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Review

Life cycle assessment of nutrient recycling from wastewater: A critical review

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ABSTRACT

Recovering resources from wastewater systems is increasingly being emphasised. Many technologies exist or are under development for recycling nutrients such as nitrogen and phosphorus from wastewater to agriculture. Planning and design methodologies are needed to identify and deploy the most sustainable solutions in given contexts. For the environmental sustainability dimension, life cycle assessment (LCA) can be used to assess environmental impact potentials of wastewater-based nutrient recycling alternatives, especially nitrogen and phosphorus recycling. This review aims to evaluate how well the LCA methodology has been adapted and applied for assessing opportunities of wastewater-based nutrient recycling in the form of monomineral, multimineral, nutrient solution and organic solid. We reviewed 65 LCA studies that considered nutrient recycling from wastewater for agricultural land application. We synthesised some of their insights and methodological practices, and discussed the future outlook of using LCA for wastewater-based nutrient recycling. In general, more studies suggested positive environmental outcomes from wastewater-based nutrient recycling, especially when chemical inputs are minimised, and source separation of human excreta is achieved. The review shows the need to improve methodological consistency (e.g., multifunctionality, fertiliser offset accounting, contaminant accounting), ensure transparency of inventory and methods, consider uncertainty in comparative LCA context, integrate up-to-date cross-disciplinary knowledge (e.g., agriculture science, soil science) into LCA models, and consider the localised impacts of recycled nutrient products. Many opportunities exist for applying LCA at various scales to support decisions on wastewater-based nutrient recycling – for instance, performing "product perspective" LCA on recycled nutrient products, integrating "process perspective" LCA with other systems approaches for selecting and optimising individual recovery processes, assessing emerging nutrient recovery technologies and integrated resource recovery systems, and conducting systems analysis at city, national and global level.

Contents

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1. Introduction

"Closing cycles" and reuse of resources from wastewater and sludge are increasingly being emphasised (Peccia and Westerhoff, 2015; Van Loosdrecht and Brdjanovic, 2014). Many technologies exist or are under development for recycling nutrients such as nitrogen and phosphorus from wastewater to agriculture (Harder et al., 2019; Mehta et al., 2015). The challenge of transitioning to resource-efficient urban water systems is often not the availability of technology for resource recovery, but the lack of planning and design methodology to identify and deploy the most sustainable solution in a given context (Guest et al., 2009). Life cycle assessment (LCA) provides a method to assess wastewater-based resource recovery alternatives from environmental sustainability perspectives, and it can be combined with other context-specific assessment techniques such as life cycle costing and risk assessments to evaluate alternatives (Guest et al., 2009).

LCA is a framework to quantify environmental impact potentials (benefits or burdens) of processes and products throughout their life cycle. In some cases, it provides insights of potential trade-offs between different environmental impacts (Hellweg and Canals, 2014) and/or economic performance when integrating with life cycle costing (Zijp et al., 2017). In the field of urban water management, LCA has become a popular approach to assess environmental performance of operating and upgrading water supply systems and wastewater treatment systems. In particular, some of these LCA studies aim to evaluate different nutrient recovery strategies (Sena and Hicks, 2018) and sludge management approaches (Pradel et al., 2016). In the future, nutrient recovery processes have to be sustainable by minimising process inputs such as water, energy and chemicals (Mehta et al., 2015), avoiding "side-waste" and maximising recovery efficiency. LCA provides a framework to consider both direct process inputs/emissions and indirect life cycle inputs/emissions to guide process development toward the goal of minimising resource inputs and wastes.

A number of recent reviews has examined LCA studies of wastewater-based nutrient recycling from various perspectives. Sena and Hicks (2018) focused on LCA of struvite precipitation from wastewater treatment and analysed literature based on major LCA steps. Heimersson et al. (2016) reviewed life cycle inventory practices for major nutrient flows in wastewater and sludge management systems. Pradel et al. (2016) explored the methodological implications for a shift of considering wastewater sludge from waste to valued products. In addition to these reviews on nutrient recycling LCA, a number of other reviews (Corominas et al., 2013; Gallego-Schmid and Tarpani, 2019; Yoshida et al., 2013) has provided comprehensive syntheses for a rapidly growing body of LCA literature on wastewater treatment and sewage sludge management.

With wastewater treatment facilities transitioning into water resource recovery facilities, LCA has to increasingly account for the functionality of resource recovery. The objective of this study is therefore to evaluate how well the LCA methodology has been adapted and applied for assessing opportunities of wastewater-based nutrient recycling. This study reviewed LCA studies that considered nutrient recycling from wastewater for agricultural land application. We synthesise and analyse their general insights (Section 4) and methodological practices (Section 5), and provide a future outlook of using LCA to assess wastewater-based nutrient recycling (Section 6). These aspects can complement with existing reviews on wastewater-based nutrient recycling LCA (Pradel et al., 2016; Sena and Hicks, 2018) and on technical aspects of nutrient recycling (Harder et al., 2019; Mehta et al., 2015). Collectively, this could help guide the use of LCA to provide quality environmental information for decision making on the planning and implementation of wastewater-based nutrient recycling.

2. Methods

2.1. Definition

In this study, "nutrient recycling" is referred to as the overall process of reusing nutrients (e.g., nitrogen, phosphorus) from waste material (i.e., urine, faeces, wastewater, sewage sludge) for agriculture through different nutrient recycling pathways. Fig. 1 shows a simplified illustration of the nutrient recycling pathways assessed by the reviewed studies (see Harder et al. (2019) for a comprehensive overview of pathways). "Nutrient recovery" is the process of extracting and/or concentrating nutrients from waste material. The recycled nutrient products are categorised as – recycled water, urine, nutrient-rich stream, biomass, undigested sludge, composted sludge, anaerobically-digested sludge, biochar, ash, struvite, phosphorus product (e.g., calcium phosphate, diammonium phosphate) and nitrogen product (e.g., ammonium sulphate).

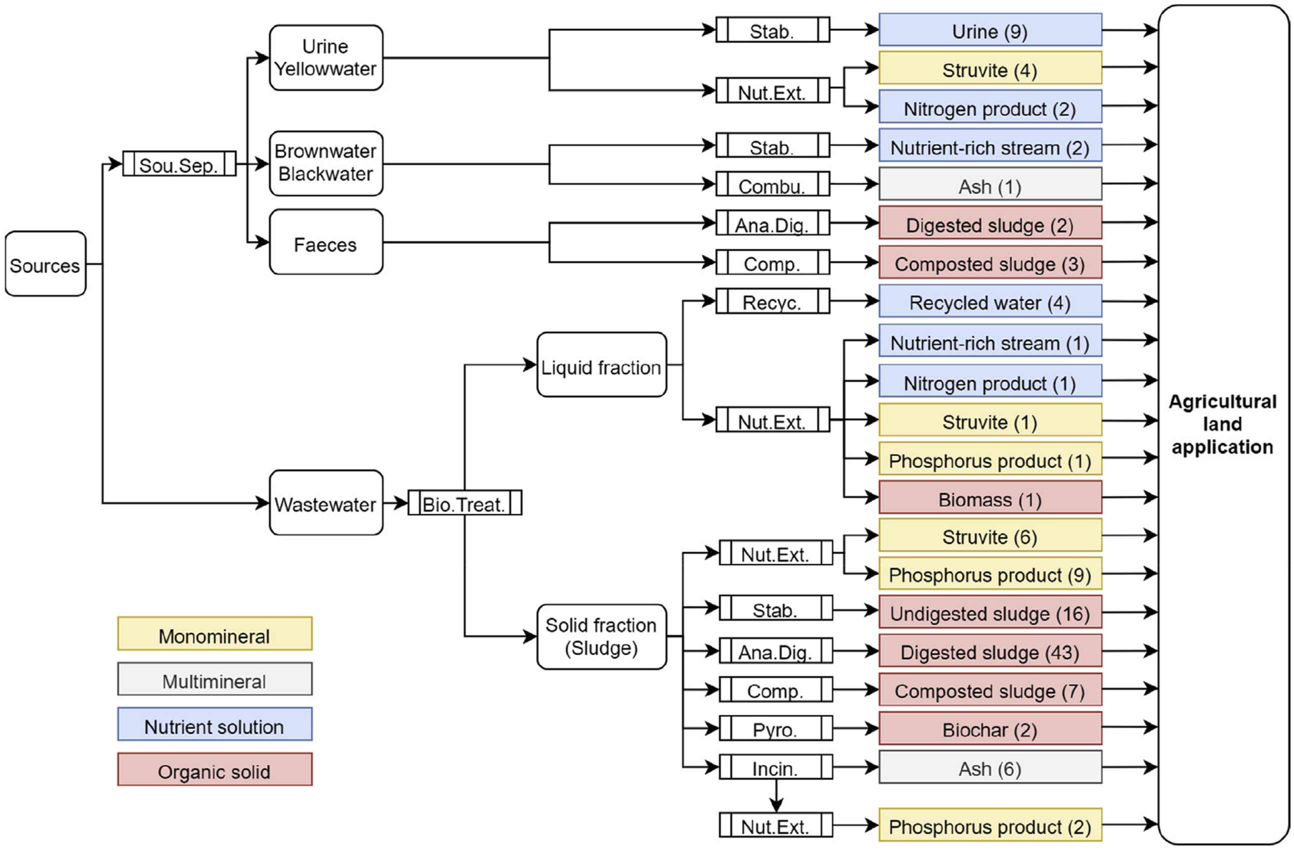

Fig. 1. Simplified wastewater-based nutrient recycling pathways. Each recycled nutrient product box shows the number of reviewed studies assessing that pathway. (Sou. Sep.: source separation; Bio. Treat.: biological treatment; Stab.: stabilisation; Nut. Ext.: nutrient extraction; Combu.: combustion; Ana. Dig.: anaerobic digestion; Comp.: composting; Recyc.: water recycling treatment; Pyro.: pyrolysis; Incin.: incineration).

2.2. LCA methods

Process-based LCA and economic input-output analysis based LCA are two primary approaches to LCA. This work only reviewed process-based LCA studies. The ISO standards for LCA (ISO 14040 and ISO 14044) are primarily concerned with the process-based approach. Following the ISO standards, the LCA framework has four phases – Goal and Scope Definition, Inventory Analysis, Impact Assessment and Interpretation. In the Goal and Scope Definition phase, the goal of the study, functional unit, system and system boundary are defined. In the Inventory Analysis phase, the physical flows of materials and energy into and out of the defined system are quantified. In the Impact Assessment phase, the physical flows quantified are translated into environmental impact indicators (e.g., global warming potential, eutrophication potential, acidification potential, ecotoxicity, human toxicity, photochemical ozone formation) based on chosen impact assessment models (e.g. CML, ReCiPe, TRACI (Hischier et al., 2010)). In the Interpretation phase, the results from the Inventory Analysis phase and the Impact Assessment phase are interpreted based on the goal defined.

2.3. Review questions

The analysis and discussion of this review is revolved around the following questions, with the primary objective of evaluating how well the LCA methodology has been adapted and applied for assessing opportunities of wastewater-based nutrient recycling.

- What are the general results from i) assessing agricultural land application of sewage sludge (Section 4.1), ii) assessing agricultural land application of other recycled nutrient products (Section 4.2), iii) comparing different recycled nutrient products (Section 4.3), and iv) assessing the impacts of source separation (Section 4.4)?
- How have process or product perspective (Section 5.1), and multifunctionality (Section 5.2) been handled?
- How have fertiliser offsets (Section 5.3) and carbon sequestration (Section 5.4) been accounted for?
- How have heavy metals, organic pollutants and pathogenic risks been accounted for (Section 5.5)?
- What are the practices of uncertainty analysis and sensitivity analysis (Section 5.6)?
- What are the methodological challenges and future opportunities of using LCA for assessing wastewater-based nutrient recycling (Section 6)?

2.4. Selection of the reviewed papers

This work targeted LCA studies that considered nutrient recycling from wastewater for agricultural land application. It includes only studies using process-based LCA method and excludes studies using purely economic input-output analysis LCA method.

A literature search was conducted to find nutrient recycling related studies (peer-reviewed original papers) using the Scopus database. In the first stage, different combinations of keywords including nutrient recycling, nutrient recovery, resource recovery, life cycle assessment, LCA, wastewater and water were used to identify applicable studies (see Supplementary Material for the combination of keywords). In the second stage, the reference lists of the reviewed

Table 1
Summary of nutrient recycling related LCA studies (more details are provided in the Supplementary Material).

Study	Type of functional unit	System boundary [a]	System expansion (fertiliser production)	Specifying nutrients recycled	Recycled nutrient product [b] (* source-separated recycling)											
					Organic solid					Nutrient solution				Monomineral		Multimineral
					BM	US	CS	DS	BC	RW	UR	NR	NP	MAP	PP	AS
Bridle and Skrypski-Mantele (2000)	Sludge	ST-AG	–	N, P, K		Y		Y								
Lundin et al. (2000)	Wastewater	SS-WT-ST-AG	Y	N, P		Y					Y*					
Suh and Rousseaux (2002)	Sludge	ST-AG	–	–		Y	Y	Y								
Maurer et al. (2003)	Nitrogen	SS-AG	–	N							Y*					
Lundin et al. (2004)	Sludge	ST-AG	Y	N, P				Y							Y	
Hospido et al. (2005)	Sludge	ST-AG	Y	N, P				Y								
Houillon and Jolliet (2005)	Sludge	ST-AG	Y	N, P, K		Y										
Svanström et al. (2005)	Sludge	ST-AG	Y	N, P				Y							Y	
Johansson et al. (2008)	Sludge	ST-AG	Y	N, P				Y							Y	
Murray et al. (2008)	Sludge	ST-AG	Y	N, P		Y	Y	Y								
Remy and Jekel (2008)	Wastewater	SS-WT-AG	Y	N, P, K		Y	Y	Y								Y
Hong et al. (2009)	Sludge	ST-AG	–	–			Y*	Y/Y*			Y*					
Pasqualino et al. (2009)	Wastewater	WT-ST-AG	Y	–			Y	Y								
Peters and Rowley (2009)	Sludge	ST-AG	Y	N, P		Y		Y								
Hospido et al. (2010)	Sludge	ST-AG	Y	N, P				Y								
Lederer and Rechberger (2010)	Sludge	ST-AG	Y	N, P				Y							Y	Y
Meneses et al. (2010)	Recycled water	WT-ST	Y	N, P						Y						
Sablayrolles et al. (2010)	Sludge	ST-AG	Y	N, P		Y	Y									
Carballa et al. (2011)	Sludge	ST-AG	Y	N, P				Y								
Linderholm et al. (2012)	Phosphorus	ST-AG	Y	N, P				Y						Y	Y	
Chen and Chen (2013)	Wastewater	WT-ST	Y	–				Y				Y				
Niu et al. (2013)	Sludge	ST-AG	Y	N, P				Y								
Dong et al. (2014)	Sludge	ST-AG	Y	N, P			Y									
Heimersson et al. (2014)	Wastewater	WT-ST-AG	Y	N, P				Y								
Mills et al. (2014)	Sludge	ST-AG	Y	N, P				Y								
Niero et al. (2014)	Wastewater	WT-ST-AG	Y	P				Y								
Rodriguez-Garcia et al. (2014)	Phosphorus	WT-ST-AG	Y	N, P				Y						Y		
Spångberg et al. (2014)	Nitrogen and phosphorus	SS-WT-AG	Y	N, P							Y*	Y*				
Xu et al. (2014)	Sludge	ST-AG	–	–		Y		Y								
Alanya et al. (2015)	Sludge	ST-AG	Y	N, P		Y		Y								
Bisinella de Faria et al. (2015)	Wastewater	SS-WT-ST-AG	Y	N, P				Y				Y*		Y*		
Bradford-Hartke et al. (2015)	Phosphorus	SS-WT-ST-AG	Y	N, P, K				Y			Y*			Y		
Gianico et al. (2015)	Wastewater	WT-ST-AG	Y	–				Y								
Ishii and Boyer (2015)	Urine	SS-WT	Y	N, P										Y*		
Kalmykova et al. (2015)	Phosphorus	ST-AG	Y	N, P				Y								
Lam et al. (2015)	Wastewater	SS-WT-ST-AG	Y	N, P		Y		Y		Y	Y*					
Meneses et al. (2015)	Wastewater	WT-ST-AG	Y	N, P				Y								
Miller-Robbie et al. (2015)	Sludge	ST-AG	Y	N				Y	Y							
Sørensen et al. (2015)	Sludge	ST-AG	Y	P											Y	
Blanco et al. (2016)	Wastewater	WT-ST-AG	Y	N, P				Y								
Fang et al. (2016)	Wastewater	WT-ST-AG	Y	N, P						Y						
Landry and Boyer (2016)	Urine	SS-WT-AG	Y	N, P										Y*		
Lin et al. (2016)	Nitrogen	WT	Y	N									Y			
Remy et al. (2016)	–	SS-WT	Y	P										Y	Y	
Gourdet et al. (2017)	Sludge	SS-AG	Y	N, P, K				Y								
Heimersson et al. (2017)	Sludge	WT-AG	Y	N, P				Y								
Igos et al. (2017)	Wastewater	SS-WT-ST	Y	N, P									Y*	Y*		
Kavvada et al. (2017)	Urine	SS	Y	N									Y*			
Kulak et al. (2017)	–	SS-WT-ST	Y	N, P		Y	Y*	Y*			Y*					
Lombardi et al. (2017)	Sludge	ST-AG	Y	N, P, K		Y	Y									
Shiu et al. (2017)	Wastewater	WT-ST-AG	Y	N, P		Y				Y						
Svanström et al. (2017)	Sludge	ST-AG	Y	N, P				Y								Y
Willén et al. (2017)	Sludge	AG	Y	N, P				Y								
Amann et al. (2018)	Phosphorus	WT-ST	Y	P										Y	Y	Y
Anastasopoulou et al. (2018)	Wastewater	SS-WT-ST	Y	N, P, K		Y	Y*				Y*					Y*
Arashiro et al. (2018)	Wastewater	WT-ST-AG	Y	N, P	Y											
Cartes et al. (2018)	Sludge	WT-ST-AG	Y	N, P				Y								
Do Amaral et al. (2018)	Wastewater	ST-AG	Y	N, P				Y								Y
Li and Feng (2018)	Sludge	ST-AG	–	–				Y	Y							
Piippo et al. (2018)	Sludge	ST-AG	Y	N, P				Y								

Shi et al. (2018)	Wastewater	SS-WT-ST	Y	N, P, K			Y*			
ten Hoeve et al. (2018)	Sludge	ST-AG	Y	P		Y				Y
Yoshida et al. (2018)	Sludge	ST-AG	Y	N, P, K	Y	Y				
Awad et al. (2019)	Wastewater	WT-ST-AG	Y	–		Y				
Pradel and Aissani (2019)	Phosphorus	WT-ST-AG	Y	N, P		Y		Y	Y	

[a] SS: source separation and treatment; WT: wastewater treatment; ST: sludge treatment; AG: agricultural land application.
[b] BM: phototrophic biomass; US: undigested sludge; CS: composted sludge; DS: anaerobically digested sludge; BC: biochar; RW: recycled water; UR: urine; NR: nutrient-rich stream; NP: nitrogen product; MAP: struvite; PP: phosphorus product; AS: ash.

studies in the first stage and the citations on these studies were checked to identify additional nutrient recycling related LCA studies. This process was repeated on the newly found studies. The following criteria were imposed in selecting the relevant LCA studies to address the review questions set out in Section 2.3.

i) Comparing systems with nutrient recycling against systems without nutrient recycling, or
ii) Comparing systems with different nutrient recycling pathways, or
iii) Assessing or optimising nutrient recovery processes that yield nutrient-rich materials including struvite, phosphorus compounds and ammonium compounds, or
iv) Considering source separation for nutrient recycling, or
v) Focusing primarily on agricultural land application of sewage sludge (with system boundary including sludge treatment and agricultural land application, but not wastewater treatment).

In total, 65 papers (published between 2000 and 2019) were identified. All these LCA studies have considered at least one pathway through which nutrients could be recycled from wastewater to agricultural land (Fig. 1). They also include agricultural land application in their system boundaries and/or account for avoiding mineral or synthetic fertilisers. This review does not include i) wastewater LCA studies that do not consider any nutrient recycling pathway (or that are not clear if they consider), ii) studies that focus on developing life cycle inventory, iii) studies that investigate co-management of sewage sludge and food waste/other organic wastes, and iv) urban water LCA studies that have a strong focus on water supply.

3. Overview of reviewed LCA studies

Table 1 provides a summary of all 65 nutrient recycling related LCA studies included in this review. They have different scopes of nutrient recycling – N recycling only (4), P recycling only (5), N and P recycling (39), and N, P and K recycling (9). Most of these studies (51 out of 65) considered agricultural land application of sewage sludge (composted, anaerobically digested or undigested). This is partly because many of these studies are about sewage sludge management. We analysed these LCA studies from a nutrient recycling perspective. 17 out of 65 studies considered nutrient recovery for specific nutrient products such as struvite, calcium phosphate and ammonium sulphate. A range of recycled nutrient products were studied through different pathways (Fig. 1). The processes of production of some of the recycled nutrient products are not mutually exclusive, but can co-exist (e.g., recycled water, digested sludge, struvite).

In life cycle assessment, a functional unit is used to normalise the results in terms of per unit of service provided by a process or a product. In that way, alternatives can be compared on the same basis. There are three major types of functional unit among these nutrient recovery related LCA studies – amount of wastewater treated (influent/effluent) (19), amount of sludge disposed (31), and amount of nutrient recycled/removed (9). Studies with a functional unit of amount of nutrient recycled/removed had a clear goal of comparing alternatives based on the primary functionality of nutrient recycling. In most of the cases, the functional units chosen were either the amount of wastewater treated, or the amount of sludge disposed. This definition of functional unit reflects that the primary function of these systems is not nutrient recycling. Instead, these systems are multifunctional, with the functions of wastewater treatment, sludge disposal, biogas generation, energy recovery, nutrient recovery and/or other forms of resource recovery.

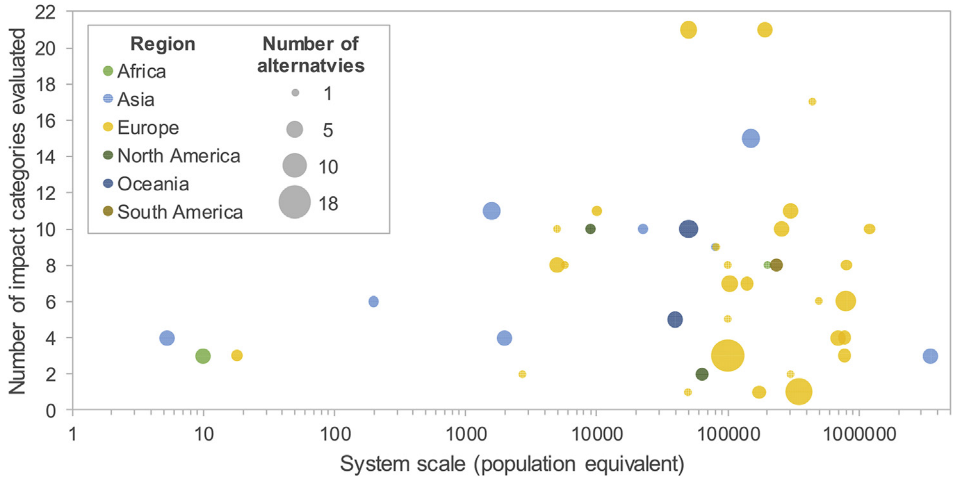

Fig. 2. Scope of 43 nutrient recycling related LCA studies reviewed in terms of i) system scale, ii) number of impact categories assessed, iii) number of nutrient recycling alternatives compared in each study, and iv) studied continents. Each dot is a study, and its size corresponds to the number of alternatives compared in that study (from 1 to 18 nutrient recycling alternatives).

With a multifunctional nature, the environmental impacts resulting from the implementation of any alternative system configuration with nutrient recycling are not solely contributed by nutrient recycling.

Fig. 2 shows 43 of the 65 nutrient recycling related LCA studies in terms of system scale, number of life cycle impact categories assessed, number of nutrient recycling alternatives compared and studied continents. 22 studies are not included in the figure because system scales were not specified. Most of the studies on smaller scale or decentralised systems were conducted in the Asian context, while those studies on the larger scale systems were predominantly conducted in the European context. Amann et al. (2018) compared 18 phosphorus recovery processes in a single study (i.e., the largest dot in Fig. 2). Bisinella de Faria et al. (2015) and Igos et al. (2017) used the full set of impact categories from the ReCiPe method (18 midpoints and 3 endpoints). Among all the nutrient recycling related LCA studies, a range of midpoint life cycle environmental impact categories were assessed, with the most-assessed categories being global warming potential (60), acidification potential (48), eutrophication potential (48), human toxicity (34), ecotoxicity (33) and photochemical ozone formation (32).

4. General insights

Among the nutrient recycling related studies, over half of them feature scenarios to compare the life cycle environmental impact potentials of systems with nutrient recycling to that without recycling. These systems are mostly conventional centralised biological wastewater treatment systems. The absolute results across these studies are often not directly comparable, unless the results are adjusted for their differences in system boundary, system scale, functional unit, impact category, impact assessment method, and reference system or scenario. In general, the results of LCA are unique to the goal and scope of each study, depending on its local context (Yoshida et al., 2013). For comparative LCA (which is the majority type of LCA studies reviewed), the focus is not on predicting absolute impacts, but rather on estimating the relative difference in potential impacts between alternatives (Rosenbaum et al., 2018). In this section, we thus focus much of our discussion on the relative results (i.e., contrasting systems with and without sludge-based nutrient recycling in Section 4.1; contrasting systems with and without non-sludge-based nutrient recycling in Section 4.2; comparing different recycled nutrient products in Section 4.3; comparing nutrient recycling from source-separated human excreta to that of non-source-separated recycling in Section 4.4) instead of the absolute quantitative results (i.e., environmental impact potentials for given systems).

4.1. Agricultural land application of sewage sludge

More studies show that compared to sewage sludge landfilling (without nutrient recycling), application of digested/undigested sewage sludge to agricultural land (with nutrient recycling) could have lower abiotic depletion potential, acidification potential, global warming potential and photochemical ozone formation (Lombardi et al., 2017; Murray et al., 2008; Pasqualino et al., 2009), but could potentially lead to greater ecotoxicity (Peters and Rowley, 2009; Shiu et al., 2017; Suh and Rousseaux, 2002) (Fig. 3(a)). Among the 11 studies that have lower global warming potential in the land application scenario, nine of them show over 50% reduction in the potential. Among the five studies that have higher ecotoxicity in the land application scenario, four of them show that ecotoxicity increases by more than five times. The percentage difference results are included in the Supplementary Material. For the positive environmental impacts of agricultural land application, fertiliser substitution contributes to lower abiotic depletion potential (Lombardi et al., 2017), while lower direct on-site emissions and carbon sequestration help reduce global warming potential (Miller-Robbie et al., 2015). The negative environmental impacts of agricultural land application are the result of excess nutrient and heavy metals present in the sludge (Pasqualino et al., 2009). The impacts on eutrophication and human toxicity are more inconsistent among studies. For studies that show a higher eutrophication potential in landfilling, they considered the emission of ammonia during lime stabilisation on sludge before landfilling (Cartes et al., 2018) and the impact of landfill leachate (Suh and Rousseaux, 2002). It is not evident that those studies with the opposite result have considered such factors (Pasqualino et al., 2009; Shiu et al., 2017), while most studies have included the direct field emissions associated with the spreading of organic matter. Studies have shown that the results for human toxicity are sensitive to the choice of life cycle impact assessment method (Heimersson et al., 2014), while the results for global warming potential, abiotic depletion and acidification potential are more independent of the impact assessment method used in wastewater treatment LCA

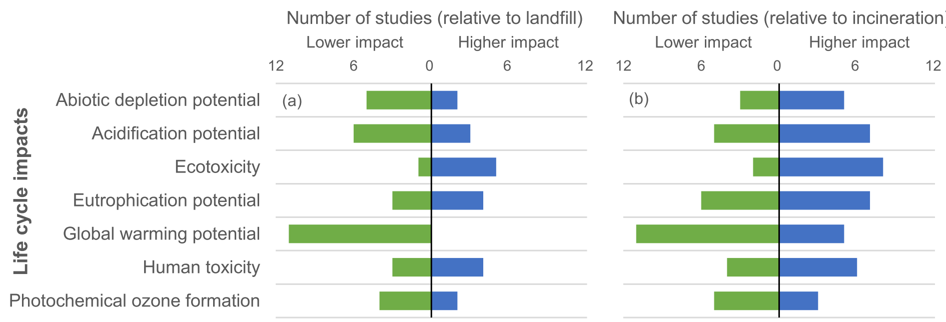

Fig. 3. Number of studies with sludge application scenario showing higher or lower life cycle impacts compared to (a) landfill scenario and (b) incineration scenario.

(Renou et al., 2008).

For comparing agricultural land application of sewage sludge to incineration (and ash landfilling), the results are less inclined to one side (Fig. 3(b)). Some studies clearly favoured agricultural land application over incineration (Chen and Chen, 2013; Niero et al., 2014), while some other studies suggested the opposite (Svanström et al., 2005; Yoshida et al., 2018). Only for global warming potential and photochemical ozone formation, there are more studies showing lower impacts for land application (i.e., 6 out of the 11 lower-impact studies and 4 out of the 5 lower-impact studies show over 50% reduction in global warming potential and photochemical ozone formation respectively.). In general, for most of the other impact categories, more studies suggested that land application has higher impacts than incineration. Similar to comparing land application to landfill scenario (Fig. 3(a)), ecotoxicity and human toxicity are major concerns for land application comparing to incineration scenario. For eutrophication potential, the different results could be attributed to the variability of water and sludge quality parameters (Niero et al., 2014), and whether ammonia emissions and run-off after land applications are being accounted for or not (Lundie et al., 2004; Yoshida et al., 2018). In many cases, studies are often not transparent enough for tracing back the possible causes of discrepancy.

4.2. Agricultural land application of non-sludge recycled nutrient products

In addition to the large number of studies with scenarios on agricultural land application of sewage sludge, eight studies have non-sludge based nutrient recycling scenarios comparing systems with recycling to systems without recycling (e.g., business-as-usual conventional systems). Most studies reported an overall positive environmental performance for their assessed systems, though these studies had a different coverage of impact categories. Meneses et al. (2010) favoured agricultural applications of recycled water over urban applications. Ishii and Boyer (2015) and Landry and Boyer (2016) found positive environmental outcomes from nutrient recycling via struvite precipitation, compared to conventional nutrient removal. Maurer et al. (2003) showed that field application of source-separated urine could lower embodied energy use and global warming potential, compared to conventional denitrification in a wastewater treatment plant. Lin et al. (2016) is one of the two studies that have multiple impact categories showing unfavourable outcomes in nutrient recycling scenarios. Their study assessed the use of zeolite adsorbent on anaerobic digester supernatant for nitrogen recovery. The production of zeolite adsorbent contributes to a large fraction of the environmental burdens.

4.3. Comparison of different recycled nutrient products

A number of studies compared different recycled nutrient products in the same study. The results of the top four most assessed environmental impacts from these studies (global warming potential, acidification potential, eutrophication potential, ecotoxicity) are shown in Fig. 4 (only including studies with non-sludge recycled nutrient products). These studies are divided into three groups based on their functional units (FU): i) the amount of wastewater treated (for the function of waste management), ii) sludge disposed (for the function of waste management), and iii) the amount of nutrient recycled (for the function of resource recovery). For each study, the results were normalised to the maximum absolute value in that study, so that there are relative results for comparison across studies (e.g., across multiple studies, whether a given recycled product has a lower environmental impact than another recycled product). The size of the bubble is proportional to the environmental impact. The larger the filled bubble, the greater is the environmental burden. For example, Linderholm et al. (2012) found that on a per unit of phosphorus basis, recycling through struvite precipitation or digested sludge has much lower greenhouse gas emissions (global warming potential) than recycling through sludge ash. An empty bubble represents a negative value (i.e., greater environmental benefit relative to the other recycled nutrient product(s) in the same study). For instance, Johansson et al. (2008) showed that on a per unit of sludge disposal basis, recycling as phosphorus product can have a higher eutrophication potential than recycling through digested sludge. If there are multiple studies comparing same set of recycled nutrient products, it may give an indication of whether a specific product is still environmental favourable over the others under a range of system configuration, inventory assumptions, methodological choices and geographical constraints.

Although there are limited estimates globally of the proportion of sewage sludge being used for agricultural purpose, available data suggested that agricultural application of sewage sludge is substantial (e.g., 45%, 60% and 55% respectively for Western Europe, Australia and the U.S.) (Mayer et al., 2016), while the practice is subject to strict regulations or even prohibited in some countries (Hudcová et al., 2019). Given that agricultural land application of sewage sludge is still a major practice of nutrient recycling in many countries, it is important to understand what the potential environmental impacts would be for a shift from this practice to other sludge disposal approaches with nutrient recycling such as sludge ash, phosphorus products derived from sludge ash. A number of studies do provide insights to this question.

Five studies examined agricultural land application of incineration ash of sewage sludge, and compared it with application of undigested/digested sludge (See Do Amaral et al. (2018), Lederer

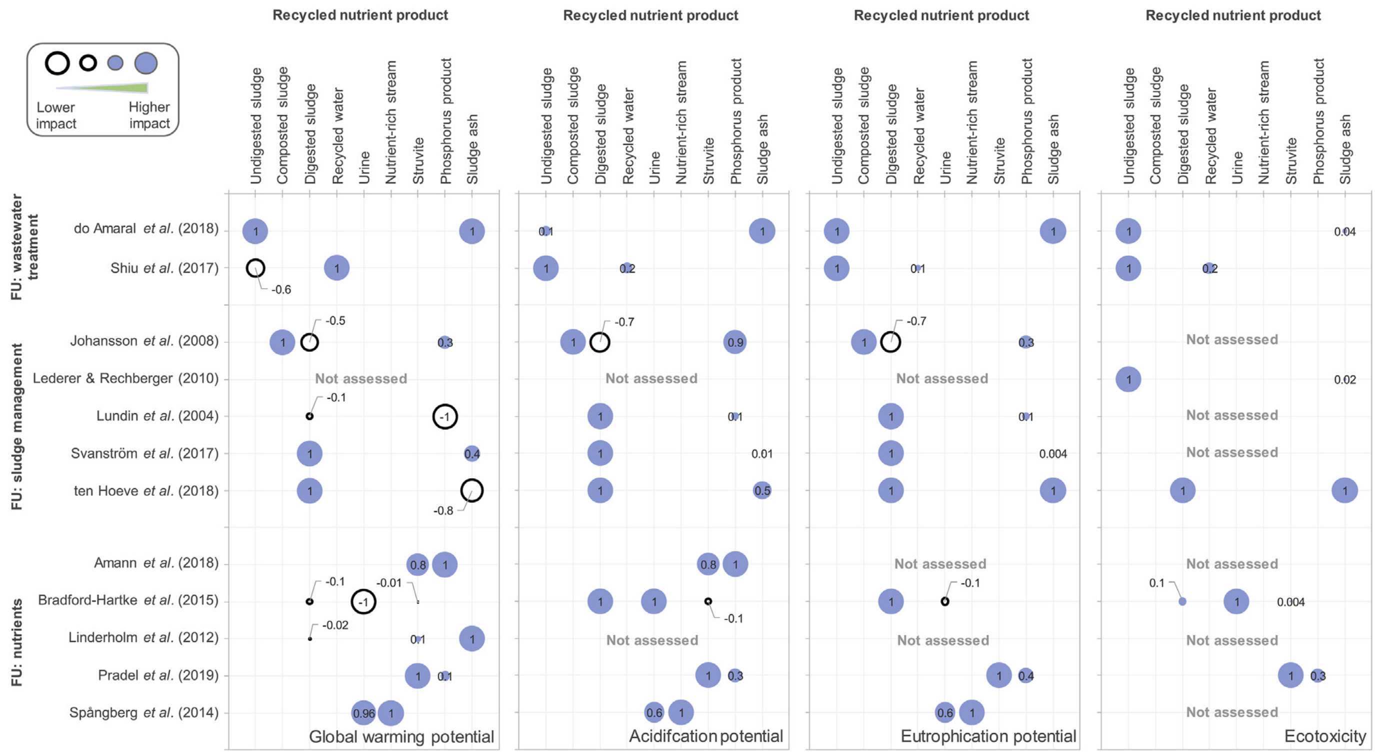

Fig. 4. Comparison of global warming potential, acidification potential, eutrophication potential and ecotoxicity for different recycled nutrient products. In each study, results were normalised for the maximum of the absolute value in that study. The larger the filled bubble (by area), the greater is the environmental burdens. An empty bubble represents a negative value (greater environmental benefits relative to other product(s)). Studies are grouped by functional unit (FU). See Supplementary Material for more details.

and Rechberger (2010), Svanström et al. (2017), ten Hoeve et al. (2018) and Linderholm et al. (2012) in Fig. 4). For the scenarios in Lederer and Rechberger (2010), Linderholm et al. (2012) and Svanström et al. (2017), the sludge ash was treated by a thermal process to remove heavy metals. This has significantly reduced the ecotoxicity of the sludge ash (Lederer and Rechberger, 2010; Svanström et al., 2017), but with a cost of higher energy footprint and global warming potential (Linderholm et al., 2012). In addition to the possibility of removing heavy metals, ash-based recycling approach can also achieve higher recycling rate of phosphorus and removal of organic micropollutants (Amann et al., 2018), but the loss of carbon and nitrogen during incineration could reduce the amount of fertiliser offset (Section 5.3) and eliminate carbon sequestration (Section 5.4).

Two studies compared agricultural land application of sewage sludge to that of phosphorus products derived from sludge streams (See Johansson et al. (2008) and Lundin et al. (2004) in Fig. 4). These phosphorus products include phosphoric acid and ferric phosphate. Both studies used the same type of functional unit of per unit mass of sludge disposed. Lundin et al. (2004) found that for most impact categories (i.e., resource depletion (excluding sulphur), global warming potential, eutrophication potential, acidification potential), recycling through land application of phosphorus products (i.e., phosphoric acid by the Bio-Con method, ferric phosphate by the Cambi-KREPRO method) outperformed that of digested sludge. On the other hand, Johansson et al. (2008) favoured digested sludge over ferric phosphate (by the Aqua Reci method) in terms of global warming potential, eutrophication potential and acidification potential. The different relative results from the two studies can be attributed to different chemical inputs of their processes and different sludge handling approaches prior to agricultural use.

Some phosphorus recovery processes, especially recovery from the digester supernatant, can reduce greenhouse gas emissions and acidification potential (by over 10% in many processes) for sewage sludge management (Amann et al., 2018). The benefits are mostly the results of fertiliser offset and reducing chemicals use downstream at the wastewater treatment plants. In addition, minimising chemical inputs in nutrient recovery processes is an important factor to enhance the environmental performance of recovery processes (Bradford-Hartke et al., 2015; Ishii and Boyer, 2015). Even though chemical additions can enable higher nutrient recovery rate, the additional environmental burdens cannot be fully offset.

4.4. Impact of source separation

Source separation enables the collection of human excreta as concentrated streams for nutrient recycling. Among all the reviewed studies, 14 studies examined systems with source separation for nutrient recycling (See Supplementary Material for studies with a scope of "source separation"). All of these 14 studies had urine separation scenarios, while six of them also considered faeces/black water separation. The two major nutrient recycling pathways from source-separated urine/faeces are through the applications of stabilised urine/faeces (9 out of 14) or recovered phosphorus/ammonium products (5 out of 14).

Despite the differences in functional unit, system size, choice of boundary, geographical area, recycled nutrient product types and configuration of the examined systems, nearly all studies showed that introducing source separation concept for nutrient recycling can lead to a lower global warming potential compared to centralised wastewater treatment with/without nutrient recycling (i.e., over 50% reduction in half of the studies). The GHG emissions reduction benefit is mainly contributed by the lower electricity consumption at the wastewater treatment plants to treat influent with lower nitrogen and phosphorus content (Spångberg et al., 2014), the lower nitrous oxide emissions from reduced amount of nitrogen denitrified (Bradford-Hartke et al., 2015; Igos et al., 2017), the reduction in flush water production (Ishii and Boyer, 2015; Landry and Boyer, 2016), and the avoided production of fertilisers (Landry and Boyer, 2016). These factors can offset the increased GHG emissions from transporting of recycled nutrient products from urban area to agriculture.

Three studies (out of five studies) suggested that scenarios on source-separated urine/faeces application have higher acidification potential (over 50%) and eutrophication potential (over 10%) than centralised treatment scenarios because of high ammonia emissions during urine storage, urine application and composting of faeces (Maurer et al., 2003; Remy and Jekel, 2008; Spångberg et al., 2014). However, acidification potential and eutrophication potential are sensitive and directly related to the spreading strategy of urine and blackwater (Maurer et al., 2003; Spångberg et al., 2014). The other two studies that show opposite results were not sufficiently transparent for one to identify possible factors contributing to the discrepancy (Lam et al., 2015; Shi et al., 2018). For ecotoxicity, it is mostly determined by the difference in heavy metal and organic contaminants in source-separated urine/faeces and synthetic fertilisers. The contaminant content assumption may have led to the varying results in the three studies that assessed ecotoxicity.

None of the reviewed studies have directly compared decentralised recovery of phosphorus/ammonium products (from source-separated urine/faeces) to centralised recovery of the same products (at wastewater treatment plants). Three studies present scenarios to compare decentralised recovery of struvite to centralised wastewater treatment for nutrient removal (but not nutrient recovery). All of them show that decentralised struvite recovery could potentially lower acidification potential, ecotoxicity, eutrophication potential, human toxicity and photochemical ozone formation (Igos et al., 2017; Ishii and Boyer, 2015; Landry and Boyer, 2016). However, the environmental performance depends on the amount of chemical inputs for struvite precipitation (Ishii and Boyer, 2015), and is also influenced by other additional process functions such as pharmaceutical removal (Landry and Boyer, 2016). Maximising nutrient recovery from source-separated human excreta with the use of additional chemicals may compromise its environmental benefits. For instance, if sodium phosphate is used in addition to magnesium oxide alone to maximise struvite precipitation, the environmental performance of decentralised struvite precipitation from source-separated urine would become highly unfavourable (Ishii and Boyer, 2015).

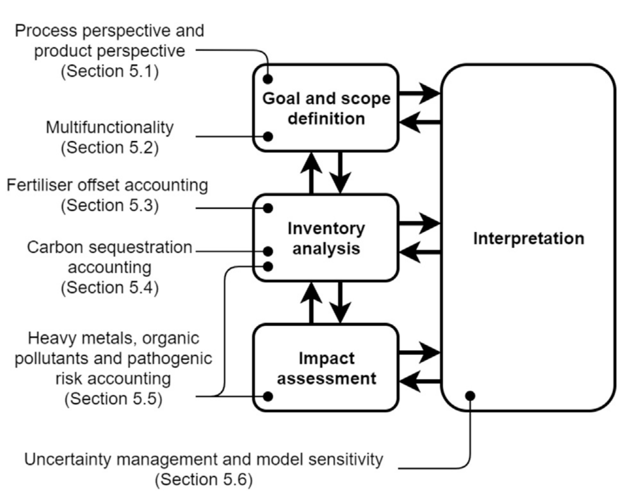

Fig. 5. Methodological aspects reviewed in the four phases of LCA.

5. Methodological practices

Some earlier LCA reviews on wastewater treatment, sludge management and resource recovery assessed studies based on each of the four LCA phases – goal and scope definition, inventory analysis, impact assessment and interpretation (Corominas et al., 2013; Gallego-Schmid and Tarpani, 2019; Sena and Hicks, 2018). In this review, we focus on some of the methodological aspects that are more related to the context of using LCA for decision making on nutrient recycling from wastewater to agriculture (Fig. 5). They include scope definition on recycling process or recycled product (Section 5.1), multifunctionality of studied systems (Section 5.2), fertiliser offset accounting (Section 5.3), carbon sequestration accounting (Section 5.4), heavy metals, organic pollutants and pathogen risk accounting for certain recycled nutrient products (Section 5.5), and uncertainty management and model sensitivity (Section 5.6).

5.1. Process perspective and product perspective

Most nutrient recycling related LCA studies took a "*process perspective*" in which a single process configuration with nutrient recycling was assessed, or different process configurations with nutrient recycling were compared. A process perspective was mostly chosen because the primary goal for most of these studies is wastewater and sludge management, while nutrient recycling is an additional function of these processes. This is reflected by the use of "input-based" functional units such as 1 m^3 of wastewater treated and 1 tonne of dry sludge disposed.

Only seven studies (out of 65 studies) took a "*product perspective*", which is getting more common in recent years with the growing interest of nutrient recycling. These studies used "output-based" functional units such as "11 kg pure phosphorus to agricultural land" (Linderholm et al., 2012) and "annual production of 1 kg of phosphorus available for plants in mineral form" (Pradel and Aissani, 2019). Most of these studies compared the recycled nutrient products to mineral or synthetic fertilisers to be substituted (or in the case of recycled water, compared to conventional water supply).

5.2. Multifunctionality

Systems are often multifunctional in nature. For instance, a sludge disposal process using anaerobic digestion with struvite crystallisation from digester supernatant would have functionalities of sludge management, energy recovery and nutrient recovery. In LCA, when there is a secondary function of the assessed system, a "system expansion" approach can be applied to credit the impacts of the secondary function (Bjørn et al., 2017). Among the studies reviewed, the primary functions of studied systems are mostly either wastewater treatment or sludge disposal. In these process perspective studies, the secondary function of nutrient recycling was predominantly considered through system expansion with substitution (52 out of 58). This accounts for the environmental impacts of avoiding the production of mineral or synthetic fertilisers (Section 5.3), considering that the recycled nutrient products are applied to agricultural land. The same substitution approach can be applied to credit biogas production for avoiding electricity generation.

The seven product perspective studies approached multifunctionality in a few different ways, though all of them used system expansion approach with substitution to credit fertiliser offset. Linderholm et al. (2012) and Kalmykova et al. (2015) only included the unit processes directly related to nutrient recycling (e.g., sludge transportation, incineration for phosphorus recovery, struvite precipitation) and excluded wastewater treatment and sludge treatment processes. Bradford-Hartke et al. (2015) excluded the existing infrastructure that remains unchanged after implementing phosphorus recovery. Amann et al. (2018) applied a similar approach to only account for the impacts from introducing new technologies and processes, relative to the reference system. Spångberg et al. (2014) used an integrated functional unit that considers both nutrient removal (i.e., removal of certain amount of nitrogen and phosphorus from wastewater) and nutrient recycling (i.e., production of fertiliser with certain amount of nitrogen and phosphorus). Unlike all the other studies, Pradel and Aissani (2019) allocated part of the environmental impacts of wastewater treatment to the "production" of sludge (a recycled nutrient product considered in the study) and included the sludge treatment in the system boundary.

Most of these product perspective studies were aiming for relative results for comparison across scenarios, instead of estimating "absolute" environmental impact potential of the recycled nutrient products. This essentially means that allocation could be avoided because functionalities other than nutrient recycling can be excluded from the system boundary, if the unit processes for delivering the other functionalities remain unchanged after introducing nutrient recycling function. Even if the non-nutrient recycling unit processes do change, all the impacts from the change could be allocated to the recycled nutrient products because it is a consequence of introducing nutrient recycling. The use of different approaches for handling multifunctionality shows that multifunctionality can be a challenge for conducting product perspective LCA, especially when multiple recycled nutrient products are produced (to be further discussed in Section 6.1).

5.3. Fertiliser offset accounting

Most of the reviewed studies (59 out of 65) accounted for the benefit of avoiding mineral or synthetic fertilisers. Among these 59 studies that had fertiliser offset accounting, nearly all of them considered nitrogen and phosphorus content in the recycled nutrient products for the offset credit. However, different levels of detail on their accounting method were documented – from giving no details (7 out of 59), to being transparent about what fertilisers being substituted (44 out of 59). In addition, these studies assumed a range of different avoided fertilisers – ammonium nitrate, monoammonium phosphate, diammonium phosphate, triple super phosphate and urea. For the bioavailability of the recycled nutrient products, struvite and urine were predominantly assumed to be 100% available compared to mineral or synthetic fertilisers, while digested sludge was mostly assumed to have 30–50% and 60–70% bioavailability for nitrogen and phosphorus respectively. Avoided fertilisers can benefit a number of environmental impact categories – global warming potential (Niero et al., 2014; Yoshida et al., 2018), freshwater eutrophication (Niero et al., 2014; Yoshida et al., 2018) and water use (Peters and Rowley, 2009).

In most studies, there is a clear spatial disconnection between the recycled nutrient products and local factors (e.g., soil-related factors, type of avoided fertiliser, fertiliser use practice). The choices of avoided fertiliser seem to be arbitrary, most likely depending on the types of fertiliser available in the life cycle inventory database used. In the case of phosphorus recovery from wastewater, the types of recovered material can vary highly (e.g., calcium phosphate, struvite, phosphorus-rich slag, single superphosphate), and it is not clear which types of fertiliser are going to be replaced by them (Amann et al., 2018).

5.4. Carbon sequestration accounting

Carbon sequestration refers to the carbon capture benefit when a fraction of the carbon content in land-applied sludge remains in the soil for a long period of time. It thus has a mitigation potential for the global warming potential impact category. While carbon sequestration is not directly related to nutrient recycling, it is relevant for sludge-based nutrient recycling pathways. In the context of shifting away from the direct land application of stabilised sewage sludge (digested/undigested), it would be of interest to understand the fate of the carbon (especially the non-biogenic fraction) and its global warming potential impact.

An earlier review on sewage sludge management found that only 3 out of 35 reviewed studies considered carbon sequestration for agricultural land application of sludge (Yoshida et al., 2013). Our review shows that the practice of considering carbon sequestration for agricultural land application of sludge remains limited (7 out of 33 studies since year 2013). Among these seven studies, the carbon sequestration accounting methods differ considerably. Yoshida et al. (2018) suggested that for their scenario of sandy loam soil and average Danish precipitation, 4.9–7.1% of the input carbon could be sequestered in the soil depending on the sludge type, based on a calibrated model from Bruun et al. (2016). Alanya et al. (2015) also used a modelling approach (Hermann et al., 2011) to quantify carbon sequestration. Literature values were used in two studies – 24% of the applied carbon (Bradford-Hartke et al., 2015) and 7% of the total amount of added carbon (Willén et al., 2017). The remaining studies do not have a clear documentation on their accounting methods. Some of these studies showed significant benefit of reducing global warming potential through carbon sequestration (Alanya et al., 2015; Liu et al., 2013; Miller-Robbie et al., 2015; Willén et al., 2017), while the other studies did not emphasise the contribution of carbon sequestration on global warming potential.

5.5. Heavy metals, organic pollutants and pathogenic risk accounting

Heavy metals, organic pollutants and pathogenic risk are of particular concern for sludge and phototrophic biomass, while they are less of concern for monomineral precipitates (e.g., struvite, urea, calcium phosphate) (Harder et al., 2019). Agricultural land application of contaminated recycled nutrient product (i.e., containing heavy metals, organic pollutants) contributes to human toxicity and ecotoxicity (midpoint life cycle impact indicators), and ultimately human health impact and natural environment impact (endpoint life cycle impact indicators).

Among the 29 studies that assessed ecotoxicity and/or human toxicity of sludge-based nutrient recycling, all but four accounted for heavy metal contamination to different extents (e.g., As, Cd, Cr, Cu, Hg, Ni, Zn, Pb). In most cases, heavy metal contents in sludge were assumed based on either literature values (Alanya et al., 2015; Bradford-Hartke et al., 2015; Suh and Rousseaux, 2002) or site-specific measurement (Do Amaral et al., 2018; Hospido et al., 2005; Shiu et al., 2017). Different metals behave entirely differently in a certain soil after land application (Alanya et al., 2015; Hospido et al., 2005), so studies then further assumed the fate of heavy metals after agricultural land application – all metals releasing to soil (ten Hoeve et al., 2018; Yoshida et al., 2018), using heavy metal transfer coefficients from literature to estimate the release of heavy metals to soil (Lederer and Rechberger, 2010; Lombardi et al., 2017) or assuming different bioavailability of metals (Blanco et al., 2016). The toxicity results are further influenced by the choice of impact assessment method in the impact assessment phase.

Organic pollutants refer to substances such as pharmaceutically active compounds, hormones, personal care products, detergents and organic compounds from polymeric pipes (Harder et al., 2019). A few studies (only 4 out of 37 studies that assessed ecotoxicity and/or human toxicity) accounted for the toxicity impacts of organic pollutants (Bradford-Hartke et al., 2015; Hospido et al., 2010; Landry and Boyer, 2016; Sablayrolles et al., 2010). Earlier studies might be limited by the information regarding organic pollutants present in sludge or other recycled nutrient products. In addition, the potential environmental impacts of the target organic pollutants can only be quantified when the chosen impact assessment method has their characterisation factors. Many organic pollutants are missing in common impact assessment methods, while more organic pollutants are being included recently (Bradford-Hartke et al., 2015). This can be a methodological area that is relevant for wastewater-based nutrient recycling LCA and requires further development.

Impacts of pathogens are not included in current impact assessment methods. There were some recent attempts to include pathogenic risk in life cycle assessment (Harder et al., 2014, 2015; Heimersson et al., 2014), or to conduct integrated assessments of life cycle environmental impacts and pathogenic risk (Anastasopoulou et al., 2018). Similar to the case of organic pollutants, LCA method has to be further adapted to the needs of environmental assessment of wastewater-based nutrient recycling opportunities. Legal barrier for the use of products recovered from wastewater is one of the uncertainties for the wide adoption of resource recovery from wastewater (van der Hoek et al., 2016). Robust accounting frameworks on potential environmental impacts would facilitate the discussion on changing legislation that restricts the use of recycled nutrient products.

5.6. Uncertainty management and model sensitivity

Uncertainty is present in all phases of an LCA, and can originate from, for example, variability, measurement errors, inventory gaps, methodological choices made and model uncertainty (Mendoza Beltran et al., 2018). Among the reviewed studies, earlier studies rarely discussed the potential uncertainty of their results, let alone performing analytical or numerical uncertainty analysis. Recent years have started seeing studies using analytical or numerical approaches to examine uncertainty (Igos et al., 2017; Kavvada et al., 2017). Most of these studies used Monte Carlo simulations to examine uncertainty propagation. In Monte Carlo simulations, each input parameter of an LCA model is represented by a probability distribution function instead of a point value. The calculation of LCA model is repeated for a fixed number of iterations using input parameters that are randomly sampled from those distributions. Probability distributed results can then be obtained. The approach helps provide a measure of confidence to the results and testify whether the conclusion drawn is robust or not (Niero et al., 2014). Most recent studies reviewed used major commercial LCA software (e.g., SimaPro, Gabi), which have built-in Monte Carlo analysis functionality and uncertainty estimates in their life cycle inventory databases. This may explain the increased popularity of the approach for uncertainty analysis.

For the research area of nutrient recycling LCA, the practice of uncertainty analysis remains limited. Considering that a lot of the reviewed studies are comparative LCA that aims to contrast the potential environmental performance of alternatives, the lack of consideration of potential uncertainty may limit the robustness of the results for decision making (Guo and Murphy, 2012). For seven studies with Monte Carlo simulations, uncertainty analysis was mostly used to establish confidence intervals in their results (with most of them specifying the percentile of the confidence interval used). Only four of them further discussed the results in the context of uncertainties. For instance, Niero et al. (2014) found that based on both uncertainty analysis and sensitivity analysis, it is inconclusive as to whether land application of sludge for recycling phosphorus is better than incineration of sludge, in terms of eutrophication and toxicity-related impacts. In another example, Igos et al. (2017) showed that pre-treating a portion of urine from the sewage network before centralised treatment with biological phosphorus removal has significant benefits compared to the conventional fully-centralised treatment approach, even considering input uncertainties.

There is an ongoing discussion on the methodological aspects of uncertainty analysis in LCA – for instance, on handling the issue of correlations between input parameters in uncertainty propagation (Groen and Heijungs, 2017), on the need of statistically supported conclusions by null hypothesis testing in comparative LCA (Henriksson et al., 2015), and on exploring a broad range of scenario space (Gregory et al., 2016). In addition to its continuous development, uncertainty analysis does not capture all uncertainties (e.g., model uncertainty, epistemological uncertainty), but the information is still useful for the result interpretation phase (Rosenbaum et al., 2018).

Sensitivity analysis is an important step for the interpretation of LCA results. Combining uncertainty analysis with sensitivity analysis helps understand the impact of any uncertainty on the results, and identify focus points for improved inventory data collection or impact assessment (Hauschild et al., 2017). Sensitivity analysis often involves changing selected input parameters one at a time to examine how sensitive the LCA results are to each change (Clavreul et al., 2012). Half of the reviewed studies have performed some forms of sensitivity analysis to examine model sensitivity. It was mainly used for understanding the influence of selected input parameters on LCA results, or for identifying key influencing input parameters. Ideally, it is conducted with uncertainty analysis, because certain input parameters could have higher uncertainty, but the results might not be sensitive to these parameters (i.e., lower sensitivity). In this case, effort would be prioritised to reduce the uncertainty of input parameters that the models are more sensitive to. This is particularly important where the technology assessed is still in its early stage of development. Sensitivity analysis can highlight areas for further research to minimise the uncertainty in more influential input parameters.

The reviewed studies highlighted some major uncertainties, including inventory data gap for non-full-scale technology (Amann et al., 2018), missing "emerging" organic contaminants in current impact assessment methods (Bradford-Hartke et al., 2015), the variability of water and sludge quality parameters (Niero et al., 2014), the variability of spatial characteristics for decentralised recovery (Kavvada et al., 2017), non-localised inventory for background data (Igos et al., 2017), temporal representativeness of background data (Igos et al., 2017), emissions from field application of recycled nutrient materials (Maurer et al., 2003; Peters and Rowley, 2009). Future LCA studies could consider some of these uncertainties to prioritise the effort for inventory data collection.

6. Future outlook

LCA can be further developed to address some of the methodological challenges (Section 6.1) and applied at various scales to support decisions on wastewater-based nutrient recycling, and more broadly integrated resource recovery from urban water systems. Future opportunities could be on performing more "*product perspective*" LCA on recycled nutrient products (Section 6.2), integrating "*process perspective*" LCA with other systems approaches for selecting and optimising individual recovery processes (Section 6.3), assessing emerging nutrient recovery technologies (Section 6.4), assessing integrated resource recovery systems (Section 6.5), and going beyond single process/product assessment for "bigger picture" systems analysis at city, national and global level (Section 6.6).

6.1. Methodological challenges

One of the challenges of performing a product perspective LCA on recycled nutrient products is to handle the multifunctional nature of most processes (Section 5.2). The question is what portion of the environment impacts of building and operating the treatment process (that provides the functions of waste disposal, nutrient recovery and other forms of resource recovery) should be allocated to the recycled nutrient product or each of the recycled nutrient products (in multi-product cases). End users may want to know the "absolute" environmental footprints of recycled nutrient products, so as to directly compare them to that of conventional fertilisers. In that case, relative results from most of the studies are not so useful because of potential inconsistencies among the accounting frameworks used. This multifunctionality issue would become even more challenging for integrated resource recovery facilities, where different types of recovery products could be yielded.

"Zero-burden assumption" is also related to the multifunctionality issue. In LCA, the zero-burden assumption refers to not considering the environmental impacts associated with waste generation (i.e., outside system boundary). For instance, the environmental impact of digested sludge application for agricultural land does not include that of wastewater treatment process under the zero-burden assumption. Pradel et al. (2016) questioned the use of zero-burden assumption on assessing the environmental footprints of "products" recovered from "wastes".

There is still a lack of empirical understanding of the release and plant availability of phosphorus in soil from recovered phosphorus products from wastewater (Melia et al., 2017) (Section 5.3). An improved modelling of the processes for the application and post-application phases of recycled nutrient products is essential. Research is undergoing to address this challenge, but mostly limited to sewage sludge (Bruun et al., 2016; ten Hoeve et al., 2018). This modelling aspect is especially important for product perspective LCA, because the use of a nutrient-based functional unit may imply that the results are more sensitive to the assumed properties of the recycled nutrient products (e.g., bioavailability, nutrient content).

Other methodological challenges are related to limitations of existing impact assessment methods. Firstly, phosphorus resource depletion is not currently included in many impact assessment methods (except the CML method, and the recent inclusion in the ReCiPe method). This means that if an "input-based" functional unit is used, the direct benefit of reducing phosphorus resource extraction is not being accounted for. Instead, only the indirect benefits of reducing resource inputs and emissions (from for example, fertiliser offset) are included. Secondly, many organic pollutants are missing in common impact assessment methods, as discussed in Section 5.5.

6.2. Product perspective LCA for nutrient recycling

Currently, a product perspective LCA for nutrient-rich materials recycling from wastewater is still rare (Section 5.1). Future research can perform more product perspective LCA to assess recycled nutrient products. Monitoring environmental and consumer concerns is considered to be a challenge for the utilisation of the recycled nutrient products from wastewater (Hukari et al., 2016). A product perspective LCA allows the end users of recycled nutrient products to compare the potential environmental impacts of agricultural land application of recycled nutrient products to that of conventional fertilisers. As more recycled nutrient products are becoming available on the market, end users may consider the environmental impacts of the recycled nutrient products, in addition to their quality and application costs. This would require a consistent accounting approach to quantify and even certificate the life cycle environmental impact potential of the recycled nutrient products.

6.3. Process perspective LCA with other systems approaches

Future research for process perspective LCA can combine life cycle assessment with other systems approaches (e.g., life cycle cost analysis, dynamic process modelling) to aid the design and optimisation of nutrient recovery processes (centralised or distributed) and their integration into existing wastewater infrastructure. While a product perspective LCA can help end users choose what recycled nutrient products suit them more from an environmental point of view, a process perspective LCA is useful for wastewater utilities to select and optimise recovery processes. Increasingly, there are studies customising and integrating LCA approaches with other modelling approaches for systems analysis of nutrient recovery processes. Kavvada et al. (2017) combined life cycle environmental assessment with economic assessment, geospatial modelling and last-mile logistics modelling to assess a decentralised nitrogen recovery approach at a city scale. Igos et al. (2017) coupled LCA with process simulation and economic assessment to assess a new source-separated urine technology for struvite precipitation and ammonia recovery. Integrated assessment coupling LCA with other systems approaches is also gaining popularity for exploring other wastewater-based resource recovery – for example, optimising the integration of wastewater energy recovery to local energy supply (Kollmann et al., 2017), multi-metric sustainability assessment of material recovery from wastewater (Zijp et al., 2017).

6.4. Using LCA to assess emerging nutrient recovery technologies

Traditionally, LCA was developed for detailed analysis of products or processes that have clear specifications of their resource consumptions and emissions (Olsen et al., 2017). LCA has now been increasingly applied with a future perspective to assess emerging technologies that do not necessarily have full-scale inventory. Assessing a technology at its early development stage with LCA can provide an opportunity to identify environmental impacts that can be potential barriers for its full-scale implementation (Fang et al., 2016), to gain insight on technical factors that require further research and development (Kavvada et al., 2017), to support scale-up (Arvidsson et al., 2018), and to ensure that only the more promising development paths are pursued before lock-in occurs (Hermansson et al., 2019).

A number of future-oriented LCA studies assessed laboratory-stage or pilot-stage nutrient recycling approaches – ion-exchange based nitrogen recovery (Kavvada et al., 2017; Lin et al., 2016), microalgae-based nutrient recovery (Arashiro et al., 2018; Fang et al., 2016), phosphorus recovery from sludge incineration ash (Amann et al., 2018; Lundin et al., 2004), phosphorus recovery using biological acidification (Pradel and Aissani, 2019), nitrogen recovery using microbial electrolysis cell (Igos et al., 2017), and decentralised facility for nutrient recycling from human excreta (Shi et al., 2018). Into the future, emerging nutrient recovery techniques such as bioelectrochemical systems and sorption processes could be subjects for life cycle assessment.

Future-oriented LCA study of emerging technologies is also referred to as prospective LCA (Arvidsson et al., 2018). Conducting prospective LCA is more challenging than conventional LCA because of potentially incomparable functional units and system boundaries between new technology and existing technology, scaling issues between laboratory/pilot-scale process and full-scale process, larger data gaps, and higher uncertainty (Hetherington et al., 2014). Some of the reviewed future-oriented LCA studies on emerging technology overcame some of these challenges by using process simulation for scaling up technology (Igos et al., 2017), assessing and reporting uncertainty (Kavvada et al., 2017), and providing transparent and detailed inventories (Amann et al., 2018).

6.5. Using LCA to assess integrated resource recovery

As we are having a better understanding on some of the recycling pathways or technologies, technical details or certain technologies should not only be assessed in isolation as it can be a barrier to identify broad patterns and opportunities in the field as a whole (Harder et al., 2019). No single technology can effectively recover all the nutrients in waste stream (Mehta et al., 2015). We can only maximise nutrient recycling by combining individual pathways and technologies. For example, Wang et al. (2018) assessed the life cycle environmental impacts of a proposed wastewater treatment system with integrated resource recovery of water (for irrigation), nutrients (through irrigated water, struvite and biosolids), energy (from combustion of nitrous oxide, methane and sludge) and material (as polyhydroxyalkanoate (PHA)).

Trade-offs and synergies may exist between different resource recovery approaches (Mo and Zhang, 2013). LCA should be used to assess wastewater systems with integrated resource recovery (e.g., energy recovery, water recycling, material recovery, co-management with food waste). The multifunctional nature of wastewater systems with integrated resource recovery implies that a single functional unit may not be ideal for comparing different configurations of these systems. The use of multiple functional units to assess alternative can provide more opportunities for cross-study comparisons (Sena and Hicks, 2018), and can also provide insights from various perspectives (e.g., nutrient recovery efficiency, wastewater treatment efficiency).

6.6. Using LCA for "bigger picture" systems analysis

Going beyond individual recovery technologies and pathways to better integrate end-user needs and the bigger picture is valuable (Harder et al., 2019). "Bigger picture" systems analysis involves understanding the performance and impacts of wastewater-based nutrient recycling opportunities in the context of the wider systems at city, national and global level. This could be particularly important for decentralised solutions as geospatial characteristics likely lead to spatial variation in the performance of distributed systems within a given region. In addition, systems analysis that uses a multi-regional approach could enable inter-regional comparison, benchmarking and learning. It also provides a better understanding of the impacts of geospatial characteristics on the system performance. A number of recent works have used systems approaches to explore the bigger picture issues of resource recovery from wastewater. Trimmer and Guest (2018) estimated for 56 cities globally the transport distances for nutrient recycled from human excreta to the surrounding agricultural demand. Wang et al. (2015) assessed the environmental impacts of transitioning wastewater treatment plants from resource removal to resource recovery in 50 countries.

While LCA was traditionally developed to assess products or processes in more confined scales, it has now also found its roles in bigger picture systems analysis for water-related issues at city, national and global level. Kavvada et al. (2016) integrated life cycle environmental modelling with spatial modelling to evaluate the impacts of spatial characteristics on the energy use and greenhouse gas emissions of city-scale non-potable water reuse. Wang et al. (2019) simulated the life cycle environmental impacts of reducing nutrient discharge from domestic wastewater globally with consideration of spatial and temporal variations. Strategic use of LCA helps identify hotspots for action and unveil potentially unintended indirect impacts, and ultimately, contributes to the planning and implementation of nutrient and resource recovery from urban water systems.

As discussed in Section 4.3, while the use of sewage sludge for agriculture land application is restricted in some countries, the practice is still a major sludge disposal method in many countries. With increasing concern on potential soil contamination, there could be a shift from this practice to other sludge disposal methods with/without nutrient recycling. LCA could provide a framework to understand the potential environmental consequences of this large-scale change of practice.

7. Conclusions

This work reviewed LCA studies that considered nutrient recycling from wastewater for agricultural land application. The implications are as follows.

- More studies suggested positive environmental outcomes from wastewater-based nutrient recycling, especially when chemical inputs are minimised and source separation of human excreta is achieved.
- While the LCA methodology has been generally well adapted and increasingly applied for assessing wastewater-based nutrient recycling opportunities, there is still a need to further harmonise and develop methodological aspects (and guidelines) that are more relevant to nutrient recycling (e.g., multifunctionality, fertiliser offset accounting, contaminant accounting). Even for the few methodology aspects that we explored, methodological inconsistencies are prominent. Inconsistent method can be a barrier for the robust use of LCA for decision making on wastewater-based nutrient recycling.
- In comparing similar studies in this review, it is evident that some studies lack transparency on their inventory, underlying methodological assumptions and system boundary (i.e., what components are being included or excluded). The lack of transparency prevents meaningful comparison across some studies, and result generalisation.
- Integrating up-to-date cross-disciplinary knowledge (e.g., agricultural science, soil science, environmental toxicology) into LCA models is needed to better quantify the environmental impacts of applying recycled nutrient products. In many studies, there is a clear spatial disconnection between the recycled nutrient products and local factors such as soil-related factors and type of avoided fertiliser.
- In many of the studies reviewed, LCA was used in a comparative context to contrast different system configurations with nutrient recycling, or to compare nutrient recycling systems against reference systems without nutrient recycling. There is a clear lack of consideration of result uncertainty in these comparative LCA studies. If the results of LCA are to be used to support decision, result uncertainty has to be explored and communicated.
- The future opportunities could be on performing more "*product perspective*" LCA on recycled nutrient products, integrating "*process perspective*" LCA with other systems approaches for selecting and optimising individual recovery processes, assessing emerging nutrient recovery technologies, assessing integrated resource recovery systems, and going beyond single process/product assessment for "bigger picture" systems analysis at city, national and global level.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

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