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Review

Life cycle assessment applied to wastewater treatment: State of the art

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ABSTRACT

Life cycle assessment (LCA) is a technique to quantify the impacts associated with a product, service or process from cradle-to-grave perspective. Within the field of wastewater treatment (WWT) LCA was first applied in the 1990s. In the pursuit of more environmentally sustainable WWT, it is clear that LCA is a valuable tool to elucidate the broader environmental impacts of design and operation decisions. With growing interest from utilities, practitioners, and researchers in the use of LCA in WWT systems, it is important to make a review of what has been achieved and describe the challenges for the forthcoming years. This work presents a comprehensive review of 45 papers dealing with WWT and LCA. The analysis of the papers showed that within the constraints of the ISO standards, there is variability in the definition of the functional unit and the system boundaries, the selection of the impact assessment methodology and the procedure followed for interpreting the results. The need for stricter adherence to ISO methodological standards to ensure quality and transparency is made clear and emerging challenges for LCA applications in WWT are discussed, including: a paradigm shift from pollutant removal to resource recovery, the adaptation of LCA methodologies to new target

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compounds, the development of regional factors, the improvement of the data quality and the reduction of uncertainty. Finally, the need for better integration and communication with decision-makers is highlighted.

Contents

1. Introduction

Life cycle assessment (LCA) is a technique to quantify the impacts associated with all the stages of a product, service or process from cradle-to-grave. LCA had its beginnings in the 1960s and since then a large number of approaches have been developed for different disciplines. In the late 1990s pressure grew to standardize LCA methodologies, which led to the development of LCA standards in the International Standards Organization (ISO) 14000 series. The ISO 14040 and 14044 standards (ISO 14040, 2006; ISO 14044, 2006) define a general methodology but are not designed to define the details for each field in which the method is used. In recent years LCA has gained popularity as an assessment tool for environmental sustainability (Guinée et al., 2011) as evidenced by the rapidly increasing number of publications and databases supporting its implementation.

Within the field of wastewater treatment (WWT), LCA was already applied in the 1990s. Since then, more than forty studies have been published in international peer-reviewed journals using an array of databases, boundary conditions, and impact assessment methods for interpreting the results. In the pursuit of more environmentally sustainable WWT, it is clear that LCA is a valuable tool to elucidate the broader environmental impacts of design and operation decisions (Guest et al., 2009; Larsen et al., 2010). With growing interest from utilities, practitioners, and researchers in the use of LCA in WWT systems, it is important to make a review of what has been achieved and describe the challenges for the forthcoming years.

Several reviews have been published on water treatment and LCA. Friedrich et al. (2007) published a paper that reviewed 20 studies on LCA and wastewater, highlighting key aspects, but did not go deep into the characterization of the studies. A book chapter on Life Cycle Analysis in Wastewater was also published (Ahmed, 2011) where an LCA framework for wastewater treatment was presented. More recently, LCA methodology was included within a review of sustainability assessments of recycled water schemes (Chen et al., 2012). In our opinion, none of these documents provided a complete and comprehensive review on wastewater treatment LCA studies and defined the challenges for the forthcoming years. Therefore, the goal of this paper is to perform a critical review of relevant papers published on the topic and to describe the challenges for LCA applied to WWT. The scope of the review includes only peer-reviewed papers published in journals and one relevant report that is publically available. Papers focused on sludge treatment and disposal without considering the

Abbreviations

Agr	Agriculture
Const	Construction
C techs	Conventional technologies
Dem	Demolition
DEHP	Di(2-ethylhexyl)phthalate
DW	Drinking water
GHG	Greenhouse Gas
ISO	International Standards Organization
kg	Kilograms
LCA	Life cycle assessment
LCIA	Life cycle impact assessment
ML	Megaliter
MBRs	Membrane bioreactors
CH_4	Methane
MEC	Microbial electrolysis cell
MFC	Microbial fuel cell
Ni	Niquel
N_2O	Nitrous oxide
NonC techs	Non-conventional technologies
Op	Operation
PO_4^{3-} eq	Phosphate equivalent
P	Phosphorus
PAH	Polycyclic aromatic hydrocarbons
PE	Population equivalent
PP	Priority pollutant
Sew	Sewer system
SD	Sludge disposal
ST	Sludge treatment
So	Source treatment
WWT	Wastewater treatment
WWTP	Wastewater treatment plant

water line have been excluded as we feel they belong more to the waste management sector than to the WWT field.

This paper is structured as follows: firstly, the historical evolution of LCA and WWT is presented briefly describing the lessons learnt in the last 17 years. Then, the reviewed studies are analysed following the LCA phases: goal & scope, inventory, impact assessment and interpretation, in order to identify common elements and distinguishing aspects. Finally the challenges in this field are identified and discussed.

2. Literature review

Although water sanitation dates from Mesopotamian times (Lofrano and Brown, 2010) the currently applied activated sludge process was not described until 1913 in the United Kingdom (Ardern and Lockett, 1914). During the 20th century, water sanitation systems protected large populations from disease. However, the society did not realize that there were other environmental costs associated with water sanitation. After the term sustainable development was defined by the World Commission on Environment and Development (WCED, 1987), some WWT practitioners and researchers incorporated LCA techniques in order to evaluate the environmental implications of WWT. The evolution of LCA is explained through the papers available in the literature and the different objectives which have been evaluated. Table 1 lists all the studies included in this review with their main characteristics (in the supplementary data of the paper more detailed information is provided).

2.1. *Evaluation of the environmental performance of conventional activated sludge technologies*

To the best of our knowledge, the first LCA study applied to wastewater treatment plants (WWTPs) published in an international peer reviewed journal was focused on the inventory phase to evaluate different small-scale WWT technologies (Emmerson et al., 1995). They highlighted the importance of including the emission of CO_2 associated with energy production, thus introducing second order (background) impacts in the evaluation of environmental performance. Electricity use was identified as one of the main contributors to the depletion of fossil resources and the generation of Greenhouse Gas (GHG) emissions. The construction and demolition phases were included in the analysis in addition to the evaluation of operation of the system. Afterwards, a more sophisticated LCA methodology was used to evaluate the societal sustainability of municipal WWT in the Netherlands (Roeleveld et al., 1997) and the results highlighted the importance of reducing effluent pollution (nitrogen, phosphorus) and minimizing the sludge production. Contrary to the previous study, it was concluded that the contribution of impacts related to energy consumption were very low. That conclusion was achieved after normalizing the results, meaning that the environmental impacts estimated from WWT in the Netherlands were expressed as a percentage of the total environmental impacts in the Netherlands. The outcome was that WWTPs contributed to less than 1% of energy consumption at that time. This example addresses the effect of normalizing the impacts in the LCA studies. Construction impacts and the use of chemicals were not found to be significant in their evaluation. Since the Roeleveld study, LCA has been applied to evaluate different types of conventional WWTPs. First, LCA has been used to characterize the environmental impact of specific case-studies (Clauson-Kaas et al., 2001; Hospido et al., 2004; Pasqualino et al., 2009; Bravo and Ferrer, 2011; Venkatesh and Brattebø, 2011). Second, LCA has been applied to the outcomes of dynamic simulation exercises using activated sludge models; in the case of Flores-Alsina et al. (2010) and in Corominas et al. 2013 control strategies for nitrogen removal were evaluated and in Foley et al. (2010a) multiple biological nutrient removal configurations were analysed. Third, LCA studies have been conducted to compare the performance of different configurations applied to a single system to improve the performance (Mels et al., 1999; Vidal et al., 2002; Rebitzer et al., 2003; Clauson-Kaas et al., 2004). Finally, multiple conventional systems have also been compared (Gallego et al., 2008; Hospido et al., 2008; Rodriguez-Garcia et al., 2011). The outcomes were very similar in all of the studies that involve nutrient removal, highlighting the trade-offs between

Table 1 – Main characteristics of the references included in the literature review.

Reference	Objective	Boundaries	Process considered	Waste disposal	Phases included	GHG emissions	FU	Impact assessment methodology
(Emmerson et al., 1995)	C techs	D	(1)(2)(ST)(SD)	Yes (Agr)	Op, Const, dem	Direct & indirect	1000 PE, 15 ys	Only inventory
(Roeleveld et al., 1997)	C techs	B	(1)(2)(3)(ST)	No	Op, Const	Direct & indirect	100,000 PE	Not specified
(Tillman et al., 1998)	Water cycle	F	(So)(2)(ST)(SD)	Yes (Agr)	Op, Const	Direct & indirect	1 PE per y	Not specified
(Brix, 1999)	NonC techs	B	(2)(3)	No	Op	No	1 m3	Only inventory
(Mels et al., 1999)	C techs	D	(1)(2)(3)(ST)(SD)	Yes	Op	No	100,000 PE	Only inventory
(Lundin et al., 2000)	Water cycle	H	(So)(2)(SD)	Yes (Agr)	Op, Const	Indirect	1 PE per y	Not specified
(Clauson-Kaas et al., 2001)	C techs	D	(2)(ST)(SD)	Yes (Agr)	Op	Indirect	1 m3	EDIP 2003
(Kärrman and Jönsson, 2001)	Water cycle	H	(DW)(So)(2)(SD)	Yes (Agr)	Op	Indirect	1 PE per y	Not specified
(Lundin and Morrison, 2002)	Water cycle	H	(DW)(2)(ST)(SD)	Yes (Agr)	Op	Indirect	1 PE per y	Not specified
(Vidal et al., 2002)	C techs	C	(2)	No	Op	Direct & indirect	1 Tn	Not specified
(Beavis and Lundie, 2003)	NonC techs	A, G	(2)(3)(ST)(SD)	Yes (Agr)	Op	Direct & indirect	1 ML	Not specified
(Dixon et al., 2003)	NonC techs	C	(2)	No	Op, Const	Direct & indirect	1 PE	Not specified
(Rebitzer et al., 2003)	C techs	F	(1)(2)(ST)(SD)	Yes (Agr)	Op	Indirect	1 PE per y	Not specified
(Clauson-Kaas et al., 2004)	C techs	D	(2)(SD)	No	Op	Direct & indirect	1 L	EDIP
(Hospido et al., 2004)	C techs	F	(1)(2)(ST)(SD)	Yes (Agr)	Op	Direct & indirect	1 m3 per d	CML 2000
(Lundie et al., 2004)	Water cycle	H	(DW)(Sew)(2)(3)(ST)(SD)	Yes (Agr)	Op, Const	Direct & indirect	1 KL	Not specified
(Muñoz et al., 2005)	NonC techs	A	(+)	No	Op	Indirect	1 m3	Not specified
(Tangsubkul et al., 2005)	NonC techs	D	(1)(2)(3)(ST)(SD)	Yes (Agr)	Op, Const	Direct & indirect	1 mL of recycled water	Not specified
(Tangsubkul et al., 2006)	NonC techs	A	(2)	No	Op, Const	Indirect	1 ML per d	Not specified
(Vlasopoulos et al., 2006)	NonC techs	A	(1)(2)(+)	No	Op, Const	Indirect	10,000 m3/d for 15 ys	CML 2000
(Lassaux et al., 2007)	Water cycle	H	(DW)(Sew)(2)(ST)(SD)	Yes (Agr)	Op, Const	Indirect	1 m3	Eco-Indicator 99
(Machado et al., 2007)	NonC techs	F	(2)(SD)	Yes (Agr)	Op, Const, Dem	Direct & indirect	1 PE	CML 2000
(Ortiz et al., 2007)	NonC techs	B	(1)(2)(3)(ST)	Yes	Op, Const, Dem	Indirect	3000 m3/d for 25 ys	CML 2000, Eco-Points 97, Eco-Indicator 99
(Gallego et al., 2008)	C techs	F	(1)(2)(ST)(SD)	Yes (Agr)	Op	Direct & indirect	1 PE per y	CML 2000
(Høibye et al., 2008)	NonC techs	D	(3)(ST)(SD)	Yes	Op	Indirect	1 m3	EDIP
(Hospido et al., 2008)	C techs	F	(1)(2)(ST)(SD)	Yes (Agr)	Op	Direct & indirect	1 PE	CML 2000
(Muñoz et al., 2008)	Impact method	A	(1)(2)	Yes (Agr)	Op	No	1 L	EDIP 97, USES-LCA
(Remy and Jekel, 2008)	Water cycle	H	(So)(2)(ST)(SD)	Yes (Agr)	Op, Const	Indirect	1 PE per y	CML
(Renou et al., 2008)	Impact method	D	(1)(2)(ST)(SD)	Yes (Agr)	Op, Const	Indirect	1 m3 per y	CML 2000, Eco-Indicator 99, Ecopoint 97, EDIP 97, EPS
(Wenzel et al., 2008)	NonC techs	D	(3)(SD)	Yes	Op	Indirect	1 m3	EDIP 2003
(Nogueira et al., 2009)	NonC techs	D	(2)(SD)	Yes (Agr)	Op, Const	Direct & indirect	1 PE	CML 2000
(Pasqualino et al., 2009)	C techs	F	(1)(2)(ST)(SD)	Yes (Agr)	Op	Indirect	1 m3	CML 2000
(Flores-Alsina et al., 2010)	C techs	F	(2)(ST)(SD)	Yes (Agr)	Op	Direct & indirect	753,3 Hm3	CML 2000
(Foley et al., 2010b)	NonC techs	D	(+)(ST)(SD)	Yes	Op, Const	Direct & indirect	2200 m3/d at 4000 mg COD/l over 10 ys	IMPACT 2002+
(Foley et al., 2010a)	C techs	F	(1)(2)(ST)(SD)	Yes (Agr)	Op, Const	Direct & indirect	10 ML/d over 20 ys	Only inventory
(Larsen et al., 2010)	NonC techs	D, F	(1)(2)(3)(+)(ST)(SD)	Yes (Agr)	Op, Const, Dem	Direct & indirect	1 m3	EDIP97
(Stokes and Horvath, 2010)	C techs	H	(1)(2)(ST)(SD)	Yes (Agr)	Op, Const	Direct & indirect	1 Ml	Not specified

(continued on next page)

Table 1 – (continued)

Reference	Objective	Boundaries	Process considered	Waste disposal	Phases included	GHG emissions	FU	Impact assessment methodology
(Bravo and Ferrer, 2011)	C techs	B	(1)(2)(3)(ST)(SD)	No	Op	Indirect	50,000 PE	CML 2000
(Pasqualino et al., 2011)	C techs	D	(1)(2)(3)(ST)(SD)	Yes	Op	Indirect	1 m3	CML 2000
(Rodriguez-Garcia et al., 2011)	C techs	F	(1)(2)(3)(ST)(SD)	Yes (Agr)	Op	Direct & indirect	1 m3 and 1 kg of PO43- removed	CML
(Venkatesh and Brattebø, 2011)	C techs	F	(1)(2)(ST)(SD)	Yes (Agr)	Op	Direct & indirect	1 m3	CML 2001
(Hospido et al., 2012)	NonC techs	F	(2)(ST)(SD)	Yes (Agr)	Op	Direct & indirect	1 m3	CML 2002, RECIPE and IMPACT 2002+
(Kalbar et al., 2012a)	NonC techs	D	(2)(ST)(SD)	Yes (Agr)	Op	Direct & indirect	1 PE per y	CML 2000
(Remy and Jekel, 2012)	Water cycle	H	(So)(2)(ST)(SD)	Yes (Agr)	Op, Const	No	1 PE per y	Not specified
(Yildirim and Topkaya, 2012)	NonC techs	D	(1)(2)(ST)(SD)	Yes (Agr)	Op, Const	Direct & indirect	1 PE	CML 2000

(DW) drinking water; (So) source treatment; (1) primary treatment; (2) secondary treatment; (3) tertiary treatment; (+) advanced treatment; (ST) sludge treatment; (SD) sludge disposal; (Sew) sewer system; C techs: Evaluation of conventional technologies; NonC techs: evaluation of non-conventional technologies; Agr: agriculture; Op: operation; Const: construction; Dem: demolition. For the boundaries refer also to Fig. 2. (more details can be found in the supplementary data).

eutrophication, toxicity and global warming impact categories caused mainly by water discharge emissions, sludge treatment and disposal and electricity use respectively. The improvement of local water quality is at the cost of regional/global effects stemming from energy and chemical production. Overall, the best alternatives seem to be the ones that result in lower nutrient emissions.

2.2. *Evaluation of non-conventional technologies*

For non-conventional technologies (NonC techs) we understand any technology which is not based on activated sludge systems followed by a sedimentation tank. The reality is that conventional WWT technologies are costly and energy demanding, which is troublesome particularly in small communities (<2000 population equivalents, PE). Constructed wetlands, biological filters and sand filtration systems have been proposed as feasible alternatives with lower environmental impacts compared to conventional technologies after using LCA (Brix, 1999; Dixon et al., 2003; Vlasopoulos et al., 2006; Machado et al., 2007; Nogueira et al., 2009; Kalbar et al., 2012a; Yildirim and Topkaya, 2012). Although these low-tech processes require larger land areas for their implementation, they are often appropriate for rural zones because of the low energy requirements and the high efficiencies to remove heavy metals.

Emerging technologies for wastewater treatment are being developed and it becomes a common practice to use LCA as the methodology to compare them against conventional technologies. This is the case for instance of microbial fuel (MFC) and electrolysis (MEC) cells (Foley et al., 2010b), advanced oxidation processes (AOPs) (Muñoz et al., 2005) or membrane bioreactors (MBRs) (Tangsubkul et al., 2006; Vlasopoulos et al., 2006; Ortiz et al., 2007; Høibye et al., 2008; Wenzel et al., 2008; Foley et al., 2010a,b; Hospido et al., 2012; Remy and Jekel, 2012). In the case of MEC technology, significant environmental benefits can be achieved through the cost-effective production of useful chemicals (e.g. hydrogen peroxide). Regarding the comparison of advanced oxidation processes, using solar energy reduces drastically the environmental impacts as the source of energy required is the key aspect. In the case of MBRs, energy use has also been pointed out as a key element that needs to be optimized in order to improve the environmental performance. It is worth noting that when using LCA in technology development, laboratory scale data is used, which certainly limits the usefulness of the results with regard to a real application.

In recent years the effect of micropollutants (priority and emerging pollutants) on ecosystems and their fate and removal in WWTP have been studied (Verlicchi et al., 2012). These pollutants include metals and organics such as pharmaceuticals and personal care products (including endocrine disrupters). As a result several technologies for micropollutants removal are being proposed (e.g. ozonation, advanced oxidation, activated carbon) and evaluated using LCA (Høibye et al., 2008; Wenzel et al., 2008; Larsen et al., 2010). Due to uncertainty surrounding characterization factors for micropollutants, these studies showed moderate or even no environmental benefits from their removal depending on the evaluated technology. Therefore, further research is needed to better characterize the implications of micropollutants in the aquatic environment.

2.3. Expanding boundaries for the evaluation of management strategies for the urban water/wastewater system

The boundaries of the WWTPs have been expanded in some studies to include the whole urban water/wastewater system, i.e. withdrawal of freshwater, drinking water production, distribution & use of drinking water, generation of wastewater and transport to the wastewater treatment plant. Several studies (Tillman et al., 1998; Lundin et al., 2000; Kärrman and Jönsson, 2001; Lundin and Morrison, 2002; Lassaux et al., 2007; Remy and Jekel, 2008, 2012) modelled the entire urban wastewater system to evaluate the environmental consequences of changing from existing centralized WWTPs to more decentralized systems. These studies concluded that separation systems (i.e. urine, faeces and grey water separation) represent environmental advantages compared to conventional centralized systems, improving the opportunities for nutrient recycling and avoiding their direct release to the environment. These advantages become more evident when the model of the wastewater system is expanded to also include the offset production of fertilizers. This was addressed by Lundin et al. (2000) who demonstrated that if the nutrients in the wastewater were returned to agriculture, the demand for mineral fertilizer in agriculture would be reduced, and the substantial environmental loads imposed by the production and use of mineral fertilizer could be avoided. Also, recovering energy from the organic matter of toilet wastewater and household biowaste in a digestion process can significantly decrease the cumulative energy demand. So, Lundie et al. (2004) expanded the boundaries to include the integrated water and wastewater system in the evaluation of the impact of Sydney total water operations for the year 2021.

The boundaries of the WWTPs have also been expanded to consider the production and distribution of reclaimed water to decrease the dependency on potable and desalinated water. Besides the evaluation of sustainability for water reclamation (Chen et al., 2012), two studies have been applied LCA in that area (Pasqualino et al., 2009, 2011). Both agree that the addition of the tertiary treatment to the traditional WWTP slightly increases the environmental impact of the plant, but this is still considerably smaller than the environmental impact of other water production methods, especially if comparing to desalination.

2.4. Comparison of sludge management strategies

This was first incorporated in LCA studies by Dennison et al. (1998). From then, several studies have been conducted, enlarging the system boundaries, including heavy metals or nitrous oxide (N_2O) emissions, and also evaluating beneficial consequences when energy is recovered from anaerobic digestion processes and nutrients are returned to the environment as soil amendment. The studies available in the literature (Suh and Rousseaux, 2002; Hospido et al., 2005, 2010; Houillon and Jolliet, 2005; Johansson et al., 2008; Hong et al., 2009; Peters and Rowley, 2009; Uggetti et al., 2011; Cao and Pawłowski, 2013; amongst other) compare sludge treatment options inside the WWTPs (anaerobic digestion, thermal process, lime stabilization, silo storage) and sludge management outside the WWTPs (agriculture spreading, incineration, wet oxidation, pyrolysis, landfill, wetland, composting and recycling with cement material). Although the studies are normally case-specific, the conclusions generally indicate that it is better to centralize sludge management and to perform dewatering at the facility in order to decrease potential impacts. Regarding technologies, anaerobic digestion combined with energy recovery is recommended combined with incineration or land application. The latter is restricted by the amount of heavy metals, priority and emerging pollutants because of their potentially significant toxicity effects. Also, the environmental impacts related to the final disposal of sludge by agricultural spreading cannot be neglected.

3. Analysis of the reviewed studies

3.1. Evaluation of LCA practices in the studies reviewed

An in-depth analysis was conducted on the reviewed studies (see Table 1) aiming at identifying the different methodological approaches followed (within the constraints of the ISO standards) and their transparency to communicate the results. Fig. 1 summaries the analysis regarding the proper definition and justification of the goal and scope, the inventory, the impact assessment and the interpretation phases. It can be seen that 100% of the studies defined the goal and scope of the project, covering a wide range of functional units and system boundaries (see the following section). Regarding the inventory, only 38% of the papers provided the inventory data within the paper or as supporting information, making the exercise reproducible (or almost) to others. The impact assessment was addressed in 82% of the studies evaluated. However, 38% of these studies did not explicitly indicate the methodology they used. Finally, only 33% of the studies provided an in depth interpretation of the results including limitations of the methodology and/or performing a sensitivity analysis. Further analysis at each of the ISO levels is provided in the following sections.

3.1.1. Goal and scope definition

Functional unit. The most commonly used functional unit in the reviewed studies is a volume unit of treated wastewater (60% of the papers used volume as m^3 or ML). However, this unit is not always representative, because it does not reflect the influent quality or the removal efficiency of the WWTP. For instance, comparing two systems with different influent loads or with different removal efficiencies might result in misleading conclusions if using volume unit only as the functional unit. In some cases (e.g. Tillman et al., 1998; Gallego et al., 2008), in order to include quality of wastewater besides quantity, the unit population equivalent is used, defined as the organic biodegradable load having a five-day biochemical oxygen demand (BOD_5) of 60 g of oxygen per day. So as to describe the functions of removing both organic matter and nutrients from water Rodriguez-Garcia et al. (2011) propose another definition of the FU expressed in terms of kg PO_4^{3-} eq.

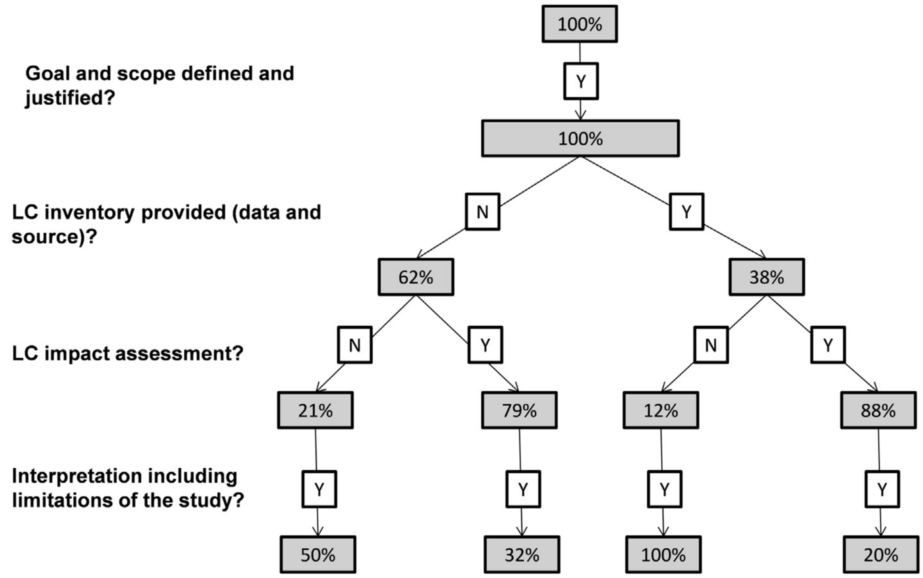

Fig. 1 – Assessment of LCA practices of 45 reviewed studies.

removed. Also, Godin et al. (2012) proposed the net environmental benefit approach which requires assessing the potential impact of releasing wastewater without and with treatment besides assessing the impact of the WWTP's life cycle. On the other hand, only 9% of the studies refer the functional unit to the life-span of the plant. In Emmerson et al. (1995) they assume that the useful life of a typical treatment works, regardless of structural type, is limited to an average of fifteen years and in Larsen et al. (2010) 30 years are used for buildings and construction, 20 years for pipes and valves and 15 years for electronic equipment. In Foley et al. (2010b) they consider 10 years of operation within the functional unit. This is conducted to consider replacement of equipment during the life of the plant.

Boundaries. With regards to the life cycle of the WWT process, 23 of the studies included only the operation of the WWTP and neglected the environmental load of the construction and demolition phases. Among the studies that did include the construction phase, 6 references found out that construction of WWTPs had an impact worth to be considered. Firstly, for low-tech processes (e.g. constructed wetlands, reedbeds) the construction phase can account up to 80% of the impact for some impact categories (Emmerson et al., 1995; Dixon et al., 2003; Vlasopoulos et al., 2006; Machado et al., 2007). Secondly, construction phase was also reported as a relevant stage for conventional activated sludge system and membrane bioreactors, with contributions up to 43% and 31% of the total impact, respectively (Ortiz et al., 2007). Finally, Remy and Jekel (2008) found out that construction affects up to 20% of the total impact for some impact categories. As these are case-specific studies highly depending on the materials used for the construction and the considered lifespan of the infrastructure no generalization is possible. In Frischknecht et al. (2007) they stated that for wastewater treatment capital goods dominate most impact category results, especially because of the sewer infrastructure (also confirmed by the findings of Roux et al. (2011)) and the diluted pollutant content in domestic wastewater. Toxicity related environmental impacts are generally sensitive to the exclusion of capital goods. Hence, capital goods cannot be excluded per se, and a justification would be required when this stage is excluded from the system boundaries.

Complete overviews of the geographical area boundaries were described in Lundin et al. (2000), Lundin and Morrison (2002), and Foley et al. (2010a) including the foreground (emissions and usages directly related with the product/process) and background (the emissions and usages related with the provision of goods or services for the foreground subsystem) sub-systems. Within the foreground sub-systems, nutrient discharges in the aqueous phase were always considered. However, only 53% of the studies included the direct greenhouse gas emissions generated either in the biological treatment, during sludge treatment or after sludge disposal in land fields. All the studies presented the selected boundaries according to the defined objectives, but no strong justification for the selection was normally provided. Fig. 2 shows the boundaries selected for the reviewed studies.

Since the beginning of LCA studies applied to wastewater treatment, sludge treatment and disposal were included in the system boundaries because of the significant contribution to the overall impacts. In fact, this sub-system has been included in 36 of the reviewed studies. The few publications that did not include sludge treatment and disposal were studies comparing non-conventional technologies especially for tertiary treatment that did not generate sludge. Agricultural application was the most common scenario for final disposal (30 papers), which took into account the positive effects of the nutrient value of the sludge and expanded the system to include the avoided production of synthetic fertilizers (i.e. Houillon and Jolliet, 2005) as well as the negative consequences associated with the heavy metals also present in the sludge (i.e. Dennison et al., 1998; Hospido et al., 2004; Pasqualino et al., 2009). One case (Larsen et al., 2010) also

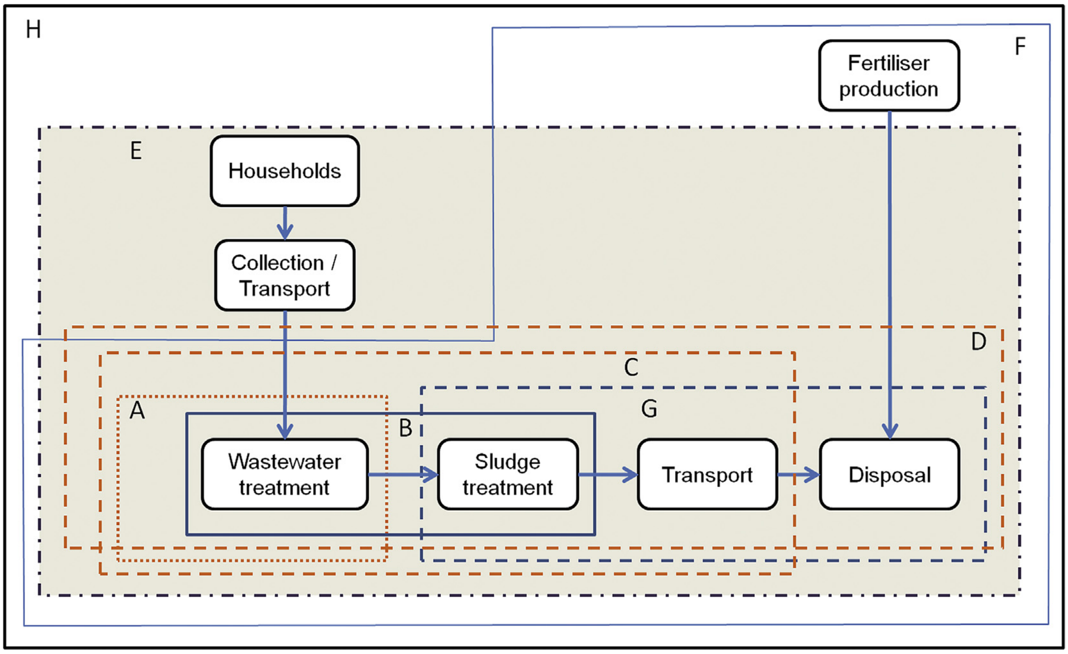

Fig. 2 – Boundaries of the urban wastewater system. Adapted from Lundin et al. (2000). The letters A until H indicate the different system boundaries of the reviewed studies listed in Table 1. Studies with the G boundaries (i.e. only dealing with sewage sludge management) were not included in this review.

included the heavy metal content of mineral fertilizers and the content of some organic pollutants (e.g. DEHP and PAH) in the sludge. However, only 6 studies included GHG emissions from the decomposition of sludge applied to agriculture (i.e. Dennison et al., 1998; Suh and Rousseaux, 2002; Houillon and Jolliet, 2005; Tangsubkul et al., 2005; Gallego et al., 2008; Hospido et al., 2008).

3.1.2. *Inventory*

Within this phase, the studies face problems associated with data availability and data quality. Data for the inventory is collected from lab or pilot facilities as well as real plants, estimation from experts, relevant literature and/or LCA databases. The foreground life cycle inventory (LCI) data is normally compiled directly from measurements, detailed design documents and vendor-supplied information. Background information (e.g. electricity generation systems, concrete and chemicals production processes) is normally provided by LCI databases, e.g. the EcoInvent (www.ecoinvent.ch). From the 22 studies that included the construction stage, original inventory data was used in 68% of them while the others estimated construction loads from other works. In a nutshell, around half of the papers revised do not include inventory data at all (49%), while others just include partial information (18%) and a remaining fraction (33%) do provide the detailed level of data that is desirable in order to reproduce the work.

3.1.3. *LC impact assessment (impact assessment methodology and impact assessment categories)*

According to the ISO standard, the third step of an LCA study is comprised of compulsory (classification and characterization) and voluntary elements (normalization and weighting).

Classification and characterization. Most wastewater LCA studies did move beyond the inventory stage to the impact assessment step. Among the 45 studies revised, 26 stated the impact assessment methodology used: 19 selected CML (Guinée, 2001), 7 EDIP 97 (Wenzel et al., 1997), 3 Eco-indicator 99 (Goedkoop and Spriensma, 2001), 2 Impact 2002+ (Jolliet et al., 2003), 1 EPS (Bengt, 1999), 2 eco-points 97 (Braunschweig et al., 1998) and 1 ReCiPe (Goedkoop et al., 2013). The remaining references did not indicate the method selected or used a mixture of characterization factors.

To the best of our knowledge, Ortiz et al. (2007), Renou et al. (2008) and Hospido et al. (2012) are the only studies that investigated whether the choice of one of the existing LCIA methods, could influence LCA results. In the study of Ortiz et al. (2007) three methods were used for the life cycle impact assessment (CML baseline 2000, Eco-Points 97 and Eco-Indicator 99). Although no specific discussion on that topic was addressed in that paper, the results of Eco-Points 97 and Eco-Indicator 99 were very similar, contrary to the results obtained with CML 2000. The work done by Renou et al. (2008) concluded that for impact categories such as global warming, acidification, eutrophication, or resource depletion, the choice of an impact assessment method is not a critical issue as the results they provide are similar. However, large discrepancies were observed with human toxicity, which has been already reported by Pizzol et al. (2011) who compared nine different methodologies with focus on impacts of metals on human health. Finally, Hospido et al. (2012) compared three impact assessment methods (CML 2000, ReCiPe and IMPACT 2002+) to evaluate the robustness of the environmental ranking obtained for four MBRs. Among the four impact categories evaluated there (i.e. eutrophication, acidification, terrestrial and freshwater ecotoxicities), the main divergences where found for eutrophication potential due to the different significance given by the different impact assessment methods to P-related emissions.

Concerning the set of impact categories evaluated, global warming potential, acidification, and eutrophication are the indicators that received more attention (being evaluated by 38, 27 and 28 out of the 45 papers, respectively). Afterwards, photochemical oxidation (17 studies) and toxicity-related aspects (18 studies dealing with human toxicity, 17 with terrestrial ecotoxicity, 15 with freshwater toxicity, and only 9 with marine ecotoxicity) were the issues of concern. Terrestrial ecotoxicity played an important role when sludge disposal options were evaluated and heavy metals or micropollutants were considered. Finally, ozone layer depletion and abiotic depletion (includes fossil energy and material depletion) were not found to be significant decision-making drivers in these studies, only being assessed by 14 and 20 papers, in that order.

Normalization and weighting. Normalization, which allows comparing all of the environmental impacts on the same scale, was used in 18 of the reviewed studies. Normalization factors were obtained from regional and global databases (e.g. PE, 1990 Denmark; SCB, Sweeden statistics; EU15 world 1994; Western Europe 90s). Weighting, which is used to convert and aggregate indicator results across impact categories into one single indicator, was only applied in 5 studies. The justification is that the process of applying weights depends on subjective value-choices that are more relevant to decision-making processes than elucidation of the relative environmental sustainability of a set of design alternatives. The approaches used in the 5 studies to define weights were the EPS-method (Steen and Ryding, 1992), the Weighted Environmental Theme and the Ecological scarcity (as applied in Baumann, 1993), the hierarchist perspective with average weighting of Eco-Indicator 99, the use of weights provided by CML 2001 methodology, or using the cardinal or ordinal scale by decision-makers based on their preferences or importance for various attributes (Kalbar et al., 2012b).

3.1.4. *Interpretation*

According to ISO 14040:2006, the interpretation should include: a) identification of significant issues based on the results of the LCI and LCIA phases of an LCA; b) evaluation of the study considering completeness, sensitivity and consistency checks; and c) conclusions, limitations and recommendations. Hence, it would be expected that the LCA studies would incorporate a sensitivity analysis to determine which parameters influence the most the LCA outcomes. However, amongst the reviewed studies, sensitivity analysis was only applied in 15 papers.

The communication of the results is a challenging issue since multiple criteria are normally combined with multiple scenarios evaluated. This creates a space of large number of dimensions difficult to explain to the audience. One of the widely used ways of presenting the results is taking a reference scenario for which the impacts are calculated and relate the impacts of the other scenarios to that reference situation. In such a way induced and avoided impacts can be calculated for each scenario. Finally, only 34% of the studies discussed the limitations of the approach and related the recommendations to these limitations.

4. Challenges

The validity of the outcomes from the reviewed studies is restricted to the limitations of the existing practices. Some conclusions might become invalid as research advances (e.g. including new pollutants, finding new factors to estimate the potential impacts, considering local environmental uniqueness, the dynamics of the environment or different time horizons). LCA users are aware of the unresolved problems of this analytical tool (Reap et al., 2008a, 2008b) and the intent of this paper is not to solve them, but to provide a list of challenges for the LCA methodology applied to WWT.

4.1. *Use of LCA to address the change of paradigm in wastewater treatment*

Given the long-term needs for ecological sustainability, the goals for WWT systems need to move beyond the protection of human health and surface waters to also minimizing the loss of resources, reducing the use of energy and water, reducing waste generation, and enabling the recycling of nutrients. There is a change of paradigm, from waste to resource recovery and water reuse which can be properly addressed by using LCA at the research stages of new technologies or at full-scale when brought into practice.

4.2. *Adaptation of LCA methodologies to new target compounds*

The developments in toxicity-related impact categories mainly relate to heavy metals and priority pollutants (PPs) (Muñoz et al., 2008; Larsen et al., 2009). Although there is a severe deficiency in our understanding of the human health implications of PPs (Novak et al., 2011), LCA methodologies are being updated to include the effect of PPs on ecotoxicity (Muñoz et al., 2008; Larsen et al., 2010; Alfonsín et al., 2012; Morais et al., 2013). Research is needed to determine more realistic factors for heavy metals discharged in the soil. The actual values related to their persistence in the environment are very high, and more research is required in order to establish more precisely the amount (bioavailable part) of heavy metals that is effectively taken up by plants and crops as well as the amount that is transferred to another phase such as leachate (Hospido et al., 2005). Moreover, organic micropollutants are also included in the most recent studies. For sludge disposal, heavy metals are still dominant compared to organic micropollutants (Hospido et al., 2010). This is also the case for the effluent of WWTPs, where micropollutants significantly contribute to aquatic ecotoxicity (Larsen et al., 2010). Regarding ecotoxicity and human toxicity the best practice methodology for the moment is probably USEtox (Rosenbaum et al., 2008), a consensus model which was developed by a group of LCIA method developers as part of the UNEP/SETAC Life Cycle Initiative. USEtox considers all metals (except Ni) to be more dangerous for human health when released to soil than when released to water, due to the fact that heavy metals in soil are more easily transferred to crops and from there to humans, either directly or through meat and milk, than when they are released to freshwater.

Complete LCA studies assessing the fate of micropollutants not only in wastewater, but also in excess sludge and sludge treatment would contribute to better understand their environmental implications.

4.3. Evaluating management practices at regional scale incorporating spatially differentiated factors

In the real world of environmental approvals, it is absolutely necessary to understand what impact the WWTP effluent will have on the receiving environment. Location-specific factors are critical, especially for the eutrophication impact category. Renou et al. (2008) discusses that eutrophication is correctly estimated if one looks at the potential impact of a treatment scenario but not at the characterization of the eutrophication state of a specific receiving stream. LCIA eutrophication factors available in databases cannot deal with the specifics of particular locations. There is a trend to incorporate spatially differentiated factors to estimate eutrophication impacts (Basset-Mens et al., 2006; Gallego et al., 2010) and to develop new methods on both spatially differentiated freshwater and marine eutrophication (mid-point and end-point) (e.g. the work of the EU research project LC-Impact, www.lc-impact.eu). The challenge here is to provide a set of "accepted" characterization factors that can be applied at regional scale.

4.4. Improving data quality and reducing uncertainty

Other challenges relate to improving data quality and availability for LCA studies in WWT. The inventory phase is normally conducted by using a mixture of experimental or full-scale data and existing databases. The goal of the study determines the accuracy required for the inventory data, and indicates where the efforts should be made in data collection. The inventory phase is crucial and should be accurately designed as for other model-based approaches (e.g. for activated sludge mechanistic model calibration following the methodology described in Rieger et al. (2013) closing mass balances for the evaluated compounds). It is crucial to identify critical aspects in the wastewater treatment sector that might influence significantly the LCA results. For instance, non-biogenic, direct gaseous emissions emitted during the secondary treatment are rarely considered in the LCA studies. In the recent years there is more concern about the greenhouse gas emissions from WWTPs, with special focus on N_2O and CH_4 (Foley et al., 2010a; Larsen et al., 2010; Corominas et al., 2012; Rodriguez-Garcia et al., 2012) which can contribute significantly to the overall GHG emissions from WWTPs. However, there is not yet consensus within the scientific community on the mechanisms behind the production of N_2O.

Mechanisms for sharing models/data and proper supplemental information in scientific publications is required to ensure high quality and comparability of studies. One of the systems is to submit in public channels (including a peer-review process) the outcomes from the studies, such as the ELCD (http://lca.jrc.ec.europa.eu/lcainfohub/datasetArea.vm) or the upcoming ILCD platforms (http://lct.jrc.ec.europa.eu/pdf-directory/ILCD-DN.pdf), as well as the inclusion of the detailed inventory as supporting information in the published papers.

4.5. Stakeholder participation for integration of obtained results in decision-making

An effort should be made to achieve wider acceptance of LCA results amongst decision-makers through continuous stakeholder participation (Guest et al., 2009), so that they provide greater value to the decision-making process. Not only is communicating the outcomes of LCA studies a difficult task (as mentioned before), but explaining the environmental processes and mechanisms on which the LCA methodology relies is particularly challenging given that they are highly complex and interactive and the models that describe them rely on assumptions that remain hidden in databases. Engaging utility personnel early in the process may achieve greater buy-in among decision-makers as to the validity of the LCA and its underlying assumptions. Additionally, the use of end-point impact categories may be perceived to be more relevant by stakeholders (Bare et al., 2000), even though the use of end-point indicators relies on additional assumptions and introduces greater uncertainty in the modelling process compared to mid-point indicators. Finally, LCA methodology should be linked to economical (Life Cycle Costing – LCC) and social (e.g., Social Life Cycle Assessment -SLCA) evaluations completing the whole picture of sustainability (e.g., Life Cycle Sustainability Assessment -LCSA) (Kloepffer, 2008).

The challenges described in the paper indicate that there is still room for improvement and a joint effort between the WWT community and the LCA community has to be made to address sustainable issues in the forthcoming years. In this sense, the "Working Group for Life Cycle Assessment of Water and Wastewater Treatment, LCA-Water WG" has been created under the umbrella of the International Water Association (IWA) with the aim of facilitating the exchange of ideas, and to develop consensual methodologies to promote better use of LCA in the urban water systems (more information: http://www.iwahq.org/1y3/networks/specialist-groups/list-of-groups/modelling-and-integrated-assessment/workinggroup-lca.html).

5. Conclusions

LCA applied to wastewater treatment is a field with 17 years of experience. Since 1995, 45 international peer-reviewed papers dealing with WWT and LCA have been published. The analysis of these papers has shown that within the constraints of the ISO standards, there is variability in the definition of the functional unit and the system boundaries, the selection of the impact assessment methodology and the procedure followed for interpreting the results. Hence, there is need to develop standardized guidelines for the wastewater treatment field in order to ensure the quality of the application of the LCA methodology.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.watres.2013.06.049.

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