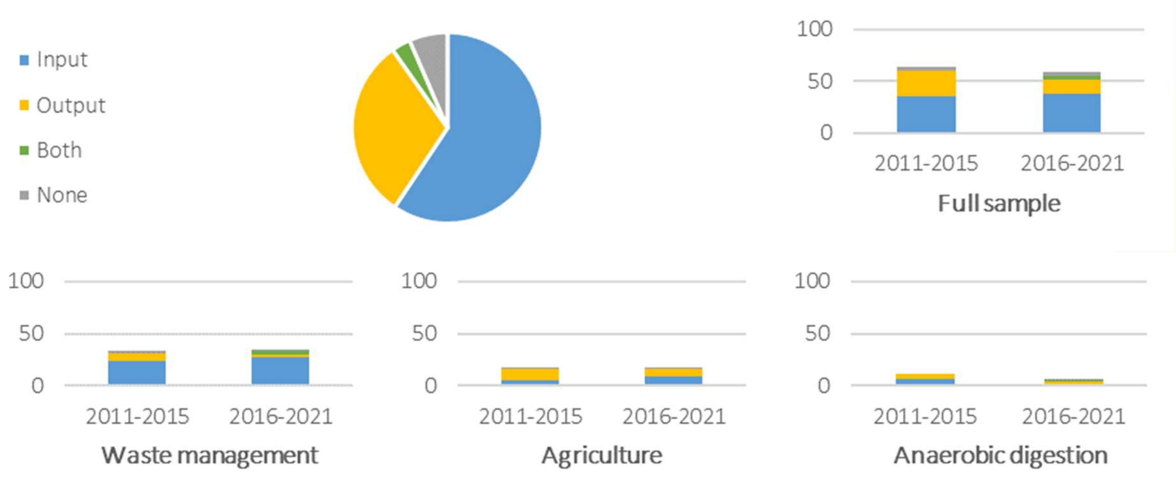


Figure 6: Functional unit used by studies in the sample including distribution among context categories and development over time.

3.5. System boundaries

3.5.1. System boundaries

Like for the FU, compliance with the ISO standard regarding the definition of system boundaries is generally high: comprehensive and consistent system boundaries are defined in 106 of the 123 studies in the sample, 108 studies illustrate the system boundaries in a flow diagram and 97 studies provide descriptions of all input and output flows to each unit process (either in written or graphical form).

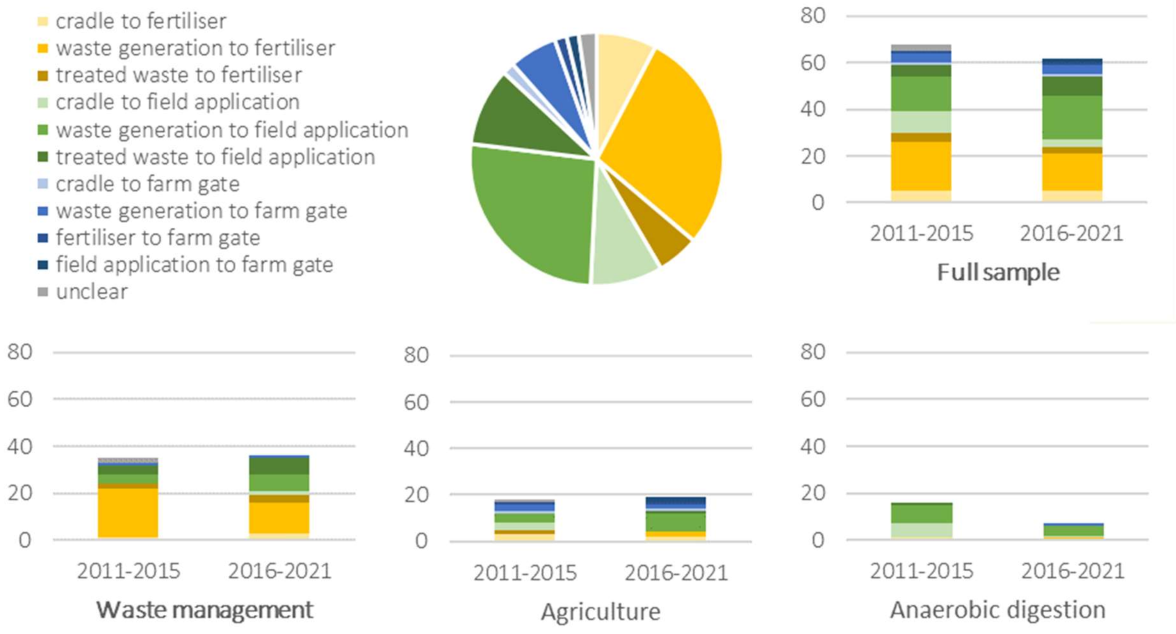
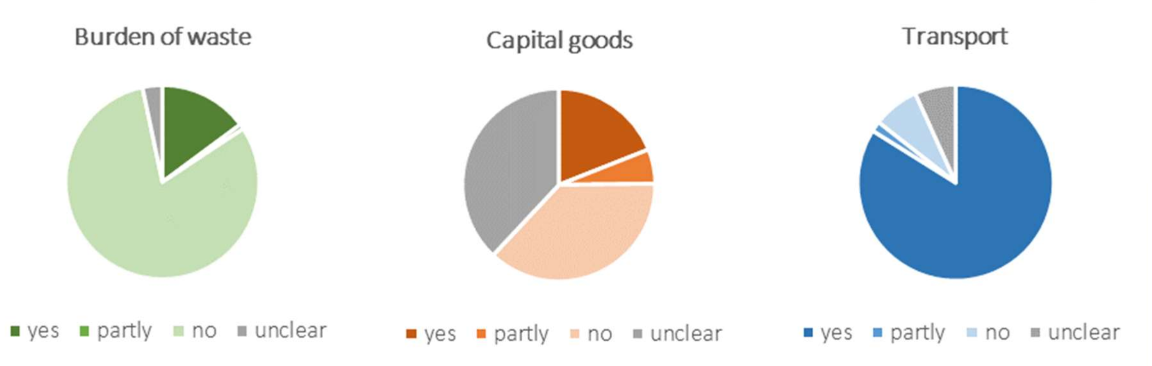


Figure 7: System boundaries of the studies in the sample including distribution among context categories and development over time.

Waste generation is with 79 of the 123 studies in the sample the dominant input boundary used, irrespective of the study context. 24 studies, especially from an agricultural or AD context, define the input boundary of the system with the cultivation of crops or in case of manure as a substrate animal farming (i.e., the "cradle"). Studies starting the analysis with treated waste are predominately those dealing with sewage sludge (14 out of 20 studies), where the step of wastewater treatment is often neglected. The output boundary, on the other hand, is highly dependent on the study context. While 61% of studies from a waste management context end the analysis with the generated fertiliser product (42 out of 69 studies), 72% of studies from an agricultural context (26 out of 36 studies) and all but two studies from an AD context[13,14] include fertiliser field application in the system boundaries. Especially studies aiming at the comparison of different (agricultural) practices often analyse the whole cultivation process up to the farm gate. However, in the period 2016-2021 also half of the studies from a waste management context, predominately those with a European context, include fertiliser field application within the system boundaries (see Figure 7).

3.5.2. Cut-off criteria

Flows and processes not considered relevant for the LCA results are eliminated from the system via cut-off. ISO standard 14040/14044 recommends defining cut-off criteria in a physical, energy or environmental perspective (e.g., flows constituting together less than 10% of the total mass or accounting for less than 10% of the environmental impact are neglected) and to verify them in a sensitivity analysis. This has only been followed by one[15] and seven studies in the sample, respectively. Notably, regarding the latter, all studies stem from a European context and six were published in the period 2016-2021.
Waste entering the system is commonly considered as burden-free, i.e., upstream impacts are cut-off to avoid double-counting (impacts should already have been considered in the precedent system generating the waste). However, Pradel and Aissani (2019)[16] argue that if the production of fertiliser from sewage sludge becomes a lucrative business for wastewater treatment plants in the future, part of the impacts of wastewater treatment should be allocated to the sludge. Only 30 of the 123 studies in the sample consider impacts from capital goods, whereas 45 studies explicitly cut off capital goods. 46 studies do not disclose whether capital goods are included in the analysis, which could also point towards a low importance given to them. On the other hand, impacts from transport are considered relevant in 101 of the 123 studies in the sample (see Figure 8). Notably, 9 of the 15 studies aiming at a comparison between MF and BBF at least partly include capital goods in the analysis.



[13] Langlois et al. 2012 (full reference see Annex I)
[14] Tonini et al. 2016 (full reference see Annex I)
[15] Kim et al. 2013 (full reference see Annex I)
[16] Full reference see Annex I

Figure 8: Cut-off of upstream impacts of waste, capital good and transport in the studies in the sample.

3.6. Multifunctionality

3.6.1. Compliance with ISO standard 14040/14044

Multifunctionality in LCA occurs for instance when a process produces several co-products. It can be solved either by allocating environmental impacts to the co-products based on criteria such as mass, economic value, etc., or by expanding the system, for instance to an alternative production route of one of the co-products, which can then be subtracted from the original system. The ISO standard 14040/14044 prescribes a hierarchy where system expansion should be preferred over allocation, where possible. Within allocation, allocation criteria based on physical relationships should be preferred over economic ones. Furthermore, the ISO standard emphasizes the disclosure of allocation criteria for waste (here meaning the waste generated by the processes within the system boundaries).

100 of the 123 studies in the sample consider problems of multifunctionality at least implicitly and 75 studies follow the prescribed hierarchy. However, only 23 studies contain a description of allocation criteria for waste. Specifically, 74 studies rely exclusively on system expansion, seven on allocation and 19 studies use a hybrid approach of both techniques. While there are no major differences over time in studies with a waste management context, exclusive application of system expansion is more common in the period 2016-2021 in studies with an agricultural or AD context, mainly at the expense of a hybrid approach. In 21 studies the principle applied to solve multifunctionality is unclear for at least part of the problems and eight studies explicitly neglect at least one issue of multifunctionality, with a decrease between the periods 2011-2015 and 2016-2021.

3.6.2. Allocation

Among the 26 studies using allocation, mass and economic value are the most common allocation bases (see Figure 9). In the period 2011-2015 a higher variety can be observed, which may partly be due to the fact that three studies[17,18,19] explicitly study the impacts of different allocation basis on the LCA results. 73% of allocation issues (including all those using economic allocation) concern the allocation of a process's impacts to its outputs, while 27% (all but one stemming from a waste management context) allocate impacts to process inputs.

[17] Rehl et al. 2012 (full reference see Annex I)
[18] Lijó et al. 2014 (full reference see Annex I)
[19] Quiros et al. 2015 (full reference see Annex I)

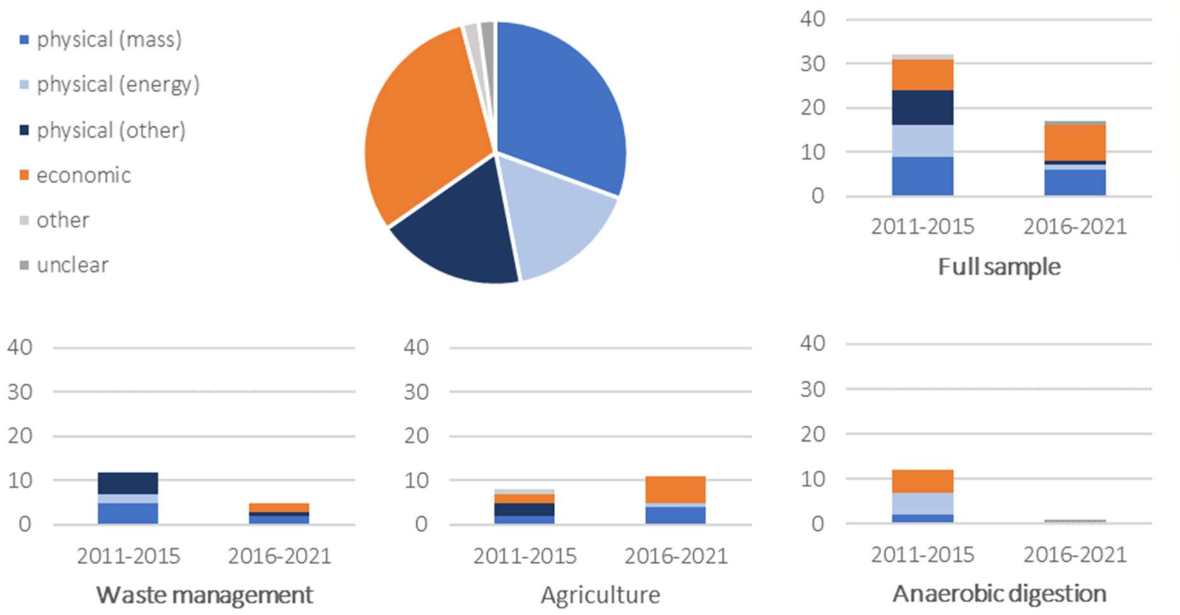


Figure 9: Allocation basis for all multifunctionality problems of the studies in the sample which are solved by allocation including distribution among context categories and development over time.

3.6.3. Substitution

Substitution is an integral part of system expansion, as multifunctionality in this case is solved by defining an alternative (substituting) provisioning route for one of the co-functions. However, it also occurs if environmental credits are assigned to a product which replaces a another. Furthermore, comparisons of products fulfilling the same function follow the same principles. Therefore, all three cases are considered in the following analysis.

3.6.3.1. Substitution of energy

Energy is substituted in 77 out of 123 studies in the sample, of which 70 substitute electricity, 32 heat and 16 fuel. In general, energy substitution is more common in European studies (applied in 59%, 42% and 13% for electricity, heat and fuel, respectively, compared to 30%, 3% and 6% in Asian studies and 13%, 2% and 3% in North American studies).

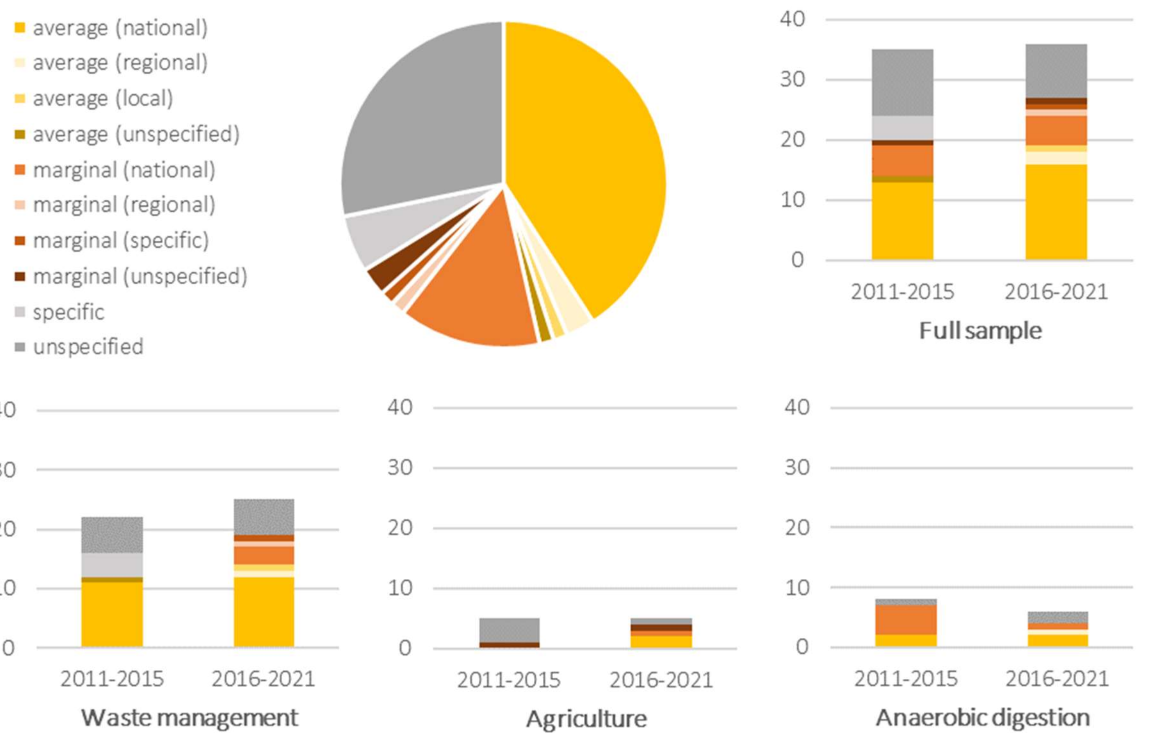


Figure 10: Substitution of electricity in the studies in the sample including distribution among context categories and development over time.

Electricity produced commonly replaces the national grid electricity of the region under study (i.e., following an attributional LCA approach), whereas heat and fuel are more often replacing the marginal commodity on the market (i.e., following a consequential LCA approach). All but one study[20] using the marginal approach have a European background. Regarding the market size, a national perspective predominates. However, in a substantial part of studies the rational which substitution is based on is not defined; moreover, in a few studies from the period 2011-2016 a case specific energy commodity independent from the market situation, is substituted (see Figures 10 and 11). As energy production is a main purpose of AD, energy substitution is with 14, 13 and six out of 18 studies for electricity, heat and fuel, respectively, particularly common in this context. On the other hand, less than one third (10 out of 36) of studies from an agricultural context substitute electricity, while heat and fuel are merely substituted in four and two studies, respectively. Furthermore, in 50% of the studies substituting energy in an agricultural context it remains unclear, what substitution is based on, compared to just 21%, 15% and 33% for electricity, heat and fuel, respectively, for studies with an AD context.

[20] Nordahl et al. 2020 (full reference see Annex I)

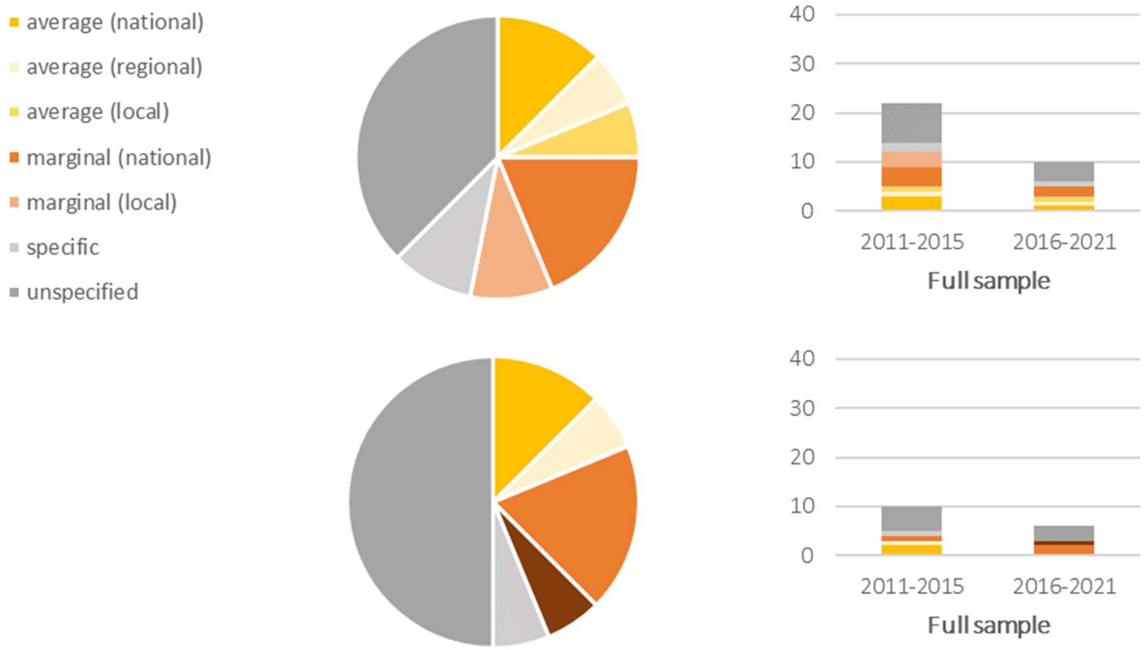


Figure 11: Substitution of heat (top) and fuel (bottom) in the studies in the sample including distribution among context categories and development over time.

3.6.3.2. Substitution of fertiliser

97 of the 123 studies in the sample consider environmental credits for BBFs due to the substitution of MFs or undertake a direct comparison between a BBF and MF. However, in more than one third of cases the assumptions taken regarding the nutrient equivalency of the different fertiliser types are not disclosed. In 25 studies or 25% of cases where substitution is applied to a BBF containing potassium (K), fertilising effects of K are disregarded (of this, six studies explicitly state to have neglected K fertiliser due to low relevance). For nitrogen (N) and phosphorus (P) this is only the case in 2% and 6% of substitution cases, respectively (see Figure 12). In a further 11%, 8%, and 14% it is not clear, whether fertilising effects of N, P and K, respectively have been considered in the substitution process. Moreover, even where assumptions regarding nutrient equivalency are explained, a simple one-to-one approach is frequently applied, not taking differences in nutrient efficiencies between different fertiliser types into account. Furthermore, in a significant part of studies it is not clear, whether a nutrient is substituted, or the type of MF substituted remains unspecified. This is true for 11%, respectively 25% of substitution cases for N, 8%, respectively 19% for P and 14%, respectively 25% for K. However, reporting on the substituted fertiliser tends to improve between the periods 2011-2015 and 2016-2021 (see Figure 13), whereas the share of studies not stating their assumptions regarding nutrient equivalency and neglecting one or more nutrients present in the BBF is higher in the later period. Even in studies explicitly aiming at the comparison of BBFs and MF, two out of 15 studies do not disclose assumptions on fertiliser substitution[21,22] and three at least partly do not specify the type of MF substituted[23,24,25].

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[21] Schmid Rivera et al. 2017 (full reference see Annex I)
[22] Montemayor et al. 2019 (full reference see Annex I)
[23] Knudsen et al. 2014 (full reference see Annex I)
[24] Schmid Rivera et al. 2017 (full reference see Annex I)
[25] Montemayor et al. 2019 (full reference see Annex I)

Studies in an AD context seem to pay the highest attention to fertiliser substitution issues as they exhibit both the lowest rate of substitution cases using a one-to-one approach and the highest rate of cases where the type of substituted fertiliser is specified. On the other hand, in studies from a waste management context the proportion of substitution cases where it is not clear, what nutrients have been considered, is highest and one-to-one substitution is most frequently applied. Irrespective of the study context, fertiliser substitution seems to be more thoroughly undertaken in European than in Asian studies. In the former assumptions on nutrient equivalency (other than the one-to-one approach) are specified for 62% of substitution cases for N, 49% for P and 13% for K, compared to 22% for N, 11% for P and 9% for K in the latter. Similarly, specification of substituted fertiliser type is more frequent in European studies.

Regarding the type of fertiliser substituted there is a high variety between the studies with ammonium nitrate (AN) and calcium ammonium nitrate (CAN) only featuring in European studies, whereas the substitution of another BBF (e.g. compost) or peat with the BBF under study is more common in studies with an Asian or North American background. Specific values for the amount of MF substituted (i.e. mineral fertiliser equivalents, MFE) are only provided in 22 studies for N, 21 studies for P and 14 studies for K. For N fertiliser equivalency, a high variety can be observed, with solid forms of BBF tending to have a lower MFE than liquid forms and compost tending to have a lower MFE than (raw) manure and digestate (see Figure 14). The majority of studies (14 for P and 11 for K) apply a MFE of 100%. Three[26,27,28] and one study[29] explicitly study influence of P, respectively K, MFE on LCA results. Of the remainder, Cubas do Amaral et al. (2018)[30] apply a P-MFE of 31% for ash, Miller 2013[31] uses a MFE of 19% for P and 45% for K for a specifically designed “BioA”-fertiliser, Morelli et al. (2018)[32] apply a P-MFE of 73% for compost and Turner et al. (2016)[33] 50% for P and 80% for K for both compost and digestate. Differentiated consideration of P and K MFE thus seems to be a topic of recent years, predominately in a European context.

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[26] Hanserud et al. 2018 (full reference see Annex I)
[27] Yoshida et al. 2018 (full reference see Annex I)
[28] Ten Hoeve et al. 2018 (full reference see Annex I)
[29] Hanserud et al. 2018 (full reference see Annex I)
[30] Full reference see Annex I
[31] Full reference see Annex I
[32] Full reference see Annex I
[33] Full reference see Annex I

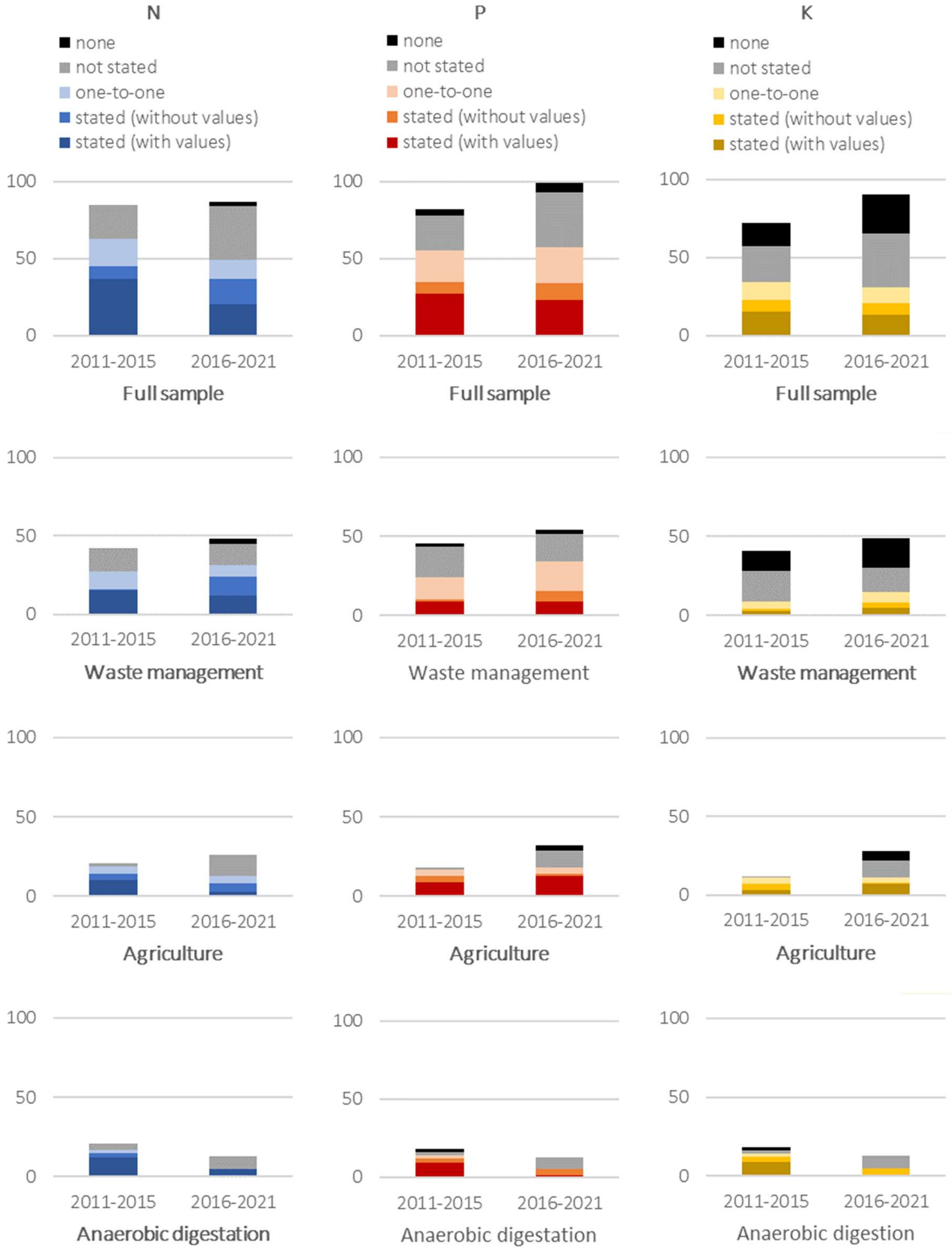

Figure 12: Reporting of assumptions on nutrient equivalency for fertilisers replaced with BBFs in the studies in the sample including distribution among context categories and development over time. None: nutrient present in BBF is disregarded in substitution.

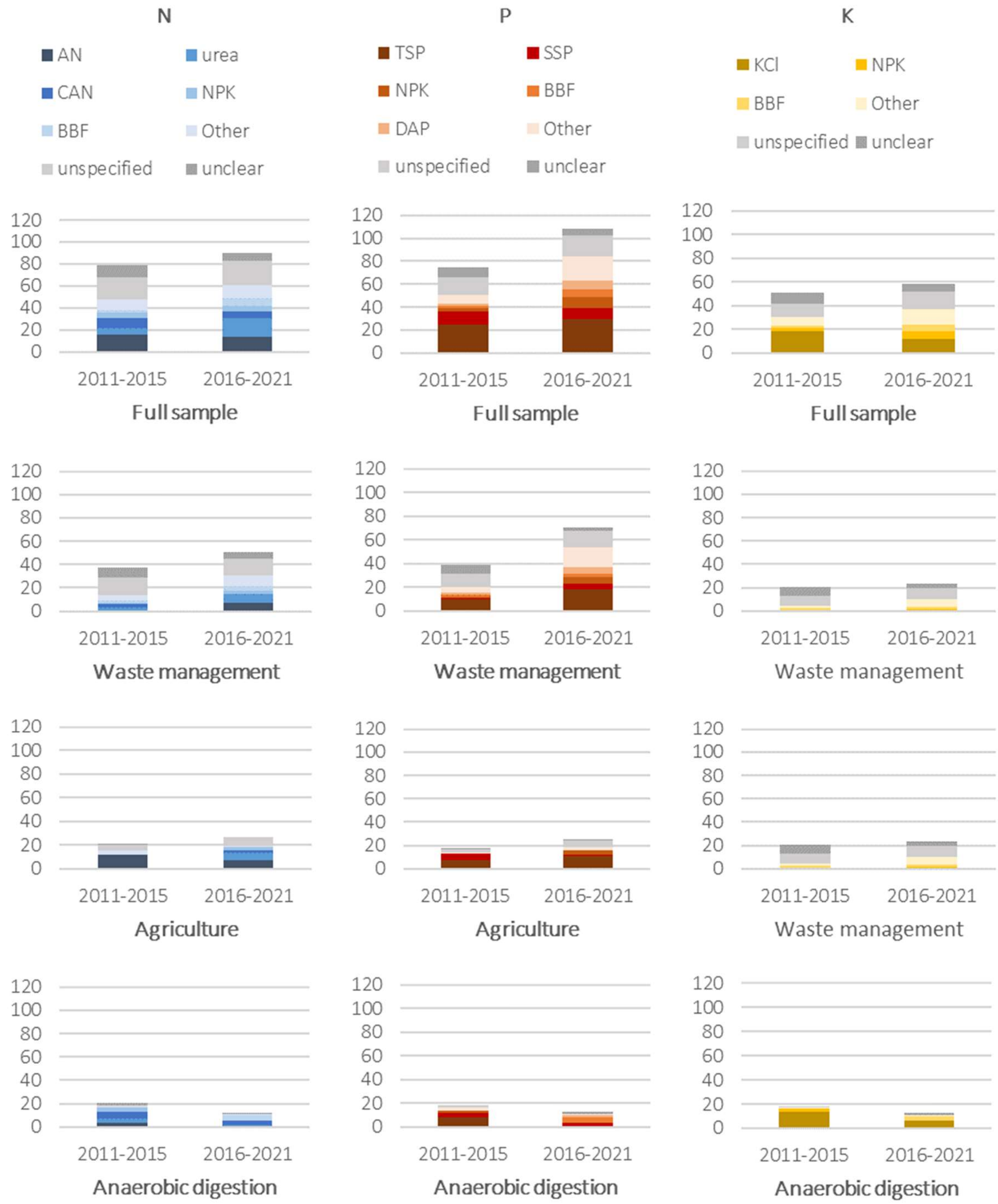

Figure 13: Type of fertiliser assumed to be replaced by BBF in the studies in the sample including distribution among context categories and development over time. AN: ammonium nitrate, CAN: calcium ammonia nitrate, TSP: triple super phosphate, SSP: single super phosphate, DAP: diammonium phosphate, KCl: potassium chloride, unclear: unclear whether the nutrient present in the BBF has been considered in the substitution.

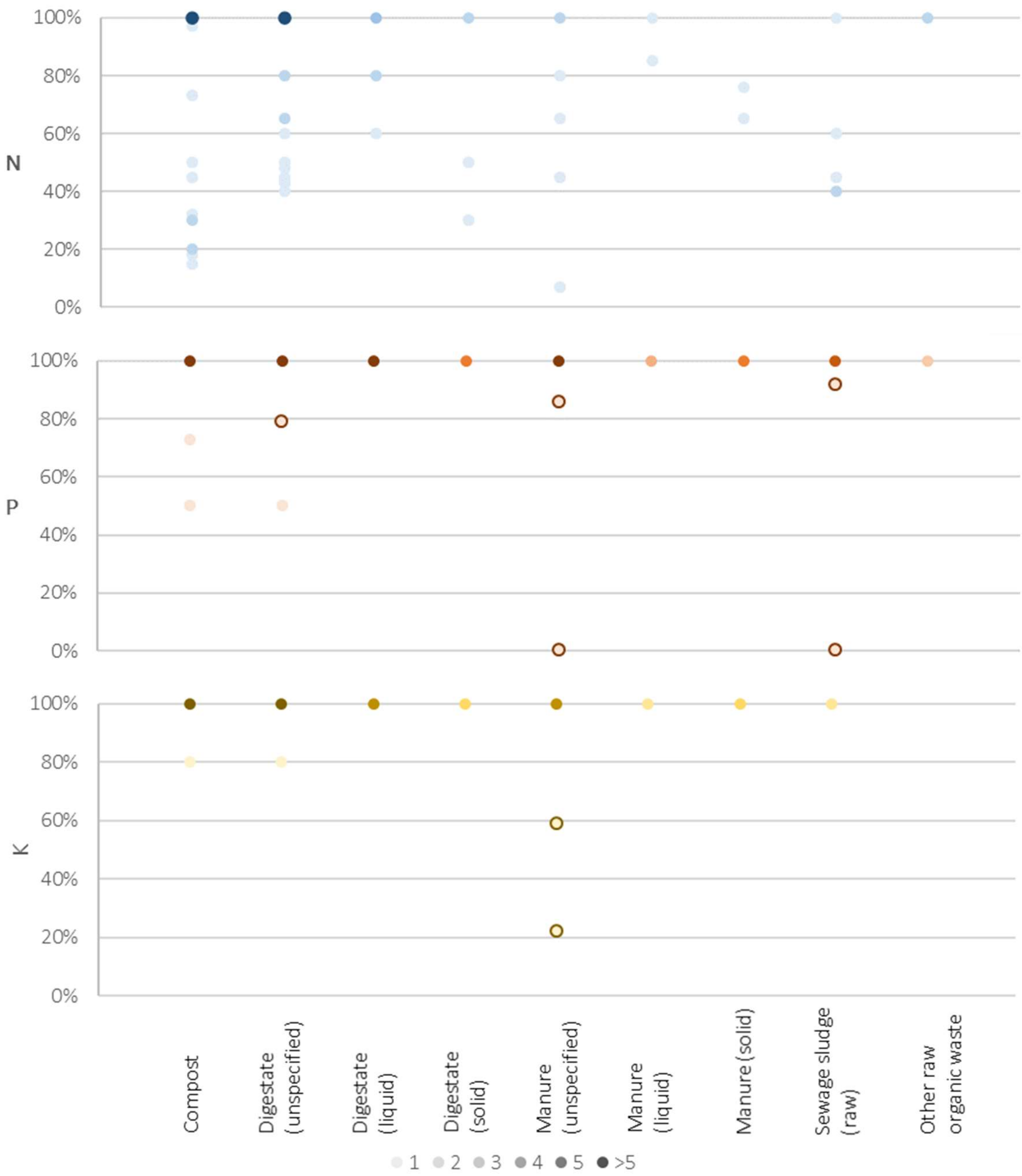

Figure 14: Nutrient equivalencies applied for most common BBFs in the sample. Values stemming from studies comparing the impact of different assumptions on nutrient equivalency on the LCA results are indicated with a dark outline.

3.7. Life cycle inventory

3.7.1. Data sources

107 out of the 123 studies in the sample report the sources of data used in the assessment. However, in 33 studies it is not clear what requirements regarding temporal, regional and technological correspondence to the system under study have been applied during data collection. Similarly, merely half of the studies (62 out of 123) disclose procedures followed if the required data could not be obtained. Moreover, only 23 studies attempt to quantify data uncertainties and no more than

three[34,35,36] analyse representativeness, consistency and reproducibility of the data used. Nevertheless, compliance with requirements of the ISO 14040/14044 has improved over time; for instance, 91% of studies in the period 2016-2021 report data sources and 31% data uncertainties, compared to 84% und 17% in the period 2011-2015. In addition, attention to data quality issues seems to be higher in North American studies (six and nine out of 13 studies report data uncertainty and procedures for missing data, respectively).

Literature is the most frequently used data source (105 out of 123 studies) followed by databases (91 studies) and operational data (74 studies). On average three different types of data sources are used per study. The average number of data source slightly increases between the periods 2011-2015 and 2016-2021 (from 3.2 to 3.4). Modelling and calculation as well as databases are more frequently used in the recent period, whereas the use of experimental data decreases (see Figure 15). There are no large differences among study context or geographical background. Studies with a waste management context (especially those from Asia) tend to use operational data (50 out of 69 studies) and estimations (24 studies) more frequently, while only one third of studies with an AD context uses operational data.

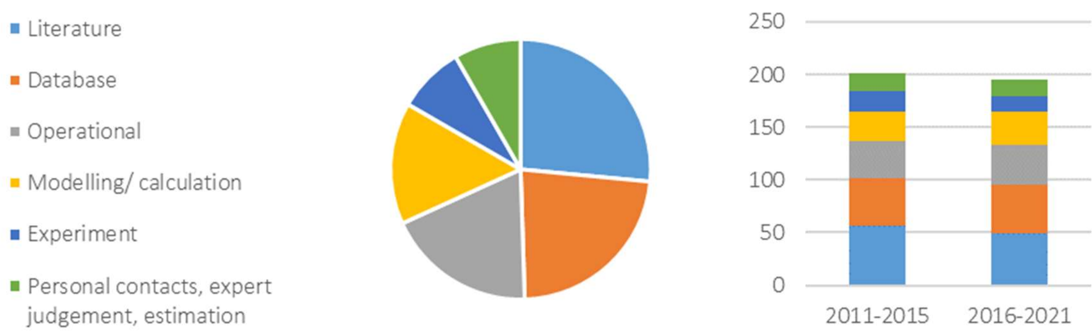

Figure 15: Types of data sources used by the studies in the samples with development over time.

3.7.2. Assumptions on electricity

Apart from the approach to solving multifunctionality (described in Chapter VIII) the type of electricity used by processes may have significant impacts on the LCA results. Therefore, the ISO standard 14040/14044 contains a requirement to disclose assumption on electricity use. This is met by 71 out of 123 studies in the sample, with a higher compliance in studies with an AD context (72% compared to 51% with an agricultural and 58% with a waste management context).

3.7.3. Assumptions on field emissions

As 73 out of 123 studies in the sample (see Chapter VII) include application of fertiliser in the system boundaries, assumptions taken on field emissions play an important role for the studies in the sample.

[34] Vázquez-Rowe et al. 2015 (full reference see Annex I)
[35] Montmayor et al. 2019 (full reference see Annex I)
[36] Kooduvalli et al. 2020 (full reference see Annex I)

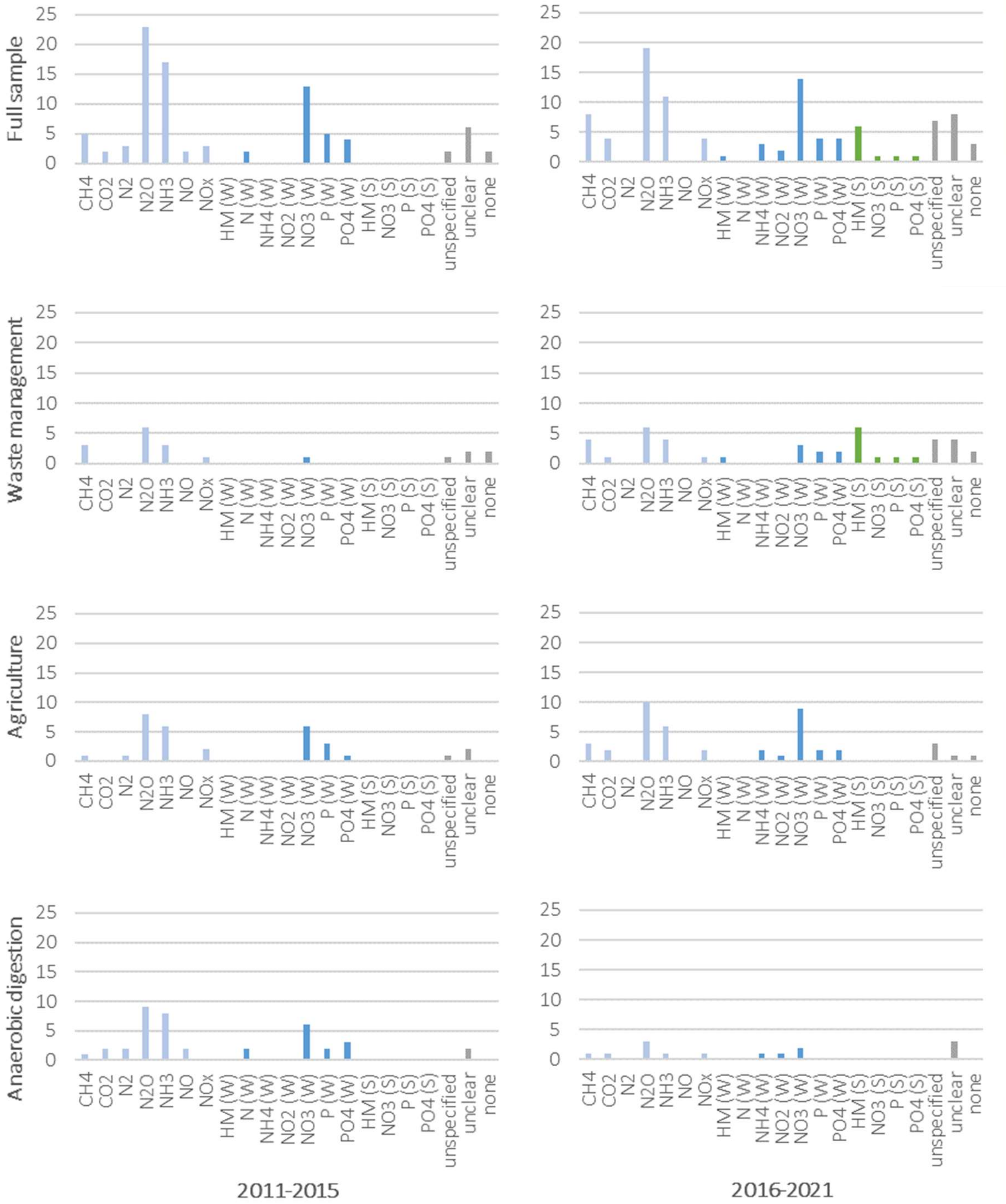

Figure 16: Inclusion of field emissions in the studies in the sample including distribution among context categories and development over time. W: emissions to water; S: emissions to soil; unspecified: type of field emissions considered not specified; unclear: unclear whether field emissions have been considered.

Inclusion of fertiliser field application does not necessarily mean, that field emissions are included in the assessment: 23 studies do not specify whether field emissions have been considered and five explicitly neglect them. For the remainder, nitrous oxide (N_2O; 42 studies), ammonia (NH_3; 28 studies) and nitrate (NO_3; 28 studies) are the most frequently analysed substances. Emissions of P are analysed in 19 studies, of which 17 address emissions to water, two to soil. Emissions of carbon (C) are also assessed in 19 studies: 13 studies deal with methane (CH_4) emissions, six with carbon dioxide (CO_2). Heavy metals are covered in seven studies, of which six are looking at emissions to water and one at emissions to soil. The number of substances analysed increases from 2011-2015 to 2016-2021, however, N_2O, NH_3 and NO_3 remain the most frequently considered (see Figure 16). Notably, six of the seven studies dealing with heavy metal emissions have been conducted in a waste management context and half of those stem from Asia and the period 2016-2021. This makes heavy metals, together with N_2O, the most frequently analysed type of field emissions in this context. Furthermore, CH_4 emissions are with seven out of 30 studies comparatively often analysed in a waste management context (compared to seven and four studies considering NH_3 and NO_3 emissions, respectively). However, for half of the studies with a waste management context that include fertiliser field application field emissions are neglected or it is not clear whether or what kind of substances are included.

3.8. Impact assessment

3.8.1. Compliance with ISO standard 14040/14044

All but one study[37] define impact categories analysed in the impact assessment and in 113, respectively 104 out of 123 studies characterisation methods and characterisation factors are properly defined. Furthermore, in the large majority of studies impact categories and characterisation factors which are representative for the problem under study (118 studies) and internationally accepted (108 studies) are chosen. Regarding the latter, two studies aiming at developing new impact indicator categories (one for odour emissions[38], one for soil quality[39]) are included in the studies considered non-ISO compliant. On the other hand, comparatively few studies provide a justification for the impact categories chosen (61 studies), include impact categories which are not mass- or energy based (24 studies) or which are spatially differentiated (14 studies) and verify the suitability of the chosen impact categories and characterisation methods in a sensitivity analysis. (29 studies). Yet, among the 13 North American studies eight use with the TRACI approach a spatially differentiated impact assessment method (see also below).

3.8.2. Impact categories

Midpoint impact categories (i.e., common mechanisms for a variety of substances affecting humans and ecosystems, such as radioactive forcing with respect to climate change[40]) clearly dominate in all study contexts, regional backgrounds, and periods. Only two and three studies rely exclusively on flow- or stock related impact categories (e.g., energy demand) and endpoint impact categories (i.e., final impacts on human health, the natural environment and natural resources[41]), respectively. However, eight out of 13 studies from a North American context include at least one flow- or stock related indicator.

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[37] Fernandez-Lopez et al. 2015 (full reference see Annex I)

[38] Peters et al. 2014 (full reference see Annex I)

[39] Oberholzer et al. 2012 (full reference see Annex I)

[40] European Commission -Joint Research Centre -Institute for Environment and Sustainability, 2010. *International Reference Life Cycle Data System (ILCD) Handbook -Framework and Requirements for Life Cycle Impact Assessment Models and Indicators*. First edition March 2010. EUR 24586 EN. Luxembourg. Publications Office of the European Union.

[41] Ibid.

All but seven studies assess global warming impacts, making it the most frequently analysed midpoint category, followed by eutrophication (91 studies) and acidification (89 studies). Photochemical ozone formation and ionising radiation are more frequently studied in the period 2011-2015, whereas human toxicity and ecotoxicity increasingly come into focus in the period 2016-2021 (see Figure 17). 31 out of 35 studies using flow- or stock related indicators deal with energy demand, while among the endpoint indicators human health, ecosystems and resources are analysed to a similar extent (15, 13 and 12 studies respectively). Only one study[42] computes a single, overall endpoint indicator value. There are no large differences between studies of different context, except for a more frequent analysis of toxicity impacts in studies with a waste management context (37 out of 69 studies include human toxicity and 30 ecotoxicity compared to eight and 10 out of 36 studies with an agricultural context and six and five out of 18 studies with an AD context). On the other hand, geographical background seems to influence the choice of impact categories. For instance, eight out of 13 North American studies, but only 18 out of 64 European and four out of 36 Asian ones use energy related impact categories.

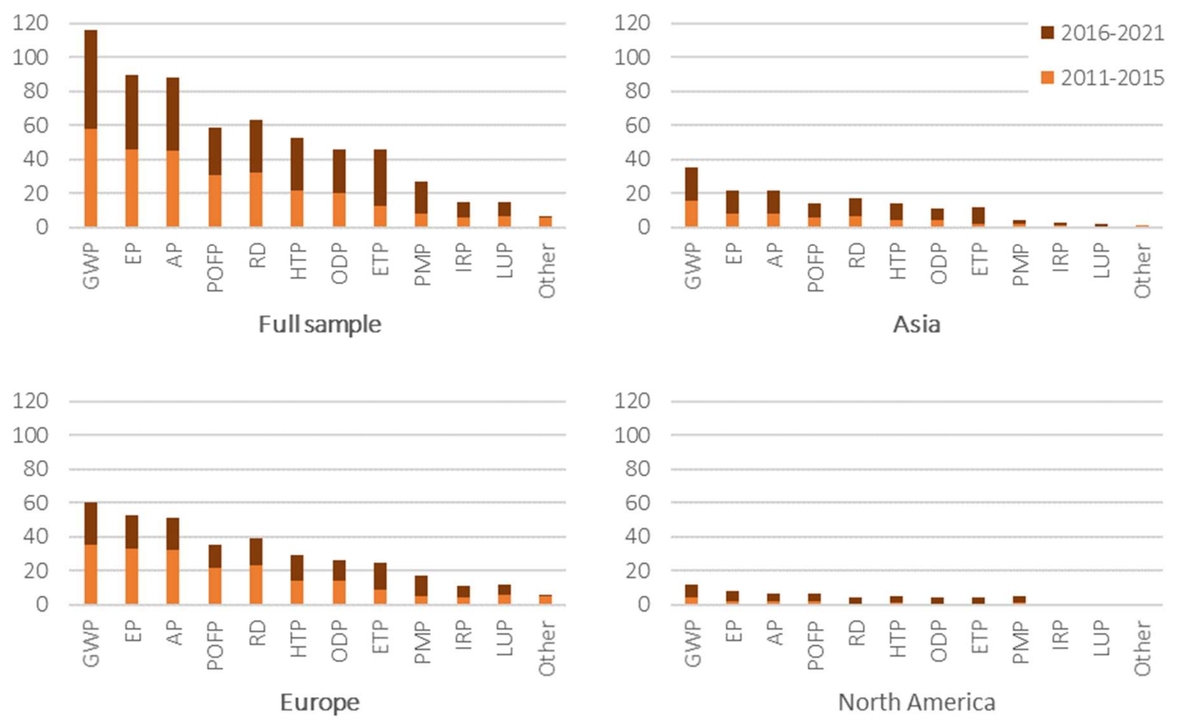

Figure 17: Impact categories used by the studies in the sample including distribution among geographical background and development over time. GWP: global warming potential, EP: eutrophication potential, AP: acidification potential, POFP: photochemical ozone formation potential, RD: resource depletion potential, HTP: human toxicity potential, ODP: ozone depletion potential, ETP: ecotoxicity potential, PMP: particulate matter potential, IRP: ionising radiation potential, LUP: land use potential.

There are multiple methods to translate input and output flows of a system into environmental impacts, some focussing on a single indicator (e.g., the IPCC on global warming[43], the ecoinvent

[42] Mehr and Hellweg 2018 (Full reference see Annex I)

[43] IPCC. Eggleston, H.S.; Buendia, L.; Miwa, K.; Ngara, T. and Tanabe, K. (editors), 2006. *IPCC guidelines for national greenhouse gas inventories*. Japan. IGES.

guidelines for cumulative energy demand (CED)[44], USEtox on human- and ecotoxicity[45]), others providing assessment methods for a multitude of impact categories (e.g. CML[46], ReCiPe[47], ILCD[48], TRACI[49], EDIP[50]).

Of the 123 studies in the sample 19 do not specify the method used for at least part of the impact categories they apply. Of the remainder around a quarter (41 studies) uses the CML framework, followed by ReCiPe (30 studies) and the IPCC approach (21 studies). However, in 12 studies the applied impact assessment method varies with the impact category analysed, while 29 studies compare different assessment methods with each other. In the period 2016-2021 ReCiPe overtakes CML as the most popular impact assessment method in the sample (see Figure 18). Furthermore, the method recommended by ILCD, which is not applied in any of the studies in the period 2011-2015 has quickly gained popularity and is used in 10 out of 59 studies from the period 2016-2021. As for impact categories, the choice of impact assessment method is highly dependent on the geographical setting. While methods developed and frequently applied in Europe (especially ReCiPe and CML) are also often applied in the Asian studies in the sample, almost half of the studies (six out of 13) with a North American background use the US developed TRACI-method. However, also differences between different context categories can be observed. The variety of methods applied is comparatively small for studies with an AD context, where CML and ReCiPe methods are dominating in the period 2011-2015 (used in nine out of 12 studies) while half of the studies from the period 2016-2021 (three out of six) only analyse global warming impacts (with IPCC method). However, the share of studies not specifying the characterisation method used is also higher than for other study types (four out of 18 studies, compared to six out of 36 studies with an agricultural and nine out of 69 with a waste management context). Half of the studies using the ILCD method are conducted in an agricultural context, whereas the increase in the use of ReCiPe method in the period 2016-2021 is mainly associated with a waste management context.

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[44] Hischier, R.; Weidema, B.; Althaus, H.-J.; Bauer, C.; Doka, G.; Dones, R.; Frischknecht, R.; Hellweg, S.; Humbert, S.; Jungbluth, N.; Köllner, T.; Loerincik, Y.; Margni, M. and Nemecek, T., 2010. *Implementation of Life Cycle Impact Assessment Methods*. ecoinvent report No. 3, v2.2. Dübendorf. Swiss Centre for Life Cycle Inventories.

[45] Rosenbaum, R.K.; Bachmann, T.M.; Gold, L.S.; Huijbregts, M.A.J.; Jolliet, O.; Juraske, R.; Koehler, A.; Larsen, H.F.; MacLeod, M.; Margni, M.D.; McKone, T.E.; Payet, J.; Schuhmacher, M.; van de Meent, D. and Hauschild, M.Z., 2008. *USEtox – The UNEP-SETAC toxicity model: Recommended characterisation factors for human toxicity and freshwater ecotoxicity in life cycle impact assessment*. The International Journal of Life Cycle Assessment 13, 532-546. https://doi.org/10.1007/s11367-008-0038-4

[46] Guinée, J.B.; Gorrée, M.; Heijungs, R.; Huppes, G.; Kleijn,R.; Koning, A. de; Oers, L. van; Wegener Sleeswijk, A.; Suh, S.; Udo de Haes, H.A.; Bruijn, H. de; Duin, R. van and Huijbregts, M.A.J., 2002. *Handbook on life cycle assessment. Operational guide to the ISO standards. I: LCA in perspective. IIa: Guide. IIb: Operational annex. III: Scientific background*. ISBN 1-4020-0228-9. Dordrecht. Kluwer Academic Publishers.

[47] Goedkoop, M.; Heijungs, R.; Huijbregts, M.; Schryver, A.D.; Struijs, J. and van Zelm, R., 2009. ReCiPe 2008. *A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level*. First edition. Den Haag. Ministerie van Volkshuisvesting.

[48] European Commission -Joint Research Centre -Institute for Environment and Sustainability, 2010. *International Reference Life Cycle Data System (ILCD) Handbook. General Guide for Life Cycle Assessment. Detailed Guidance*. First edition March 2010. EUR 24708 EN. Luxembourg. Publications Office of the European Union.

[49] Ryberg, M., Vieira, M.D.M., Zgola, M., Bare, J. and Rosenbaum, R.K., 2013. *Updated US and Canadian normalization factors for TRACI 2.1. Clean Technologies and Environmental Policy*. ISSN: 1618-954X. http://dx.doi.org/10.1007/s10098-013-0629-z.

[50] Hauschild, M. And Potting, J., 2005. *Spatial differentiation in Life Cycle impact assessment –The EDIP2003 methodology*. Environmental News No. 80, 2005. Institute for Product Development. Technical University of Denmark.

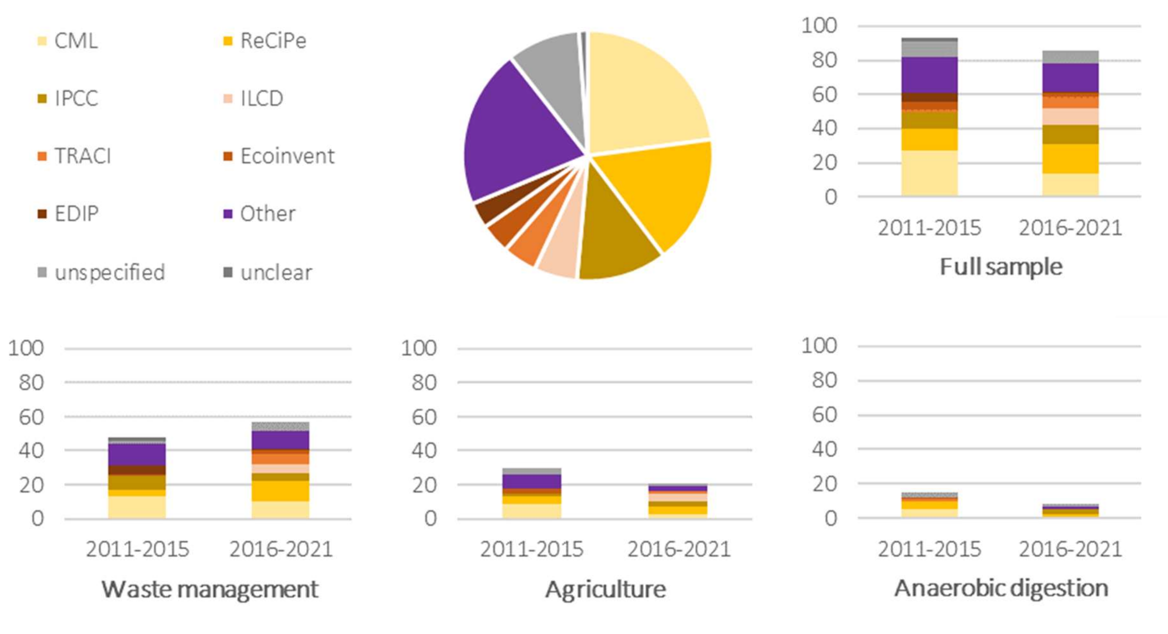

Figure 18: Impact assessment methods used by the studies in the sample including distribution among context categories and geographical background, as well as development over time. For descriptions of the impact assessment methods see references in the main text.

The choice of impact categories and of assessment method are partly interdependent as not all impacts are included in all methods. For instance, the CML approach does not contain provisions for the assessment of impacts associated with particular matter, while with the TRACI method, impacts due to land use changes cannot be analysed. However, studies frequently do not use the whole set of impact categories contained in an assessment method. While all but one study using the CML method apply it to eutrophication and acidification[51], this is only the case for 60-70% of studies using the ReCiPe and ILCD approach. Similarly, resource depletion is more frequently analysed in studies using the CML method (61% compared to 53% for ReCiPe and 50% for ILCD), whereas studies using the ReCiPe and ILCD approach more frequently include toxicity impacts (50% each for both human and ecotoxicity, compared to 44% and 27% for ecotoxicity, respectively for the CML approach).

3.8.3. LCA software

83 of the 123 studies in the sample make use of a LCA software when performing the impact assessment. As there are slight differences in the way how processes can be modelled in the software and in the databases and impact assessment methods included, the choice of software can be relevant. Three types of software are clearly dominant in the sample: SimaPro (applied in 34 studies), GaBi (24 studies) and in the period 2016-2021 also EASEWASTE and its follow-up product EASETECH (10 studies). This pattern can be observed across all study context and geographical backgrounds, with a slightly higher dominance of SimaPro in European studies (21 out of 47 studies reporting the use of a LCA software). EASETECH/EASEWASTE, which was developed for LCAs on waste management, is also most frequently applied in this context (seven out of 10 studies reporting the use of EASETECH/EASEWASTE). However, in the period 2016-2021 it has also been used in three studies with

[51] El Hannandeh et al. 2015 (full reference see Annex I)

an agricultural context[52,53,54]. Overall, reporting of the use of a LCA software increases between the periods 2011-2015 and 2016-2021 (from 63% to 73% of all studies), most pronouncedly in Asian studies with a waste management context (from 42% to 92%).

3.9. Optional elements and interpretation

Optional LCA elements comprise normalisation, i.e., referring the results achieved for an impact to reference information, as well as grouping and weighting, where different impact categories are ranked, and results are converted and possibly aggregated into a single value. While there is no requirement for these elements, the ISO standard 14040/14044 contains guidelines on how these steps should be conducted. Whenever different LCA scenarios are compared to each other, impact assessment results should not be weighted. In spite of this, weighting is used in 12 of the 123 studies in the sample (all of them conducting comparative LCAs and mostly from the period 2011-2015). Of these, four studies do not provide unmodified results and in only three studies[55,56,57] assumptions on weighting factors are subjected to sensitivity analysis. Normalisation is applied in 41 studies, 19 of which not reporting unmodified results. Normalisation tends to be more frequently applied in studies with a waste management context.

Finally, the ISO standard 14040/44 contains provisions to point out any limitations associated with the interpretation of results and to emphasise the relativity of LCA results (i.e., stating that they do not predict on category endpoints, the exceeding of thresholds, safety margins or risks). While 105 studies comply with the former requirement, merely seven studies include a statement on the relativity of LCA results.

4. DISCUSSION AND CONCLUSIONS

In line with previous assessments on similar topics[58,59,60] the present analysis revealed high discrepancies in the way how BBFs are treated in LCA.

While compliance with basic requirements of the ISO standard 14040/14044, such as proper definition of the goal, FU and system boundaries, as well as reasonable choice of impact categories and impact assessment methods is generally high, especially in the most recent period, verification of data quality or assumptions on cut-off criteria and weighting factors (if applied) is often lacking.

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[52] Ten Hoeve et al. 2016a (full reference see Annex I)
[53] Ten Hoeve et al. 2016b (full reference see Annex I)
[54] Ten Hoeve et al. 2019 (full reference see Annex I)
[55] Sparrevik et al. 2013 (full reference see Annex I)
[56] Vázquez-Rowe et al. 2015 (full reference see Annex I)
[57] Di Maria and Micale 2015b (full reference see Annex I)
[58] Maier, M.S.; Stoessel, F.; Jungbluth, N.; Juraske, R.; Schader, C. and Stolze, M., 2015. *Environmental impacts of organic and conventional agricultural products – Are the differences captured by life cycle assessment?* Journal of Environmental Management, 149, 193-208. http://dx.doi.org/10.1016/j.jenvman.2014.10.006
[59] Notarnicola, B.; Sala, S.; Anton, A.; McLaren, S.J.; Saouter, E. and Sonesson, U., 2017. *The role of life cycle assessment in supporting sustainable agri-food systems: A review of the challenges*. Journal of Cleaner Production, 140, 399-409. http://dx.doi.org/10.1016/j.jclepro.2016.06.071
[60] Brockmann, D.; Pradel, M. And Hélias, A., 2018. *Agricultural use of organic residues in life cycle assessment: Current practices and proposal for the computation of field emissions and of the nitrogen mineral fertilizer equivalent*. Resources, Conservation & Recycling, 133, 50-62. https://doi.org/10.1016/j.resconrec.2018.01.034

However, the largest barrier to making LCAs of BBFs comparable currently seems to be the insufficient consideration of fertiliser efficiencies and emissions in the field, often combined with a lack of reporting. As already observed by Brockmann et al. (2018)[61] and in addition to missing information on how emissions are calculated also many studies in the present sample use national methods to calculate emissions from BBFs, making comparability difficult. As a general framework for all kinds of LCA applications, the ISO standard 14040/14044 alone cannot solve this issue. Several attempts have been made in recent years, especially in a European context, to increase understanding of the problem by quantifying the effects of different assumptions on fertiliser equivalencies on LCA results[62,63,64] or to develop models of nutrient behaviour in the field[65,66,67]. Yet often these models are only applicable in a limited geographical and climatic region and to date none of them has reached widespread popularity. The recently published JRC's suggestions for updating the Product Environmental Footprint method[68] contains recommendations for the treatment of multifunctionalities and modelling of emissions in an agricultural context, including fertiliser application. However, as for the ISO standard, more detailed specifications on how to apply these rules to BBFs are likely necessary to obtain comparable LCA results.

It should be noted that comparing fertiliser efficiencies is not a straightforward issue. As noted by many studies in the present sample, the amount of nutrients taken up by plants does not only depend on the type of fertilising products, but also on application techniques, climatic and soil conditions and (in case of P) the previous fertilising history. These factors are often out of the control of the BBF manufacturer. It is thus justifiable to consider BBFs as intermediate products and end the assessment at the industrial gate, i.e., excluding all impacts occurring in the agricultural system. Nevertheless, for a fair comparison between different types of MF and BBFs, differences in nutrient use efficiency and emissions during and after field application have to be taken into account.

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[61] Ibid.
[62] Hanserud et al. 2018 (full reference see Annex I)
[63] Yoshida et al. 2018 (full reference see Annex I)
[64] Ten Hoeve et al. 2018 (full reference see Annex I)
[65] Bruun, S.; Yoshida, H.; Nielsen, M.P.; Jensen, L.S.; Christensen, T.H. and Scheutz, C., 2016. *Estimation of long-term environmental inventory factors associated with land application of sewage sludge*. Journal of Cleaner Production, 126, 440-450. http://dx.doi.org/10.1016/j.jclepro.2016.03.081
[66] Brockmann, D.; Pradel, M. And Hélias, A., 2018. *Agricultural use of organic residues in life cycle assessment: Current practices and proposal for the computation of field emissions and of the nitrogen mineral fertilizer equivalent*. Resources, Conservation & Recycling, 133, 50-62. https://doi.org/10.1016/j.resconrec.2018.01.034
[67] Nielsen, M.P; Yoshida, H.; Raji, S.G.; Scheutz, C.; Jensen, L.S.; Christensen, T.H. and Bruun, S., 2019. *Deriving environmental life cycle inventory factors for land application of garden waste products under northern European conditions*. Environmental Modeling & Assessment, 24, 21-35. https://doi.org/10.1007/s10666-018-9591-9
[68] Zampori, L. and Pant, R., 2019. *Suggestions for updating the Product Environmental Footprint (PEF) method. JRC Technical Reports*. EUR 29682 EN. Luxembourg. Publications Office of the European Union.

ANNEX I – LCA studies included in the analysis

ALI, M.; Wang, W. and Chaudhry, N., 2016. *Application of life cycle assessment for hospital solid waste management: A case study*. Journal of the air & waste management association, 66 (10), 1012-1018. http://dx.doi.org/10.1080/10962247.2016.1196263

AMANN, A.; Zoboli, O.; Krampe, J.; Rechberger, H.; Zessner, M. and Egle, L., 2018. *Environmental impacts of phosphorus recovery from municipal wastewater*. Resources, Conservation & Recycling, 130, 127-139. https://doi.org/10.1016/j.resconrec.2017.11.002

ANDERSEN, J.K.; Boldrin, A.; Christensen, T.H. and Scheutz, C., 2011. *Mass balances and life cycle inventory of home composting of organic waste*. Waste Management 31, 1934-1942. http://doi.org/10.1016/j.wasman.2011.05.004

ANKATHI, S.K.; Potter, J.S. and Shonnard, D.R., 2018. *Carbon Footprint and Energy Analysis of Bio-CH4 from a Mixture of Food Waste and Dairy Manure in Denver, Colorado*. Environmental Progress & Sustainable Energy, 37 (3), 1101-1111. http://doi.org/10.1002/ep

ANTONOPOULOS, I.-S.; Karagiannidis, A.; Tsatsarelis, T. and Perkoulidis, G., 2013. *Applying waste management scenarios in the Peloponnese region in Greece: a critical analysis in the frame of life cycle assessment*. Environmental Science and Pollution Research, 20, 2499-2511. https://doi.org/10.1007/s11356-012-1139-y

AZIZ, R.; Chevakidagarn, P. and Danteravanich, S., 2015. *Environmental Impact Evaluation of Community Composting by Using Life Cycle Assessment: A Case Study Based on Types of Compost Product Operations*. Walailak Journal of Science and Technology, 13 (3), 221-233. http://doi.org/10.14456/wjst.2016.21

AZIZ, R.; Chevakidagarn, P. and Danteravanich, S., 2018. *Composting system improvement by life cycle assessment approach on community composting of agricultural and agro industrial wastes*. Indonesian Journal of Environmental Management and Sustainability, 3, 69-75. https://doi.org/10.26554/ijems.2018.3.3.69-75

BARTZAS, G.; Zaharaki, D. and Komnitsas, K., 2015. *Life cycle assessment of open field and greenhouse cultivation of lettuce and barley*. Information Processing in Agriculture 2, 191-207. http://dx.doi.org/10.1016/j.inpa.2015.10.001

BATUECAS, E.; Tommasi, T.; Battista, F.; Negro, V.; Sonetti, G.; Viotti, P.; Fino, D. and Mancini, G., 2019. *Life Cycle Assessment of waste disposal from olive oil production: Anaerobic digestion and conventional disposal on soil*. Journal of Environmental Management, 237, 94-102. https://doi.org/10.1016/j.jenvman.2019.02.021

BECKER JR., A.M.; Yu, K.; Stadler, L.B. and Smith, A.L., 2017, Co-management of domestic wastewater and food waste: A life cycle comparison of alternative food waste diversion strategies. Bioresource Technology, 223, 131-140. http://dx.doi.org/10.1016/j.biortech.2016.10.031

BERNSTAD, A.; La Cour Jansen, J. and Aspegren, H., 2011. *Life cycle assessment of a household solid waste source separation programme: a Swedish case study*. Waste Management & Research, 29 (10), 1027-1042. http://doi.org/10.1177/0734242X11406170

BRUUN, S.; Hansen, T.L.; Christensen, T.H.; Magid, J. and Jensen, L.S., 2006. *Application of processed organic municipal solid waste on agricultural land – a scenario analysis*. Environmental Modelling and Assessment, 11, 251-265. http://doi.org/10.1007/s10666-005-9028-0

CHEN, R.; Rojas-Downing, M.M.; Zhong, Y.; Saffron, C.M. and Liao, W., 2015. *Life Cycle and Economic Assessment of Anaerobic Co-digestion of Dairy Manure and Food Waste*. Industrial Biotechnology, 11 (2), 127-139. http://doi.org/10.1089/ind.2014.0029

CHI, Y.; Dong, J.; Tang, Y.; Huang, Q. and Ni, M., 2015. *Life cycle assessment of municipal solid waste source-separated collection and integrated waste management systems in Hangzhou, China*. Journal of Material Cycles and Waste Management, 17, 695–706. https://doi.org/10.1007/s10163-014-0300-8

CHIEW, Y.L.; Spångberg, J.; Baky, A.; Hansson, P.-A. and Jönsson, H., 2015. *Environmental impact of recycling digested food waste as a fertilizer in agriculture – A case study.* Resources, Conservation and Recycling, 95, 1-14. http://dx.doi.org/10.1016/j.resconrec.2014.11.015

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DE LUCIA, B.; Cristiano, G.; Vecchietti, L.; Rea, E. and Russo, G., 2013. *Nursery Growing Media: Agronomic and Environmental Quality Assessment of Sewage Sludge-Based Compost*. Applied and Environmental Soil Science, 2013, 565139. http://dx.doi.org/10.1155/2013/565139

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