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Life cycle assessment of wastewater treatment in developing countries: A review

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Abstract

Within developing countries, wastewater treatment (WWT) has improved in recent years in but remains a high priority sustainability challenge. Accordingly, life cycle assessment (LCA) studies have recently started to analyse the environmental impacts of WWT technologies on the specific context of less developed countries, mainly in China and India. This work presents a comprehensive review of this knowledge with the aim of critically analysing the main conclusions, gaps and challenges for future WWT-related LCAs in developing countries. The most commonly assessed technologies in the 43 reviewed articles are different variations of activated sludge and extensive treatments applied in decentralized systems; however, studies focused on advanced technologies or new sources of pollution (e.g. micropollutants) are still lacking. Goal and system boundaries are normally clearly defined, but significant stages for some technologies such as the construction and sludge management are frequently not included and functional units should be defined accordingly to specific conditions in developing countries. At the inventory level, a more concise description of sources and technical parameters would greatly improve the quality of the LCAs along with accountability of direct greenhouse gas emissions. Eutrophication and global warming are the two

most commonly assessed impacts; however, the calculation of terrestrial ecotoxicity when the sludge is used for agricultural purposes, of water use and of the land use change impacts associated to extensive technologies should be encouraged. The estimation of more site-specific databases, characterization factors (especially for eutrophication) or normalization and weighting values combined with more affordable access to background databases and LCA software, would deeply increase the accuracy of WWT-related LCAs in developing countries. An increased usage of the uncertainty analysis should be encouraged to assess the influence of these gaps in the final interpretation of the results. The review finishes with a summary of the main challenges and research gaps identified and with specific guidelines for future researchers to avoid the most common shortcomings found in the reviewed studies.

Keywords: *LCA; eutrophication; sanitation; environmental impacts, sludge; sustainability*

1. Introduction

Wastewater management has been a significant part of civilizations throughout the millennia, consequently evidencing many sociological aspects and technological advances throughout the ages (Lofrano and Brown, 2010). By the mid-19th century, Europe had engineered their first modern wastewater treatment (WWT), reaching an established status by the end of the last century. This, combined with a working regulatory framework and economic growth, led to consistent improvements in environmental quality and public health within the continent (Angelakis and Snyder, 2015; Shifrin, 2005). Most developed countries have well-functioning wastewater collection and treatments infrastructure (covering over 91% of their population) (UNICEF and WHO, 2017); nowadays concerns are moving towards the presence of emerging contaminants (Gavrilescu et al., 2014; Archer et al., 2017; Tiedeken et al., 2017), with measures already under way to diminish their presence in the environment (Bui et al., 2016; Barbosa et al., 2016). However, the WWT situation varies significantly in developing countries around the globe.

Although improvements have been made in the last decades, with over 2.1 billion people gaining access to improved sanitation since 1990, obstacles remain to reach the necessary level of WWT worldwide (UNDP, 2016). Around 2.5 billion people still lack sufficient sanitation services, which combined with the rapid urbanization, population growth and substandard infrastructure, promotes the spread of diseases and unbridled release of nutrients, thereby generating significant health and environmental concerns (UNICEF and WHO, 2017). Therefore, to achieve the United Nations (UN) sustainable development goal 6 “Clean Water and Sanitation” by 2030, a considerable effort in the construction and operation of wastewater treatment plants (WWTPs) should be applied in these underdeveloped nations in the following years (UN-Water, 2017a).

In order to select the technology that should be applied in each case, compliance with stipulated environmental regulatory standards and technology costs should be considered alongside other aspects like geographic location, socioeconomic conditions or regional and global environmental impacts (Marlow et al., 2013; Ren and Liang, 2017; Arroyo and Molinos-Senante, 2018). Regarding the latter, the life cycle assessment (LCA) methodology is often used to evaluate environmental impacts of products and services, considering a set of energy and material inputs and outputs throughout their life cycle, allowing the quantification of the effects of the entire system under study (e.g. climate change and resource depletion potentials), and not only aspects related to the facility surroundings (ISO 14040:2006, 2006). However, LCAs are seldom considered during the design and evaluation of WWTs in developing countries (Ahmed, 2010), although LCA methodology enables better decision making due to inclusion of more variables (Flores-Alsina et al., 2010; Larrey-Lassalle et al., 2017). Therefore, it is necessary to incorporate the analysis of reliable indicators from LCA and to review previous studies in the specific context of developing countries to acknowledge the main lessons learned and gaps detected.

The review of Corominas et al. (2013) showed that several LCAs of WWTs in many developed countries have been published since 1995, but the adoption of this tool in developing

countries is much more recent. In fact, from the 45 studies analysed by Corominas et al. (2013), only one paper was focused on a developing country and was from the same year the review was published (Kalbar et al., 2013). Recently, Zang et al. (2015) and Hernández-Padilla et al. (2017) reviewed 53 and 46 WWT-related LCAs; again, only seven and two studies, respectively, were in developing countries. The absence of specific focus on developing countries and the lack of studies until recent years are behind this low representation. To the best knowledge of the authors, no review has analysed the common conclusions of the numerous WWT-related LCAs focused on developing countries in recent years and the major challenges for the future in this field.

In an effort to provide an overview of the situation of LCA in the WWT sector in developing countries, this paper proceeds as follows. Firstly, there is a review of current LCAs on available WWTs in developing countries, critically commenting on the technical parameters and technologies assessed in the literature. Secondly, the development of each LCA step (goal and scope, inventory, impact assessment methodologies and interpretation) is analysed in detail. The main findings of these papers are discussed, and, finally, this review concludes with the main challenges and gaps detected, assessing in detail those of specific significance for developing countries, and proposes recommendations and future research lines to improve LCA practice in developing countries.

2. Review of LCA studies on WWT in developing countries

When selecting the review studies, the following keywords were initially examined in Scopus: (life AND cycle AND assessment OR LCA) AND (wastewater AND treatment). From the total of 703 results (April 2018), those 205 with authors affiliated in developing countries (according to OECD (2018)) were selected. The same process was then applied to the studies that either were cited or have cited those 205 articles. Those searches were complemented with the results of searching: (life AND cycle AND assessment OR LCA) AND (wastewater AND treatment) AND the main developing countries by population (Mexico, Brazil, China, India, Thailand, Egypt, South

Africa, Algeria and Turkey). A similar process was consequently followed in Science Direct, Google, Google Scholar, and Web of Science. After reading the abstract of the preselected studies, 144 were analysed in depth. Finally, the 43 articles presented in Table 1 have been selected because the life cycle assessment of wastewater treatment was the main goal of the research and the location of the treatment is within developing countries. Studies that considered other aspects of the water cycle (e.g. water supply or wastewater collection) but had a special focus on the treatment of the wastewater have been included as well. In accordance with previous reviews (Corominas et al. 2013; Zang et al. 2015), those studies focusing exclusively on the sludge treatment and disposal without considering the treatment of the water were not considered as they related more to the waste field than to the wastewater treatment sphere.

The OECD (2018) classifies developing countries in relation with the level of income in four groups: i) least developed countries; ii) other low-income countries; iii) lower middle-income countries; and iv) upper middle-income countries. It is noticeable that none of the reviewed studies are located in the 49 countries included on the two first groups with lower incomes and more than 50% are concentrated in just two Asian countries (India and China). The UN sustainable development goal 6.3 aims to half the proportion of untreated wastewater in 2030 compared with the levels of 2015 (UN-Water, 2017b). This challenge implies a special effort in constructing and operating WWTPs in least and low-income countries (92% of the wastewater was untreated in 2015) although the task is also significant in lower middle-income countries and upper middle-income countries (72% and 62% on the water was untreated in 2015, respectively) (UN-Water, 2017a). Hence, it is necessary to geographically diversify to include studies focussed on the poorest countries and their specific characteristics, evaluating from the environmental perspective the expected development of new WWTPs in the following years. Although further implementation is necessary, the general improvement of WWT in developing countries has already been significant in the last 15 years, particularly in Asia (WHO and UNICEF, 2017). Accordingly, the implementation of WWT-

associated LCAs in developing countries has grown in parallel and exponentially in recent years (77% of the papers in Table 1 have been published in the last 5 years). However, considering the above-mentioned potential growth in WWT in the near future and the lack of geographical diversity, it should be expected that the LCA research into WWT in developing countries will continue to grow in the following years.

Table 1 - Articles included in the review and main characteristics.

Reference	Country	Technology ^a	Functional unit ^b	System boundaries ^c	Impact methodology ^d
Zhang and Wilson (2000)	China	CT	1 GJ	D	Consumed energy
Pillay (2006)	South Africa	CT; AT	1 m ³	E +WR	CML
Mahgoub et al. (2010)	Egypt	CT	1 m ³	E	Eco-indicator 99*
Zhang et al (2010)	China	CT	1 kJ	D	Consumed energy
García Alarcón et al (2011)	Mexico	CDT	1 m ³	A	TRACI
Hong et al (2011)	China	NCT	10,000 m ³	D	CML
Pan et al (2011)	China	CDT; DEC	CO ₂ eq. /kg BOD rem.; 1 p.e.	E +WR	IPCC
Fernández-Torres et al (2012)	South Africa	NCT	1,000 t/d	A	Impact 2002+
Roushdi et al. (2012)	Egypt	CT; DEC	1 m ³	E +WR	Eco-Indicator 99*
Wang et al. (2012)	China	CT	105 m ³ /d 20 yr	F	IPCC
Cornejo et al (2013)	Bolivia	CDT; DEC	1 m ³ for 20 yr	E +WR	IPCC, CED,Eco-indicator95
Kalbar et al. (2013)	India	CT; DEC	1 p.e. per yr	D	CML
Li et al (2013)	China	CT; DEC	105 m ³ /d 50 yr	D	CML
Ontiveros and Campanella (2013)	Argentina	CT	2,000 m ³ /d; 9,259 p.e.	D	CML
Tong et al. (2013)	China	CT: NCT	1 m ³	D	CML
Zhu et al (2013)	China	CT	kg PO ₄ eq. removed	F	CML
Kalbar et al. (2014)	India	CT	1 p.e. yr	B	CML
Risch et al. (2014)	Egypt	CT; DEC	1 p.e.	G	Recipe*
Lam et al (2015)	China	CDT; DEC	WW p.e. per yr ^f	D +WR	LIME-2*
Meneses-Jácome et al (2015)	Colombia	NCT	1 m ³ biogas	D	Impact 2002+
Vera et al. (2015)	China	NCT	3,000 m ³ /d and yr	B	CML
Wang et al. (2015)	China	CT	10,000 m ³ /d; 25,600 p.e.	D + WR	CML
García-Montoya et al (2016)	Mexico	CDT	1.p.e per yr	E	Impact 2002+
Fajardo et al (2016)	Colombia	NCT	1 kg/s	D	EDP 2013
Limphitakphong et al. (2016)	Thailand	CT	1 m ³	C	LIME*
Singh et al. (2016)	India	CT	1 m ³	C	IPCC, CED
Casas Ledón et al (2017)	Chile	DEC	700 p.e.; CO ₂ eq. /kg BOD removed	D	IPCC
De Oliveira Schwaickhardt et al (2017)	Brazil	AT	1 m ³ / 3 h	A	ReCiPe
Diniz et al (2017)	Brazil	DEC	1 kg biomass	A	ReCiPe*
Hernández-Padilla et al. (2017)	LAC ^e	CT; DEC	1 m ³	D	Impact World+, 2002+, ReCiPe
Kamble et al. (2017)	India	DEC	1 m ³	D +WR	CML
Kulak et al. (2017)	India	CT; DEC	5.3 p.e.	G	ReCiPe; IPCC
Laitinen et al (2017)	Mexico	CDT; DEC	1,000 m ³	F +WR	IPCC

Lu et al. (2017)	China	CT	m ³ in last 28 d	D	Eco-indicator 99*
Lutterbeck et al (2017)	Brazil	AT; DEC	333.9 t 10 yr	A +WR	ReCiPe*
Palomares-Rodríguez et al (2017)	Mexico	CDT	180 m ³ /d	A	TRACI
Raghuvanshi et al. (2017)	India	CDT	1,500 m ³ /d 50 yr	D +WR	ReCiPe*
Singh et al. (2017)	India	CDT	1 m ³	D	CML
Zhao et al (2017)	China	DEC	100 L	A	CML and E-balance
Polruang et al. (2018)	Thailand	CT	1 m ³	F +WR	CML
Singh and Kansal (2018)	India	CT	1 m ³	E	IPCC, CED
Singh et al. (2018)	India	CT	1 m ³	D +WR	CML
Wang et al. (2018)	China	AT	1 t COD eq. removed	A	CML

^a CT: Evaluation of conventional technologies; NCT: Evaluation of non-conventional technologies; CDT: Evaluation of conventional technologies in decentralized systems; DEC: Evaluation of extensive technologies in decentralized systems; AT: Evaluation of advanced technologies.

^b p.e. = person equivalent.

^c A = Wastewater treatment; B = Wastewater and sludge treatment; C = B + sludge transport; D = C + disposal of sludge; E = D +household wastewater collection and transport; F = D + fertilizers production; G = E + fertilizers production; WR = water effluent reuse.

^d The * indicated endpoint methodologies. CED = cumulative energy demand.

^e LAC: Latin American and the Caribbean countries. Results for 22 LAC countries obtained by applying national electricity mix and characterization factors to data of an average of the wastewater treatment plants of the region.

^f Wastewater (urine, faeces, and grey water) discharged annually by one person.

3. Technical parameters

3.1. Type of wastewaters

The most common type of wastewater in the reviewed articles is urban effluent (representing 26 from the 43 studies, 60% of the total, refer to Figure 1 and Table S1 in supporting information (SI)), related to combined discharges to sewage systems from households, septic tanks, industries, commerce and storm water. The domestic or residential wastewaters were specifically considered in four occasions (Zhang et al., 2010; Risch et al., 2014; García-Montoya et al., 2016; Kamble et al. 2017) and from rural locations in another four (Cornejo et al. 2013; Lam et al. 2015; Casas Ledón et al. 2017; Lutterbeck et al. 2017). For developed countries, the review of Corominas et al. (2013) found that the majority of the LCAs in developed countries considered urban or domestic wastewaters in their studies (over 90% of the articles compared to 70% in this review).

The evaluation of industrial wastewaters can be found in seven papers, covering a variety of processes. For example, Hong et al. (2011) and Vera et al. (2015) analysed, respectively, the treatment of pulp and paper and corn starch processing wastewater in China; Fernández-Torres et al. (2012) the case of saline wastewaters from mining industry in South Africa; the treatment of slaughterhouse wastewaters were evaluated in Mexico and Colombia (Meneses-Jácome et al., 2015; Fajardo et al., 2016). Finally, the treatment of hospital wastewater was carried out in Brazil by De Oliveira Schwaickhardt et al. (2017) and of a university campus in India by Raghuvanshi et al. (2017).

Figure 1 - Types of wastewaters found in the reviewed LCAs of wastewater treatments in developing countries

3.2. Influent wastewater quality and pollutants removal

The biological oxygen demand (BOD) or chemical oxygen demand (COD) of the influents were stated in 30 papers (70%; Table S1 in SI). The BOD of urban wastewaters averaged at 202 mg/L. Rural wastewaters showed a higher average BOD, of 536 mg/L, possibly due to lower dilution. The average removal of BOD from urban wastewaters by the treatments in this review is 81% and from rural wastewaters it is 85%. In relation COD, followed similar variabilities, where urban wastewaters averaged at 408 mg/L (mean removal by the treatments of 78%), rural community 797 mg/L (mean removal 82%) and industrial wastewaters 8,469 mg/L (mean removal 78%).

The total suspended solids (TSS) was covered in 18 papers, with an influent average value for this parameter in the treatments of 195 mg/L from urban wastewaters (mean removal 86%) and 1,429 mg/L in industrial wastewaters (mean removal 70%). Two parameters commonly measured during wastewater treatments, total phosphorus (TP) and total nitrogen (TN), were shown in 21 papers. For the former, its content reached concentrations in urban wastewaters as high as 27.0 mg/L, averaging at 9.6 mg/L (mean removal 27%). In relation to the TN, its content in urban wastewaters averaged at 41.3 mg/L (mean removal 42%). Other common wastewater parameters of the influent (temperature, pathogens and metals) were seldom displayed in the reviewed papers. The presence of

micropollutants such as pharmaceutical and personal care products (PPCPs) in the influent wastewaters have not been a subject of interest in any of the reviewed papers even though found at high concentrations in WWTPs and freshwaters in these regions (Liu et al. 2013; Wu et al., 2014; Ebele et al., 2017).

Figure 2 compares the above-mentioned parameters from urban wastewaters in developing countries with those from developed countries analysed in Corominas et al. (2013). Figure 2 shows a higher influent BOD, COD and SS (averaging in 251, 551 and 252 mg/L respectively) in developed countries. However, their effluent concentration is considerably lower, resulting in average higher removal efficiency in developed countries. For instance, the average removal of BOD in developing countries (81%) is significantly lower than in developed countries (94%). Although TP and TN showed similar influent concentrations in both cases, their removal was found to be higher in developed countries. The average removal in developed countries was 75% for TP and 72% for TN and these figures are nearly two to three times higher than in developing countries. These differences can be justified by stricter discharge requirements and the most common use of modified activated sludge and chemically enhanced treatments in developed countries.

Figure 2 - Comparison of the biological oxygen demand (BOD), chemical oxygen demand (COD), suspended solids (SS), total phosphorus (TP) and total nitrogen (TN) concentrations in the influents (inf) and effluents (eff) of the life cycle assessments of urban wastewater treatments in (a) developing countries and (b) developed countries.

3.3. Treatments size and lifespan

Excluding bench, household and pilot-scales ($<0.6 \text{ m}^3/\text{d}$ or approximately 5 p.e.) and unspecified capacities, the treatments were evaluated in 34 papers for influent flows from $24 \text{ m}^3/\text{d}$ up to $567,000 \text{ m}^3/\text{d}$. The largest treatments ($>300,000 \text{ m}^3/\text{d}$) are in Egypt, Thailand and India (Mahgoub et al., 2010; Roushdi et al., 2012; Limphitakphong et al., 2016; Polruang et al., 2018; Singh and Kansal, 2018). The treatment of industrial wastewaters was evaluated from influent flows varying from $112 \text{ m}^3/\text{d}$ to $35,000 \text{ m}^3/\text{d}$. The lifespan of the treatments was defined in 18 papers. For larger centralized plants, common assumptions for infrastructure endurance were 50 years (Li et al., 2013; Singh et al., 2016; Laitinen et al., 2017; Singh and Kansal, 2018). The assumption for wastewater treatments located in rural communities were shorter, from 10 to 20 years (Cornejo et al., 2013; Lam et al., 2015; Lutterbeck et al., 2017; Casas Ledón et al., 2017).

3.4. Treatment technologies

Most of the articles (74%) evaluated secondary treatments biological activated sludge (AS) processes and its variations; it is also the most common practice by LCA practitioners in developed countries, as shown in Corominas et al. (2013). The most common variants found were the anaerobic-anoxic-oxic (A2O), Bardenpho 5-stages, membrane bioreactor (MBR) and sequencing batch reactors (SBR).

An extensive collection system to transport wastewater is more than often coupled to large centralized facilities where AS treatment is applied. However, this treatment strategy has drawbacks such as a significant demand of resources for the construction of the wastewater collection systems and high energy and maintenance requirements. This would potentially result in substantially higher costs for lower density districts or locations with uncertainty related to population growth (Massoud et al., 2009; Roefs et al., 2017; Jung et al., 2018). In this sense, extensive wastewater treatments in

decentralized systems (e.g. constructed wetlands, septic tanks, ponds, latrines) provide a feasible alternative to complement or even substitute centralized wastewater treatment strategies due to its relative reliability, simplicity and efficiency. This is particularly true for developing countries in tropical regions (Kivaisi, 2001; Massoud et al., 2009).

In this sense, decentralized strategy was found to be the primary topic in sixteen papers for the treatment of urban, domestic, campus and residential wastewaters (Table 1). The decentralized wastewater systems in rural areas was the main topic in four other articles. The most prevalent treatment were CWs, either as a single vertical-flow (VFCW) (Lam et al., 2015) or horizontal subsurface-flow (HSSF) (Casas Ledón et al., 2017), and after a UASB followed by biological filter (Lutterbeck et al., 2017). The UASB followed by ponds was evaluated by Cornejo et al. (2013) in rural Bolivia.

Non-conventional treatments were assessed for industrial wastewaters. Flotation followed by ponds was evaluated by Hong et al. (2011) for pulp and paper wastewater treatment. In South Africa, Fernández-Torres et al. (2012) studied eutectic freeze crystallisation and evaporative crystallisation for high salinity wastewater treatment from mining activities. For the treatment of chemical, grain, and oil industries wastewaters, Tong et al. (2013) evaluated modified AS process in combination with continuous membrane filtration (CMF), reverse osmosis (RO) and electrodeionization (EDI). Meneses-Jácome et al. (2015) considered expanded granular sludge bed (EGSB) for poultry viscera wastewater treatment. Vera et al. (2015) studied corn starch wastewater treatment by fermentation, flocculation, filtration, drying and aeration-clarifier. For slaughterhouses, Fajardo et al. (2016) and Palomares-Rodríguez et al. (2017) evaluated modified AS and UASB processes.

The application of advanced treatments technologies was presented in four studies. They covered the use of ultra-violet (UV) (De Oliveira Schwaickhardt et al., 2017; Lutterbeck et al., 2017; Wang et al., 2018), ozonation (Pillay 2006; De Oliveira Schwaickhardt et al., 2017), and granular activated carbon (GAC) (Pillay, 2006). The use of advanced oxidation processes (AOPs) for

wastewater treatment have been assessed through LCA in developed countries, but not in developing countries, even though showing potential for application in decentralized treatments (Gallego-Schmid et al., 2019; Chong et al., 2012). Finally, the use of multi-effect distillation in developing countries to treat natural brackish water with a high boron and arsenic content has been recently assessed with the LCA methodology (Tarpani et al., 2019).

The reuse of treated wastewater was a matter of concern in 11 papers. Pillay (2006) studied the reuse in industries of Durban (South Africa) and Polruang et al. (2018) for irrigation in Thailand. Lutterbeck et al. (2017), Cornejo et al. (2013), Pan et al. (2011) and Lam et al. (2015) studied WWTs to allow wastewater reuse for different purposes in rural communities of Brazil, Bolivia and China. Laitinen et al. (2017) provided the evaluation of treatments for wastewater reuse in irrigation and García-Montoya et al. (2016) in a residential complex in Mexico. In India, Singh et al. (2018) and Kamble et al. (2017) considered the reuse as a tap water replacement and Raghuvanshi et al. (2017) for irrigation. Finally, Roushdi et al. (2012) studied CWs to enable effluents with sufficient quality for industrial or less demanding uses in Egypt.

4. Life cycle phases

The development of a full LCA implies four main phases according to ISO standards (ISO 14040:2006, 2006; ISO 14044:2006, 2006): goal and scope definition, inventory analysis, impact assessment and interpretation. The level of coverage of several aspects of these phases are presented in Table 1 and Figure 3 and discussed in the following sections.

Figure 3 - Life cycle assessment phase coverage in the 43 studies reviewed

4.1. Goal and scope definition

The goal and scope definition are the initial phase of an LCA. In this stage, the intentions of the study are justified, the functional unit (FU) defined, and the system boundaries described. All the reviewed papers have provided reasoning for their research, but few pursued an assessment of the whole life cycle of the WWT, with only three including a “cradle-to-grave” approach, i.e. building, operation and decommissioning of the treatment (Figure 3). The FU and system boundaries are reviewed next.

4.1.1. Functional unit (FU)

The FU defines the quantification of the identified functions (performance characteristics) of the product. The primary purpose of a FU is to provide a reference in which the inputs and outputs

are related. This reference is necessary to ensure comparability of LCA results; this is particularly critical when different systems are being assessed, and to ensure that comparisons are made on a common basis (ISO 14040:2006, 2006).

Most of the reviewed papers (29 out of 43 in Table 1) considered the volume treated as the FU, in line with the tendency observed in the literature (Corominas et al., 2013). However, several authors have stated that the FUs based in volume do not reflect the removal efficiency of the WWTPs or the initial quality of the influent (Gallego et al., 2008; Wang et al., 2018). To overcome this problem, population or person equivalent (p.e.), is commonly used as an alternative FU. However, the definition of p.e. should be clear. Several articles from Table 1 consider p.e. as FU but calculated in very different forms, and therefore, they are not directly comparable. In LCA articles (e.g. Gallego et al., 2008; Risch et al., 2014; Tillman et al., 1998), p.e. is commonly considered equal to an organic load of 60 g of oxygen per day of five-day biological oxygen demand (BOD₅). However, according to Mara (2004), 60 g BOD₅ is representative value for developed countries and a more accurate value to be used in developing countries is 40 g BOD₅. Similarly, Kalbar et al. (2013, 2014) considered 50 g BOD₅/p.e. as more representative of India for their LCA study. Other studies consider p.e. as FU from different approaches like p.e./household (Kulak et al., 2017), the urine, faeces and grey water discharged annually by p.e. (Lam et al. (2015) or volume of water consumed per p.e. (Casas Ledón et al., 2017). Again, these data must be specifically considered for developing countries, because the amount of p.e./household or water volume consumed per p.e. varies considerably from developed to developing countries. Other FUs focused on the removal efficiency of the treatment (e.g. COD eq. (Wang et al., 2018), PO₃⁴⁻ eq. (Zhu et al., 2013) or BOD (Pan et al, 2011; Casas Ledón et al., 2017)). Finally, other studies which relate the impacts to the amount of by-product produced like kg of algae biomass (Diniz et al, 2017) or m³ biogas (Meneses-Jácome et al., 2015). Zang et al. (2015) suggested that WWT-related LCA studies should analyse their results using more than one FU to clarify the system under study and avoid misunderstandings

in the conclusions. Some studies in developing countries (e.g. Ontiveros and Campanella 2013, Wang et al., 2018) express their FU in volume and p.e. but an analysis of the influence of the choice is lacking in the interpretation of the results.

4.1.2. System boundaries

As expected, all studies include the operation of the WWT as part of the system boundaries (Table 1). However, only 14 of the 43 studies considered the construction stage and only three considered the final decommissions (see Table S2 in SI). Most of the studies do not show a significant environmental contribution from the construction stage, except for the indicator of cumulative energy demand or when extensive treatment technologies with high amount of materials transported for construction and low operational consumption (e.g. constructed wetlands or pond systems) are evaluated. These extensive treatment technologies can play a crucial role in developing countries in the future due to their low operational cost and easy maintenance, particularly in rural areas where sanitation conditions are usually poor and there are less constraints associated to land occupation (Yildirim and Topkaya, 2012; Lam et al., 2015).

Regarding the reviewed articles, Kable et al. (2017) studied the purification of water with different layers of soil filters in a bioreactor (“soil biotechnology treatment”) and concluded that the construction phase contributed to all the environmental impacts more than the operation phase of the plant with the only exception of eutrophication. Cornejo et al (2013) concluded that the construction stage accounted for >90% of the embodied energy in two water systems in rural areas of Bolivia with combinations of an UASB reactor and constructed ponds treatments due to the low electricity and material consumption to operate and maintain these natural systems. Hernández-Padilla et al. (2017) concluded that CO₂ from the cement production used to construct pond systems was the most important contributor to long-term global warming. Finally, Lutterbeck et al. (2017) estimated that 67% of the environmental impacts of a wastewater treatment system composed of an UASB

combined with an anaerobic filter, subsurface constructed wetlands and ultraviolet disinfection reactors were related to the construction stage. However, the authors declared that the initial consideration of 10 years as lifespan of the facilities is conservative and could be expanded and, therefore, reduce the relative impacts associated to the construction when compared to the operation. Although the importance of the construction stage for constructed wetlands has been highlighted in previous studies (Dixon et al., 2003; Machado et al., 2007), several papers from Table 1 do not consider the construction stage when analysing the impacts of constructed wetlands or ponds (Pan et al., 2011; Roushdi et al., 2012; Kalbar et al., 2013; Casas Ledón et al., 2017; Laitinen et al., 2017).

Sludge treatment and disposal is normally included amongst LCA wastewater treatment system boundaries due to its significant contribution to overall impacts (Corominas et al. 2013). The studies which did not include this stage are normally non-conventional technologies that do not produce sludge. The WWT-related LCAs in developing countries follow the same pattern, with 71% (31 out 43) of the papers including this stage, being the landfilling and agricultural application the most common final disposal options (16 and 15 papers, respectively; Table S2 in SI). However, other common LCA practices associated to this stage were not so frequent in studies from developing countries. For example, only six of the 15 studies with agricultural application expanded the system boundaries and included the positive effects of the nutrient value of the sludge by considering the substitution of chemical fertilizers. The reason behind this low ratio remains unclear, but the lack of data about the nitrogen, phosphorus and potassium content in the sludge or access to chemical fertilizer background data could be the potential explanation. On the other hand, several studies have confirmed that the land application of WWTP sludge containing heavy metals can lead to negative toxicity effects (Sharma et al., 2017; Yoshida et al., 2018), as wastewater-derived fertiliser has been shown to have substantially larger heavy metal loads compared to synthetic fertilisers (Foley et al., 2010a). From the 15 studies applying sludge for agricultural purposes, only 3 (21%) measure the emission of heavy metals to the soil (Table S2 in SI). The complexity and relative high cost of these

analyses (EMSL, 2013) make them less common in developing countries. Accordingly, Mahgoub et al. (2010) and Roushdi et al. (2012) used the literature data from Hospido et al. (2005) to model the impacts of the sludge composting and land application. However, these data can only be considered a rough approximation as they are based on the heavy metal soil emissions for the Spanish WWTP studied by Hospido et al. (2005), with different original conditions. Moreover, the final emission of heavy metals to the soil depend on the sludge quality (Niero et al, 2014) and are site-dependent, because previous condition of the agricultural soil where the sludge is applied should be considered as well (Tsadilas et al., 2005; Wang et al. 2008).

4.2. Life cycle inventory (LCI)

The inventory phase encompasses inputs and outputs of energy and materials in and out of the system boundaries, compiled according to the defined FU. As shown in Figure 3, 60% of the studies considered some measured data about their WWTs to accomplish the LCA goal defined during their study. From the 43 studies reviewed, 88% were deemed to have sufficiently detailed their inventory sources, while five were insufficiently described or commented on during this phase. These were based on the following criteria: (i) inventory processes have not been described; (ii) inventory does not allow results reproducibility; and (iii) inventory does not support the results discussed in the paper.

The LCI is essential during LCAs since it directly affects the impact assessment phase and it is of utmost importance for the interpretation phase and conclusions. A clear definition and description of the LCI is crucial to ensure the reproducibility of the study and should clearly state and describe the primary and background data and their sources. The first problem affecting the reproducibility of LCIs found during this review is derived from a common procedure during WWPs inventories, which is the miscellany of data sources (Corominas et al., 2013). It has been observed in the reviewed articles that values from direct measurements were often used simultaneously with

literature and official reports as proxy data of requirements for materials, chemicals and energy without providing sufficient description and evaluation of their potential incompatibility or lack of representativeness to the LCA goal and purpose. Although a common practice, this may cause an additional source of concern for LCAs in developing countries primarily as their climate and socioeconomic aspects generally vary more widely (especially in larger ones such as China, India and Brazil) and thus the national or literature averages for wastewater composition may be less representative for the WWT under evaluation (Mara, 2004; Lesage and Samson, 2016). Secondly, another major barrier found was that background processes in databases were not named in the papers, making the reproduction of the LCA complicated. Furthermore, cut-off values were seldom mentioned during the LCI data compilation.

The principal source of impacts like global warming, acidification or photochemical oxidation, especially in intensive WWT technologies, has normally been considered the indirect emissions associated mainly to the production of the purchased electricity for the operation of the WWTP (Hospido et al., 2007; Machado et al., 2007; Niero et al., 2014) and to a lesser extent, to other sources like the production and use of chemicals (Slagstad and Brattebo, 2014). This fact justifies that all the 31 studies that include any site adaptation of the data considered the use of country electricity mix; however, in 24 cases this is the only adaptation (Figure 3 and Table S3 in SI). Regarding this matter, Hernández-Padilla et al. (2017) studied, when comparing extended aeration and pond system technologies, the implications of using specific country electricity mix for 22 Latin American and Caribbean (LAC) countries instead of an average mix for the whole region. The authors concluded that the use of country-adapted electricity mixes was crucial because the selection of the best treatment technology according to human health damage changed for 16 countries (73%).

The electricity consumption for non-extensive WWT of urban wastewaters in developing countries was found to be significantly lower than in developed countries, the latter based in the

studies assessed by Corominas et al. (2013). As illustrated in Figure 4, within developing countries the average electricity consumption for urban wastewaters by non-extensive treatments is 0.42 kWh/m³, while in developed countries this value is 0.50 kWh/m³. As commented during section 3.2, this may have to do with more stringent discharge parameters in developed countries, which requires more intense treatments (e.g. tertiary) or the average size of the WWTs sampled.

Figure 4 - Comparison of electricity consumption (in kWh/m³) between life cycle assessments for non-extensive treatments of urban wastewaters in developing and developed countries

The country-specific electricity mixes for developing countries can be obtained from several sources (e.g. World Bank statistics (WB, 2018)). However, the background data used to describe the electricity production and other key processes (e.g. construction materials or chemicals production) are still based in databases that were developed within European and North American conditions. Several of the studies in Table 1 (e.g. Tong et al., 2013; Li et al., 2013) claim that the lack of specific background data for developing countries reflecting local situations can seriously compromise the certainty of the results. In China, several advances towards a Chinese life cycle inventory database have been achieved in recent years (Gong et al., 2006; IKE, 2014), allowing some WWT studies to adapt the background data at national level beyond only the electricity mix (Zhang et al., 2010; Li et

al. 2013; Lam et al., 2015). However, a lack of exhaustiveness, especially in construction materials has been detected (Li et al., 2013) and available databases are still found not large enough, forcing researchers to use non-Chinese databases to model unavailable materials (Lam et al., 2015). Projects under development to create national LCI databases in other developing countries like India, South Africa, Thailand, Egypt or Brazil (Thinkstep, 2017; Ecoinvent, 2018a; UNEP, 2016) combined with free access to databases (Ecoinvent 2018b; OpenLCA, 2018) are expected to improve the accuracy of LCI of WWTP in the in consequential years. Accessibility of research and articles can be another barrier (only 16% of the studies are open access) and free or low-cost accessibility to relevant information should be encouraged for developing countries.

4.2.1. Accounting for direct greenhouse gas (GHG) emissions

The importance of considering direct GHGs emissions from the WWT processes has been highlighted in recent years (Corominas et al., 2012; Flores-Alsina et al., 2011; Lorenzo-Toja et al., 2016). However, only around half the studies in Table 1 included direct GHGs emissions (Figure 3 and Table S3 in SI). The aerobic WWT processes produce direct CO₂ emissions (Zang et al., 2015), N₂O is emitted from the secondary biological treatment, the effluent and sludge-receiving environment (Ahn et al., 2010) and the direct emissions of CH₄ are related with the anaerobic wastewater and/or sludge treatment processes (Daelman et al., 2012). Direct CO₂ emissions are considered biogenic and recommended to be accounted as GHG-neutral by IPCC (2006). However, Griffith et al. (2009) stated that not all of these emissions should be considered biogenic because up to 10% of the total organic carbon present in wastewaters may be of fossil origin (mainly related to detergents). This calculation was made for New York (USA) and may be overestimated for developing countries as specific measurements are still lacking. Rodríguez-García et al. (2012) concluded that these non-biogenic CO₂ emissions can be of similar importance to global warming potential (GWP) as those associated to electricity. Only 15 of the reviewed studies include direct

emissions of CO₂ (Table S3 in SI) and in all cases from estimations based on mass balance (e.g. Mahgoub et al., 2010) or in the IPCC (2006) methodologies (e.g. Polruang et al., 2018). From these 15 studies, 66% considered all the direct released CO₂ as a contributor to GWP, even if these emissions were biogenic.

In the case of N₂O, recent studies have also highlighted that these emissions can have a dominant role for the GWP impact of WWTP (Rodríguez-García et al. 2012; Foley et al. 2010a) with contributions approximately three times more than electricity use (Rodríguez-García et al., 2014). Direct emissions of N₂O can be influenced by different aspects like the concentration of dissolved oxygen, ammonium and nitrite, the COD/N ratio, a short sludge retention time, the presence of toxic compounds (like increased sulphide concentration), low temperatures or high salinity (Kampschreur et al., 2009). These are specific and local factors and, therefore, there are large uncertainties associated to the calculation of these emissions (Foley et al., 2010b), but several efforts have been made to quantify these emissions more accurately (Ahn et al., 2010; Corominas et al., 2012; Foley et al., 2010b; IPCC, 2006; Rodríguez-García et al., 2012). However, from the perspective of studies in developing countries, only 13 (30%) estimated any N₂O emissions, either based on mass balances (e.g. Pan et al. 2011; Lu et al., 2017) or using generic emission factors found in the literature (e.g. Singh et al., 2017; Wang et al., 2015). To reduce the uncertainty, Lam et al. (2015) considered specific N₂O emissions factors for different stages: i) operation of the activated sludge (Doka, 2003) and of vertical-flow constructed wetland (Søvik et al., 2006); ii) sludge drying in reed beds (Uggetti et al., 2012); and iii) emissions of the dried sludge when applied for agricultural purposes (Doka, 2003; Hobson, 2000).

Finally, in the case of CH₄, data acquisition is challenging because difficulties in their measurement due influence of site-specific conditions of the exposed system. For instance, in the case of CWs, the CH₄ emissions are influenced by different aspects like environmental conditions (higher water/soil/air temperature and solar radiation increase the emissions) and the choice and

agronomic management of vascular plants (Mander et al., 2014; Maucieri et al., 2017). Almost 50% of the reviewed studies considered CH₄ emissions, mainly associated with its potential for energy recovery when the sludge is anaerobically digested (e.g. Wang et al., 2012) or when analysing constructed wetlands or pond systems (e.g. Cornejo et al. 2013), but again using estimations based on mass balances or generic emissions factors from the literature (Hospido et al., 2005; Lundin et al., 2000; IPCC, 2006; Rolim, 2000) instead of direct measurement. Laitinen et al. (2017) concluded that direct emissions of CH₄ were the main contributor to GWP for a constructed wetland in Mexico. However, the authors declared the need for more studies to reduce the uncertainty because CH₄ emission vary significantly depending on the wetland type, choice of vegetation, seasonal and regional characteristics and other studies (Mander et al., 2014) reported values up to 80% higher. Other studies in Chile (Casas Ledón et al., 2017) and Bolivia (Cornejo et al., 2013) confirmed the significant role of CH₄ emissions in the GWP impact of constructed wetlands or pond systems.

To summarise, the percentage of LCA studies that consider direct emissions of GHGs in developing countries is low. When LCA studies consider direct emissions, the data is based on generic assumptions with a high level of uncertainty. Therefore, real measurements and transparency in the calculation methods applied should be encouraged, particularly as the contribution of N₂O and fossil-originated CO₂ emissions to GWP are potentially even more important than other factors currently considered (e.g. electricity and chemical consumption).

4.3. Life cycle impact assessment (LCIA)

Global warming potential (GWP) and eutrophication potential (EP) are the most common impact categories considered (95% and 79% of the studies, respectively), with seven studies measuring GWP exclusively (Table S4 in SI). These numbers are in concordance with previously published LCAs on WWTP, where EP is the most relevant impact category and GWP is included due to its political and social significance (UN, 2017) and as a reference of other energy dependent

impacts such as acidification or photochemical oxidation formation (Zhu et al., 2013; Rodríguez-García et al., 2011). Other LCA studies also highlight the importance of considering terrestrial ecotoxicity potential (TETP), mainly due to the negative effect of the heavy metals presence in the sludge applied for agricultural purposes (Charlton et al., 2016; Yoshida et al., 2018). However, only 21 papers (49%) include TETP among the calculated impacts (Table S4 in SI). This lack of consideration can be justified because: i) the articles do not include the sludge stage; ii) the preponderance of other final disposal of the sludge rather than agricultural in developing countries (e.g. landfilling); or iii) the lack of consideration of the effect of heavy metals (see section 4.1.2).

Although developed countries displayed a higher electricity consumption for wastewater treatment (see section 4.2 and Figure 4), the GWP of non-extensive WWTs treating urban wastewaters in these countries were found to be lower than developing countries. Figure 5 suggests urban WWTs in developed countries (from studies in Corominas et al., 2013) have, on average, a GWP of 0.39 kg CO₂-eq./m³ while in developing countries this value is 0.53 kg CO₂-eq./m³. This difference is most likely justified by the diverse participation of fossil fuels in the electricity mix. A caveat is necessary considering the uncertainty associated to GWP in WWTs (see section 4.2.1). The values shown in Figure 5 are total CO₂-eq. emissions per m³ of treated wastewater and thus includes studies that consider, or don't consider, direct GHGs (at similar proportions for developed and developing countries - 40% of the studies only consider indirect greenhouse gas emissions and 60% both indirect and direct).

Figure 5 - Comparison of the global warming potential (in kg CO₂-eq./m³) between life cycle assessments for non-extensive treatments of urban wastewaters in developing and developed countries

Regarding other impacts, land use indicators are normally only relevant for the assessment of natural treatment systems, as these indicators can help to avoid the overestimation of their environmental benefits, especially when compared with conventional wastewater treatment technologies (Zang et al., 2015). Nevertheless, this is not common practice in developing countries because from the eight articles in Table 1, which compare natural and conventional treatments, only three consider land use indicators. The importance of these low investment natural treatments for developing countries, the progressive incorporation of land use indicators to methodologies available in commercial software (e.g. ReCiPe (Goedkoop et al., 2009)) and the ongoing studies to improve the knowledge about the effects of land transformation on ecosystem quality (De Baan et al., 2012; Maia de Souza et al., 2013) should increase the implementation and reliability of these indicators in future WWT-related LCAs.

The water use impact, which measures the potential effects on water availability for human users and ecosystems, has recently started to gain importance in WWT-related LCAs. However, a coordinated assessment method is still pending (Boulay et al., 2015). This impact factor is of special

importance in developing countries, where water scarcity and the lack of access to clean freshwater are among the most prevalent challenges for a sustainable development (UN-Water, 2017a). Risch et al. (2014) demonstrated the influence of the water use impact category when comparing three WWT technologies (activated sludge, activated sludge with polishing ponds and vertical flow reed bed) in France, Spain and Egypt. For the case of Egypt, water use had a significant influence on the human health and ecosystem damage scores because of the lower Human Development Index of the country (i.e. the consumption of 1 m³ of freshwater has higher impact in human health than in developed countries) and due to the aridity of the region, that gives freshwater a very important ecological value.

From the above-mentioned impact categories, GWP is the only considered a site-generic impact category (Norris et al., 2002) and, therefore, a regionalization at characterization level is not necessary. However, in the case of EP and TETP, the potential impact can vary depending on the source location, the transportation of the pollutants between different environments and the sensitivity of the receptor ecosystems, and therefore, are considered site-dependent impact categories (Gallego et al., 2010, 2011). Similarly, land use impacts are highly spatial- and temporal-dependent (Mila i Canals et al., 2007).

In relation with the amount of impact categories considered, only 26% of the studies in Table 1 use all the impacts categories available in the methodologies and 40% assess only three or less impact categories (Table S4 in SI). These can be influenced by the fact that only 48% of the studies declare the use of commercial LCA software. This lack of software can slow down and narrow the scope of the research as the researchers must do their own calculations. The development of open access software (e.g. OpenLCA (2018) or CCalC (2018)) or new policies of free or reduced fee LCA software for educational institutions in developing countries (e.g. PRé Consultants (2018)) are expected to have a positive impact in future years.

4.3.1. Impact assessment methodologies

The impact methodology CML developed by the University of Leiden (Guinée et al., 2001) is the most used (16 studies) followed by the greenhouse gases characterization factors developed by the International Panel of Climate Change (IPCC, 2013), used in seven studies, most of them focused only on GWP (Table 1). From the perspective of developing countries, several authors suggested that the use impact methodologies developed in European and North American context (e.g. CML (Guinée et al., 2001), ReCiPe (Goedkoop et al., 2009), Impact 2002+ (Humbert et al., 2012) and TRACI (Bare, 2011)) can affect their results in impacts that are site-dependant or when normalization or weighting factors are considered (e.g. Palomares-Rodríguez et al., 2017; Lam et al., 2015). Accordingly, Zhao et al. (2017) compared the results of the implementation of bioaugmentation in a constructed wetland using a site-generic method like CML (Guinée et al., 2001) and e-Balance (IKE, 2014), an impact methodology designed specifically for China context. The authors found that both methodologies were complementary, because an in-depth perspective was provided by e-Balance (e.g. the need to further enhance pollutants removal of bioaugmentation). However, CML provides a more comprehensive analysis as it provides a wider coverage of environmental indicators and new conclusions (e.g. the importance to reduce the indirect impacts associated to inocula production).

Hernández- Padilla et al. (2017) compared the environmental impacts of extended aeration and pond treatments in the LAC country context using three life cycle impact assessment (LCIA) methodologies: i) Impact World+ (Bulle et al., 2012), a regionalized LCIA methodology, which provides CFs for LAC countries, and ii) Impact 2002+ (Humbert et al., 2012) and iii) ReCiPe hierarchical version (Goedkoop et al., 2009), two midpoint-damage methodologies developed for a European context. The results for the best alternative treatments were contradictory depending of the methodology used, especially when considering the impacts in the ecosystem quality damage

category. The authors recommended a further analysis of the structure of each impact methodology in order to be able to interpret the results accurately.

4.3.1.1. Regional characterization factors

Several studies have calculated regionalised characterization factors (CF) for EP for the US (Norris, 2002), Finland (Seppälä et al., 2004); Brittany, France (Basset-Mens et al., 2006), Galicia, Spain (Gallego et al., 2010) and Sweden (Henryson et al., 2018) and certain methodologies (e.g. Lime (Japan), TRACI (US) and LUCAS (Canada),) were developed to consider regional features. The need for eutrophication CFs to account for regional differences in sensitivities of environmental receptors in developing countries have been highlighted in several of the WWTPs reviewed studies from Table 1 (Kalbar et al. 2013; Wang et al., 2015; Lam et al. 2015). The recently developed E-Balance impact methodology includes eutrophication CFs for China (Bai et al., 2017) and there are three methodologies with site-specific CFs for aquatic eutrophication for several developing countries: Impact World + (Bulle et al., 2012), ReCiPe 2016 (Huijbregts et al., 2016) and LC-Impact (Verones, 2016). However, the effective use of these CFs is limited as these methodologies have not yet been integrated into commercial LCA software (Patouillard et al., 2018) and fate of nutrients from agricultural soils (where sludge from WWTP is commonly applied) to water are not available worldwide (Helmes et al., 2012). Furthermore, CFs for marine eutrophication are not available for Recipe 2016 and LC-Impact and are only available for atmospheric emissions in the case of Impact World+, although recent publications can help to incorporate these marine eutrophication CFs in the near future (Cosme and Hauschild, 2017; Cosme et al., 2017). Even so, Hernández-Padilla demonstrated the influence in the results of using spatially-differentiated eutrophication CFs (from Impact World +) instead of site-generic for 22 LAC countries when comparing the ecosystem quality damage of extended aeration and ponds system treatments. When the electricity mix is similar, extended aeration becomes the best option in those countries less sensitive to eutrophication. For

terrestrial toxicity (and in general toxicity-related impacts) significant progress and consensus have been achieved for the CFs of some compounds (e.g. organic compounds with model USEtox (Hauschild et al., 2008)). However, more spatial differentiated CFs for key pollutants in WWTPs effluents and sludge (e.g. heavy metals and PPCPs) are needed to reduce the uncertainty associated to toxicity categories (Zang et al., 2015).

Finally, in the case of land use, recent works have started the development of site-dependent CFs (De Baan et al., 2013; Liu et al., 2010) and inventories (Koellner et al., 2013). Nonetheless regionalization is still in its infancy when compared with other impacts, thereby deeper and detailed assessment is still needed.

4.3.1.2. Use of normalization and weighting

The implementation of normalization is relatively common (15 studies), but in the case of weighting is scarce (only four papers). The lack of specific site-adapted normalization and weighting factors in developing countries (Pizzol et al., 2017) is a drawback to its use, especially when considering that one of the major impacts in WWTP, eutrophication, has a direct impact on the regional environments. Generic World factors (e.g. Tong et al., 2013; Polruang et al., 2017) or from different geographical areas (e.g. Europe (e.g. Mahgoub et al., 2010; Roushdi et al. 2012; Meneses-Jácome et al., 2015; Wang et al., 2015; Lu et al., 2017) or Japan (Limphitakphong et al, 2016)) have been used due to the absence of more specific information. Moreover, Kalbar et al. (2013, 2014) and García-Montoya et al. (2016) justified not including normalization due to the lack of reference values for India and Mexico, respectively.

The most important advances in this topic have been made in China with several WWTP studies using Chinese normalization and weighting factors (Vera et al., 2015; Li et al., 2013; Zhao et al., 2018). Bai et al. (2017) calculated specific Chinese normalization and weighting factors and obtained different environmental hierarchy for four discharge scenarios of a WWTP with activated

sludge treatment when compared them with the results obtained with other non-Chinese normalization (CML (Guinée et al., 2001)) and weighting factors (BEES (Suh and Lippiatt, 2012)), EPA (Lippiatt, 2007) and EDIP 2003 (Hauschild and Potting, 2005)). Further development of general weighting factors and specific freshwater and terrestrial toxicity-based normalization indicators for China have been highlighted as main priorities for future WWT-related LCAs (Bai et al., 2017; Zhao et al., 2018; Li et al., 2013; Wang et al., 2015). To develop widely accepted weighting factors, the involvement of wide range of stakeholders is crucial. For example, Bai et al. (2018a,b) investigated the understanding and preferences of stakeholders using conjoint analysis and multiple weighting approach of several LCA impact categories for different wastewater treatment alternatives. The findings allowed to better comprehension of the tendencies in judgment and concerns towards the impacts of wastewater treatment, facilitating the LCA outcome communication and, consequently, stakeholder engagement and the decision-making process.

4.4. Interpretation

The important results arising from the previous phases are discussed and checked for its consistency in the interpretation, followed by limitations and recommendations based on the findings (ISO 14040:2006, 2006). The limitations regarding the assessments were a matter of concern in 34 papers (79%) and the comparison with other literature results applied in 25 (Figure 3 and Table S5 in SI).

Any kind of uncertainty analysis is only applied in 23% of the studies (Figure 3), slightly lower than the 35% accounted for developed countries by the review of Corominas et al. (2013). The studies with conventional technologies normally focus their sensitivity analysis on the electricity mix. All concluded the significant importance of the quality of these data as the environmental result are clearly influenced when other country-electricity mix (Fernández-Torres et al., 2012; Hernández-

Padilla et al., 2017), hypothetical changes in the sources (Pillay, 2006) or future electricity scenarios (Polruang et al., 2018) are considered.

For extensive treatment technologies, the sensitivity analyses are mainly focused on the CH₄ emission associated to constructed wetlands and pond systems. Cornejo et al. (2013) discovered that CH₄ emissions substantially influence GWP when the input value was modified by 20% in two treatments in Bolivia consisting in an UASB reactor or a facultative pond followed in both cases by two maturation ponds in series. Casas Ledón et al. (2017) observed that the variability in the emissions of CH₄ during the four seasons of the year was significant for GWP impact and higher than the changeability among the two species (*Phragmites australis* (Phr), *Schoenoplectus californicus* (Sch)) considered for a constructed wetland. Laitinen et al. (2017) reported that when the 25% and 75% quartile values of CH₄ and N₂O emissions from constructed wetland addressed by Mander et al. (2014) were applied instead of average values, the greenhouse gas fluxes were decreased and increased by 70% and 35%, respectively. Therefore, more field studies are required that measure the direct emissions of greenhouse gases (specially CH₄) in constructed wetlands and pond systems to reduce the level of uncertainty of the results that highly influence the GWP impact of these extensive treatment systems, which are especially attractive for developing countries (see sections 4.1.2 and 4.4). The effect in the decision making of what energy is considered as replaced if these CH₄ emissions are used as a biogas has also been assessed. Laitinen et al. (2017) compared the production of biogas from a constructed wetland (CH₄ recovery) and from an activated sludge plant (biogas produced through anaerobic digestion of the sludge). The authors concluded that constructed wetlands have lower GWP impact compared with activated sludge plants when the replaced energy come mainly from fossil fuels (especially if coal is the main source) because the latter produce less biogas and, therefore, only a limited amount of energy is replaced.

A study of the uncertainty of the inventory data is a necessary but not common practice in the WWT related-LCAs in developing countries. Only 23% of the studies include any kind of

uncertainty analysis (Figure 3) compared with 40% in developed countries (Corominas et al., 2013). The pedigree matrix (Weidema and Wesnæs, 1996) or Monte Carlo analysis (Heijungs and Huijbregts, 2004) are common tools in LCA and their use should be general and not exceptional (e.g. Hernández-Padilla et al., 2017; Kulak et al., 2017) in WWT-related LCAs in developing countries. The Monte Carlo analysis tool is available for free (e.g. OpenLCA, 2018) and the increase of availability and improvement of uncertainty values for the pedigree matrix has been a constant in databases like Ecoinvent in recent years (Ciroth et al., 2016). The progressive incorporation of developing-country specific data to commercial LCA software should increase the reliability and facilitate the access to these uncertainty factors in the near future.

5. Differences in LCAs between developed and developing countries

The aim of this review has been to emphasise the achievements and challenges in WWT in developing countries; however, some of the conclusions can be applied to LCAs of WWTs in general or to LCAs from other sectors in developing countries. Table 2 summarises the main challenges, research gaps and potential future research lines identified in this review and in what sections of the article have been discussed. In order to clarify the significance for poorer countries, the specific influences of the identified challenges in developing countries were compared with developed countries and were classified in low, medium and high categories in Table 2. This was based on the information contained in the reviewed articles and the criteria of the authors. Those challenges identified as of high influence for developing countries are discussed in detail in following sections.

5.1. Geographical and temporal distribution

The geographical diversity of studies is poor in developing countries when compared with developed countries. The reviews from Corominas et al (2013), Zang et al. (2015) and Hernández-Padilla et al. (2017) indicated that at least one wastewater treatment-related LCAs has been

developed within 23 out of the 34 most industrialized countries. However, as shown in Table 1, this type of study has been carried out in only 27 out of the 147 lesser developed countries, with more than 50% of the studies focus on China and India and with no studies in the 49 countries with lower incomes. Regarding the temporal scale, LCA of WWT has been systematically applied in developed countries since the beginning of the 2000s (Corominas et al. 2013), but have grown exponentially in developed countries within the last five years (Table 1). This justifies the development of reviews and best practice guidelines specifically focussed on less developed countries.

5.2. Technical parameters

The evaluation of urban wastewater influents in developing and developed countries showed that on average the values of the parameters considered in LCAs of developed countries are higher for BOD, COD and SS than those in the LCAs in developing countries, and similar in terms of TN and TP (see Figure 2). In developed countries values are slightly above and in developing countries slightly below “medium” strength sewage (Metcalf and Eddy, 1991) in terms of BOD, COD and SS. However, the removal of pollutants from urban wastewaters in WWTs was found, on average, to be higher in developed countries than in developing countries and, therefore, the values of the mentioned parameters in the effluents are lower (section 3.2). The differences in the removal rates were especially high for nitrogen and phosphorus. LCAs in developed countries considered average removal rates of 72% for TN and 75% for TP while in developing countries these values were 42% and 27%, respectively.

There is a lack of LCA studies focused on advanced treatments for the removal of emerging contaminants in developing countries. Several LCAs have been developed in recent years for advanced treatments but always in developed countries (e.g. Chatzisyneon et al. 2013; Gallego-Schmid et al., 2019; Zepon Tarpani and Azapagic 2018). These treatments techniques are key for the

removal from wastewater of new pollutants like PPCPs that pose significant risks to the environment (Rizzo et al., 2013; Luo et al. 2014; Yin et al., 2017). Moreover, there is evidence that the concentrations of some of these compounds, particularly antibiotics, are similar or higher in the wastewaters of developing countries, especially in Asia (Tran et al., 2018; Mohapatra et al., 2016). These compounds have also been found in the drinking water of Chinese and Brazilian cities (Sun et al., 2015; Machado et al., 2016). Therefore, it is suggested that developing countries begin to monitor the risks of these compounds to the environment in the light of recent European studies (Ågerstrand et al., 2015; Barbosa et al., 2016; Oldenkamp et al., 2016) and consider practices and advanced treatment options for their removal in WWTPs (Ahmed et al., 2017; Grandclement et al., 2017). Nevertheless, the environmental benefit of applying advanced treatments for their removal from wastewaters must be evaluated carefully through LCA (Wang et al. 2015; Zepon Tarpani and Azapagic, 2018).

5.3. Goal and scope

Regarding the FU, traditional quantities used in LCA for the calculation of p.e. (e.g. 60 g BOD₅) are based in developed countries figures and should be adapted to more accurate values for developing countries (e.g. 40-50 g BOD₅). Similar adaptations should be established when the functional unit considered is amount of p.e./household or water volume consumed per p.e. The inclusion of construction stage should be mandatory for extensive treatment technologies due to its influence in the impact. This recommendation is of high significance for developing countries due to the crucial role that this type of treatments is expected to play in less wealthy countries (see section 4.1.2).

5.4. Life cycle inventory

The lack of country-specific databases for developing countries forces authors in developing countries to use databases mainly focused on Europe and North America, increasing the uncertainty of the studies. The electricity consumption is a key parameter for the environmental impacts of WWTs (especially for intensive treatments) and should be specifically measured for each treatment and in each location. The average electricity consumption was found to be significantly lower in developing countries and the adoption of generic literature values from developed countries without careful assessment can heavily influence the LCA results. Related to this, the electricity mix should be considered at least at the country level to reduce uncertainties. Finally, discrepancies between national databases and the assessed WWTs influent wastewater quality, energy and chemical usage may be an additional issue in developing countries. This problem can be increased in large countries like China, Brazil, India or Mexico where, due to the high geographical variability, subnational regional-specific databases are needed, like those already available in large developed countries like Canada (Lesage and Samson, 2016) or Australia (Foley and Lant, 2009). The high cost of quality background databases can also be another limitation for developing countries and open or low-cost access should be encouraged.

5.5. Impact assessment

None of the aspects considered for impact assessment are evaluated as high significance for developing countries, mainly because there are general challenges for LCA-wastewater treatment studies. However, the calculation of specific regional characterization factors for eutrophication and normalization and weighting values is much less developed in poorer countries (see sections 4.3.1.1 and 4.3.1.2). Finally, a harmonized assessment method to calculate water use impact is of special importance in developing countries, where water scarcity and the lack of access to clean freshwater are common.

5.6. Interpretation of the results

As commented during section 3.4, extensive wastewater treatments have been suggested as an interesting alternative to provide a transition to more sustainable urban wastewater management. Research has been intensive over the practicability of this type of treatments in developing countries, especially for CWs (Zhang et al., 2014; Wu et al., 2015). In this sense, 31% of the studies of this review consider some form of extensive wastewater treatment compared with 12% in developed countries (Corominas et al., 2013). Therefore, the key parameters and associated uncertainty for these types of treatments like direct and indirect GHG emissions and the effects of change in the land use should be accounted in the interpretation stage especially when compared with centralized treatment.

Table 2 - Main challenges and research gaps for life cycle assessments (LCA) of wastewater treatment (WWT) in developing countries.

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Challenges/gaps	Description	Guidelines and future research	Influence ^a	Section
<i>Geographical and temporal distribution</i>				
Geographical	No representation of the poorest countries	Encouragement of studies in least developed countries beyond China and India	High	2
Temporal	Exponential increase of WWT- related LCAs in last five years, expected to be continued in next years	Development of reviews and best practice guidelines	High	2
<i>Technical parameters</i>				
Key parameters	The reviewed studies show, in general, a lack of detailed information about the influent, the effluent and the sludge produced.	WWT-related LCAs should minimally state the influent flow, sludge retention time, hydraulic retention time, influent and effluent biological oxygen demand, chemical oxygen demand, total phosphorus, and total nitrogen.	Medium	3.2
Technologies	Lack of studies focused on advance treatments and emerging contaminants	Studies focus on advanced treatments (e.g. advanced oxidation process) and emerging contaminants (e.g. pharmaceutical and personal care products) should be encouraged	High	3.1 and 3.4
<i>Goal and scope</i>				
Functional unit (FU)	Predominantly use volume and different definitions of person equivalent	More clarity in FU definition and analysis of the results using more than one FU	High	4.1.1
Construction stage	Construction stage is not included in most of the studies	The inclusion of construction stage should be mandatory for extensive treatment technologies	High	4.1.2
Sludge for agricultural use	The benefit of the avoidance of chemical fertilizers or the effect of the content of heavy metals are not considered	These aspects should be included in the system boundaries, and if not possible, address the influence in the conclusions of the study	Medium	4.1.2
<i>Life cycle inventory</i>				
Poor description of inventories	Inventories are not shown or with lack of detail, mixing measured and bibliographic data or not including the sources of background data. All these facts reduced the reproducibility of the studies	A life cycle inventory table should be included, describing in detail primary and background data and sources.	Low	4.2
Lack of site-specific databases	Use of databases mainly focused on Europe and North America, with lack of adaptation to developing countries context.	To increase the number of national-specific or regional-specific (for large countries) databases To increase open or low-cost access for current databases and studies The electricity mix should be considered at least at the country level	High	4.2

No accountability of direct emissions	No quantification of fossil-related CO ₂ , CH ₄ or N ₂ O emissions	To include a clear definition of how biogenic CO ₂ is accounted Development of in-situ measurements or, if not possible, use of generic emissions factors with an analysis of uncertainty	High	4.2
<i>Impact assessment</i>				
Low number of impacts	Global warming and eutrophication are calculated, but no other highly influential impacts like terrestrial toxicity (agricultural use of sludge), land use (extensive treatments) or water use.	Calculation of global warming and eutrophication should be mandatory and calculation of other impacts like eco-toxicities, land use or water use is highly recommended	Low	4.3
No site-specific characterization factors (CF)	No regionalized CFs for eutrophication are available for developing countries	Undergoing projects to develop regionalized CFs for eutrophication should continue and effective integration in commercial LCA software should be encouraged	Medium	4.3
Lack of normalization and weighting values	The studies are forced to use normalization and weighting values from the World or from different geographic areas (e.g. Europe)	Encouragement of the calculation of national or regional specific normalization and weighting factors in line with recent developments for China	Medium	4.3
<i>Interpretation</i>				
Lack of uncertainty analysis	Less than a quarter of the studies include any kind of uncertainty analysis	Key parameters like direct emissions of greenhouse gases (GHGs) (with high level of uncertainty) and electricity mix (high influence in the impact) should be tested with uncertainty analysis (e.g. sensitivity and Monte Carlo analysis or pedigree matrix)	Medium	4.4
Extensive treatments	Key aspects are not considered when compared with intensive treatments	Direct and indirect GHG emissions and the effects of change in the land use should be accounted when analysing the impacts of extensive treatments, especially when compared with intensive treatments	High	4.4

^a Specific relevance of the identified challenges in developing countries when compared with developed countries. Challenges evaluated as high are particularly relevant for developing countries; however, when the significance is evaluated as low, that challenge has similar significance in developed and developing countries.

6. Conclusions

The increasing urbanization and economic growth experienced in many developing countries in recent decades have led to an increasing pressure in their sanitation, as is evident in their lack of planning to accommodate efficient sanitation infrastructure. Several challenges are still to be addressed to overcome these gaps and policies and initiatives are necessary to guarantee universal and adequate sanitation coverage for the population living in these countries. This research has reviewed 43 papers in the literature and presented several points to aid in the development and management of wastewaters treatments in developing countries through the practice of LCA. It has found that LCAs of WWTs in developed countries would greatly benefit from the encouragement of studies within the poorest countries, the development of best practice guidelines and inclusion of advanced wastewater treatments for the removal of emerging contaminants. Additionally, better definition of FUs, development of national and regional life cycle databases and the inclusion of the construction phase and interpretation of the effects of land use change and direct GHGs emissions for extensive treatments would also be beneficial. Such actions should embrace current and future knowledge about environmental management to minimize the impacts of these systems and achieve the sustainable sanitation goals proposed by the United Nations.

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Appendix A.

Supplementary data Supplementary data to this article can be found online at <https://doi.org/10.1016/j.watres.2019.01.010>.

Declarations of interest

None

References

- Ågerstrand, M., Berg, C., Björleinius, B., Breitholtz, M., Brunström, B., Fick, J., Gunnarsson, L., Larsson, D.G.J., Sumpter, J.P., Tysklind, M., Rudén, C., 2015. Improving environmental risk assessment of human pharmaceuticals. *Environ. Sci. Technol.* 49, 5336–5345.
- Ahmed, M.T., 2010. Life Cycle Analysis in Wastewater: A Sustainability Perspective. In *Waste Water Treatment and Reuse in the Mediterranean Region*, eds. D Barceló, M Petrović, pp. 125–54. Springer-Verlag, Berlin, Germany.
- Ahmed, M.B., Zhou, J.L., Ngo, H.H., Guo, W., Thomaidis, N.S., Xu, J., 2017. Progress in the biological and chemical treatment technologies for emerging contaminant removal from wastewater: A critical review. *J. Hazard. Mater.* 323, 274–298.
- Ahn, J.H., Kim, S., Park, H., Rahm, B., Pagilla, K., Chandran, K., 2010. N₂O emissions from activated sludge processes, 2008–2009: results of a national monitoring survey in the United States. *Environ. Sci. Technol.* 44 (12): 4505–4511.
- Angelakis, A., Snyder S., 2015. Wastewater Treatment and Reuse: Past, Present, and Future. *Water*. 7 (9): 4887–95.
- Archer, E., Petrie, B., Kasprzyk-Hordern, B., Wolfaardt, G.M., 2017. The fate of pharmaceuticals and personal care products (PPCPs), endocrine disrupting contaminants (EDCs), metabolites and illicit drugs in a WWTW and environmental waters. *Chemosphere* 174, 437–446.
- Arroyo, P., Molinos-Senante, M., 2018. Selecting appropriate wastewater treatment technologies using a choosing-by-advantages approach. *Sci. Total Environ.* 625, 819–827.
- Bai, S., Wang, X., Zhang, X., Zhao, X., Ren, N., 2017. Life cycle assessment in wastewater treatment: Influence of site-oriented normalization factors, life cycle impact assessment methods, and weighting methods. *RSC Adv.* 7 (42): 26335–26341.

Bai, S., Zhu, X., Wang, X., Ren, N., 2018a. Identify stakeholders' understandings of life cycle assessment results on wastewater related issues. *Sci. Total Environ.* 622–623, 869–874.

Bai, S., Zhao, X., Wang, D., Zhang, X., Ren, N., 2018b. Engaging multiple weighting approaches and Conjoint Analysis to extend results acceptance of life cycle assessment in biological wastewater treatment technologies. *Bioresour. Technol.* 265, 349–356.

Barbosa, M.O., Moreira, N.F.F., Ribeiro, A.R., Pereira, M.F.R., Silva, A.M.T., 2016. Occurrence and removal of organic micropollutants: An overview of the watch list of EU Decision 2015/495. *Water Res.* 94, 257–279.

Bare, J.C., 2011. The tool for the reduction and assessment of chemical and other environmental impacts (TRACI). *Clean Technol. Environ. Policy* 13 (5): 687–696.

Basset-Mens, C., Anibar, L., Durand, P., van der Werf, H.M.G., 2006. Spatialised fate factors for nitrate in catchments: modelling approach and implication for LCA results. *Sci. Total Environ.* 367: 367–382.

Boulay, A.-M., Bare, J., De Camillis, C., Döll, P., Gassert, F., Gerten, D., Margni, M., Motoshita, M., Núñez, M., Pastor, A.V., Ridoutt, B., Schencker, U., Shirakawa, N., Vionnet, S., Worbe, S., Yoshikawa, S., Pfister, S., 2015. Consensus building on the development of a stress-based indicator for LCA-based impact assessment of water consumption: outcome of the expert workshops. *Int. J. Life Cycle Assess.* 20 (5): 577–583.

Bui, X.T., Vo, T.P.T., Ngo, H.H., Guo, W.S., Nguyen, T.T., 2016. Multicriteria assessment of advanced treatment technologies for micropollutants removal at large-scale applications. *Sci. Total Environ.* 563–564: 1050–67.

Bulle, C., Humbert, S., Jolliet, O., Rosenbaum, R., Margni, M., 2012. IMPACT world+: a new global regionalized life cycle impact assessment method. In: Proceedings LCA XII, Washington, Tacoma, United States.

Casas Ledón, Y., Rivas, A., López, D., Vidal, G., 2017. Life-cycle greenhouse gas emissions assessment and extended exergy accounting of a horizontal-flow constructed wetland for municipal wastewater treatment: A case study in Chile. *Ecol. Indic.* 74: 130–139.

CCalC, 2018. Carbon Calculations over the Life Cycle of Industrial Activities. Available at: <http://www.ccalc.org.uk/> (last access: 09/08/2018).

Charlton, A., Sakrabani, R., Tyrrel, S., Rivas Casado, M., McGrath, S.P., Crooks, B., Cooper, P., Campbell, C.D., 2016. Long-term impact of sewage sludge application on soil microbial biomass: An evaluation using meta-analysis. *Environ. Pollut.* 219, 1021–1035.

Chirisa, I., Bandaiko, E., Matamanda, A., Mandisvika, G., 2017. Decentralized domestic wastewater systems in developing countries: the case study of Harare (Zimbabwe). *Appl. Water Sci.* 7, 1069–1078.

Chong, M.N., Sharma, A.K., Burn, S., Saint, C.P., 2012. Feasibility study on the application of advanced oxidation technologies for decentralised wastewater treatment. *J. Clean. Prod.* 35: 230–38,

Ciroth, A., Muller, S., Weidema, B., Lesage, P., 2016. Empirically based uncertainty factors for the pedigree matrix in ecoinvent. *Int. J. Life Cycle Assess.* 21 (9): 1338–1348.

Cornejo, P.K., Zhang, Q., Mihelcic, J.R., 2013. Quantifying benefits of resource recovery from sanitation provision in a developing world setting. *J. Environ. Manage.* 131: 7–15.

Corominas, L., Flores-Alsina, X., Snip, L., Vanrolleghem, P.A., 2012. Comparison of different modeling approaches to better evaluate greenhouse gas emissions from whole wastewater treatment plants. *Biotechnol. Bioeng.* 109 (11): 2854–2863.

Corominas, L., Foley, J., Guest, J.S., Hospido, A., Larsen, H.F., Morera, S., Shaw, A., 2013. Life cycle assessment applied to wastewater treatment: State of the art. *Water Res* 47(15): 5480–92.

Cosme, N., Hauschild, M.Z., 2017. Characterization of waterborne nitrogen emissions for marine eutrophication modelling in life cycle impact assessment at the damage level and global scale. *Int. J. Life Cycle Assess.* 1–13.

Cosme, N., Mayorga, E., Hauschild, M.Z. 2017. Spatially explicit fate factors of waterborne nitrogen emissions at the global scale. *Int. J. Life Cycle Assess.* 1-11.

Daelman, M.R.J., van Voorthuizen, E.M., van Dongen, U.G.J.M., Volcke, E.I.P., van Loosdrecht, M.C.M., 2012. Methane emission during municipal wastewater treatment. *Water Res.* 46, 3657–3670.

De Baan, L., Alkemade, R., Koellner, T., 2012. Land use impacts on biodiversity in LCA: a global approach. *Int. J. Life Cycle Assess.* 18: 1216–1230.

De Baan, L., Mutel, C., L., Curran, M., Hellweg, S., Koellner, T., 2013. Land use in life cycle assessment: global characterization factors based on regional and global potential species extinction. *Environ. Sci. Technol.* 47: 9281–9290.

De Oliveira Schwaickhardt, R., Machado, E.L., Lutterbeck, C.A., 2017. Combined use of VUV and UVC photoreactors for the treatment of hospital laundry wastewaters: Reduction of load parameters, detoxification and life cycle assessment of different configurations. *Sci. Total Environ.* 590–591: 233–241.

Diniz, G.S., Tourinho, T.C.O., Silva, A.F., Chaloub, R.M., 2017. Environmental impact of microalgal biomass production using wastewater resources. *Clean Technol. Environ. Policy* 19 (10): 2521–2529.

Dixon, A., Simon, M., Burkitt, T., 2003. Assessing the environmental impact of two options for small-scale wastewater treatment: Comparing a reedbed and an aerated biological filter using a life cycle approach. *Ecol. Eng.* 20 (4): 297-308.

Doka, G., 2003. Life Cycle Inventories of Waste Treatment Services. Ecoinvent report no. 13. Swiss Center for Life Cycle Inventories, Dubendorf, Switzerland.

Ebele, A.J., Abou-Elwafa Abdallah, M., Harrad, S., 2017. Pharmaceuticals and personal care products (PPCPs) in the freshwater aquatic environment. *Emerg. Contam.* 3, 1–16.

Ecoinvent, 2018a. Life Cycle Inventories (LCI) Project. Available at: <http://www.ecoinvent.org/about/projects/sri-project/sri-project.html> (last access: 09/08/2018)

Ecoinvent, 2018b. Free educational license for universities and high schools in non-OECD countries. Available at: <http://www.ecoinvent.org/database/buy-a-licence/licence-types/licence-types.html> (last access: 09/08/2018)

EMSL, 2013. Commercial List Pricing. Available at: <https://www.emsl.com/PDFDocuments/Marketing/LabServicesGuide.8.27c.pdf> (last access: 09/08/2018)

Fajardo, T., Pinilla, P., Bojacá, V., Pinilla, R., Ortiz, J., Acevedo, P., Tatiana, F., Paula, P., Vivian, B., Rigoberto, P., Jannet, O., Paola, A., 2016. Life cycle assessment to identify environmental improvements in an aerobic waste water treatment plant. *Chem. Eng. Trans.* 49, 493-498.

Fernández-Torres, M.J., Randall, D.G., Melamu, R., von Blottnitz, H., 2012. A comparative life cycle assessment of eutectic freeze crystallisation and evaporative crystallisation for the treatment of saline wastewater. *Desalination* 306: 17-23.

Flores-Alsina, X., Gallego, A., Feijoo, G., Rodríguez-Roda, I., 2010. Multiple-objective evaluation of wastewater treatment plant control alternatives. *J. Environ. Manage.* 91 (5): 1193–1201.

Flores-Alsina, X., Corominas, L., Snip, L., Vanrolleghem, P.A., 2011. Including greenhouse gas emissions during benchmarking of wastewater treatment plant control strategies. *Water Res.* 45, 4700–4710.

Foley, J., Lant, P., 2009. Regional normalisation figures for Australia 2005/2006-inventory and characterisation data from a production perspective. *Int J LCA*, 14 (3): 215–224.

Foley, J., de Haas, D., Hartley, K., Lant, P., 2010a. Comprehensive life cycle inventories of alternative wastewater treatment systems. *Water Res.* 44 (5): 1654–66.

Foley, J.M., de Haas, D., Yuan, Z., Lant, P., 2010b. Nitrous oxide generation in full-scale biological nutrient removal wastewater treatment plants. *Water Res.* 44(3): 831–844.

Gallego, A., Hospido A., Moreira M.T., Feijoo, G., 2008. Environmental performance of wastewater treatment plants for small populations. *Resour. Conserv. Recycl.* 52 (6): 931–40.

Gallego, A., Rodríguez L, Hospido, A, Moreira MT, Feijoo, G., 2010. Development of regional characterization factors for aquatic eutrophication. *Int. J. Life Cycle Assess.* 15 (1): 32–43.

Gallego, A., Rodríguez, L., Hospido, A., Moreira, M.T., Feijoo, G., 2011. Development of LCA characterization factors for terrestrial eutrophication at regional scale. *Global J. Environ. Sci. Technol.* 1 (21): 1–13.

Gallego-Schmid, A., Tarpani, R. R. Z., Miralles-Cuevas, S., Cabrera-Reina, A., Malato, S., Azapagic, A., 2019. Environmental assessment of solar photo-Fenton processes in combination with nanofiltration for the removal of micro-contaminants from real wastewaters *Sci. Total Environ.* 650: 2210–2220.

García Alarcón, J.S., Martínez, A.R., Orozco, I.H., 2011. Life cycle assessment to municipal wastewater treatment plant. *Chem. Eng. Trans.* 24: 1345–1350.

García-Montoya, M., Sengupta, D., Nápoles-Rivera, F., Ponce-Ortega, J.M., El-Halwagi, M.M., 2016. Environmental and economic analysis for the optimal reuse of water in a residential complex. *J. Clean. Prod.* 130: 82–91.

Gavrilescu, M., Demnerova, K., Aamand, J., Agathos, S., Fava, F., 2014. Emerging pollutants in the environment: Present and future challenges in biomonitoring, ecological risks and bioremediation. *N. Biotechnol.* 32 (1): 147–56.

Goedkoop, M., Heijungs, R., Huijbregts, M., De Schryver, A., Struijs, J., Van Zelm, R., 2009. Recipe 2008. A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level. Ministry of housing spatial planning and the environment. Ministry of Housing Spatial Planning and the Environment, The Hague, the Netherlands.

Gong, X.Z.H., 2006. Basic database researches for the life cycle assessment of materials. Dissertation of Doctor's Degree, Beijing University of Technology, Beijing (China).

Grandclement, C., Seyssiecq, I., Piram, A., Wong-Wah-Chung, P., Vanot, G., Tiliacos, N., Roche, N., Doumenq, P., 2017. From the conventional biological wastewater treatment to hybrid processes, the evaluation of organic micropollutant removal: A review. *Water Res.* 111, 297–317.

Griffith, D.R., Barnes, R.T., Raymond, P.A., 2009. Inputs of fossil carbon from wastewater treatment plants to U.S. rivers and oceans. *Environ. Sci. Technol.* 43 (15): 5647–5651.

Guinée, J.B., Gorreë, M., Heijungs, R., Huppes, G., Kleijn, G.R., van Oers, R.L., Wegener, L., Sleeswijk, A., Suh, S., de Haes, H.A. Udo, de Bruijn, H., van Duin, H.R., Huijbregts, M.A.J., 2001. Life Cycle Assessment, an Operational Guide to the ISO Standards. Part 2a: Guide. Kluwer Academic Publishers, Dordrecht, the Netherlands.

Hauschild, M.Z., Goedkoop, M., Guinée, J., Heijungs, R., Huijbregts, M., Jolliet, O., Margni, M., De Schryver, A., Humbert, S., Laurent, A., 2008. Identifying best existing practice for characterization modeling in life cycle impact assessment. *Int. J. Life Cycle Assess.* 18 (3): 683–697.

Hauschild, M.Z., Potting, J., 2005. Spatial differentiation in life cycle impact assessment. The EDIP2003 methodology. Institute for Product Development, Technical University of Denmark Lyngby, Denmark.

Heijungs, R.; Huijbregts, M.A.J. 2004. A review of approaches to treat uncertainty in LCA. Proceedings of the 2nd Biennial Meeting of iEMSs, Complexity and integrated resources management, 14-17 June 2004, Osnabrück, Germany, pp. 332-339. Elsevier, Orlando, USA.

Helmes, R. J.K., Huijbregts M.A.J., Henderson A. D., Jolliet, O., 2012. Spatially explicit fate factors of phosphorous emissions to freshwater at the global scale. *Int. J. Life Cycle Assess.* 17: 646-654.

Henryson, K., Hansson, P.-A., Sundberg, C., 2018. Spatially differentiated midpoint indicator for marine eutrophication of waterborne emissions in Sweden. *Int. J. Life Cycle Assess.* 23 (1): 70-81.

Hernández-Padilla, F., Margni, M., Noyola, A., Guereca-Hernández, L., Bulle, C., 2017. Assessing wastewater treatment in Latin America and the Caribbean: Enhancing life cycle assessment interpretation by regionalization and impact assessment sensibility. *J. Clean. Prod.* 142: 2140-2153.

Hobson, J., 2000. CH₄ and N₂O emissions from waste water handling. In: IPCC Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories, pp. 441-454. Intergovernmental Panel on Climate Change, Geneva, Switzerland.

Hong, J., Ismail, Z.Z., Hong, J., 2011. Economic and environmental assessment of pulp and paper industrial wastewater treatment. *Applied Mech. Mat.* 90-93: 2929-2932.

Hospido, A., Moreira, M.T., Feijoo, G., 2007. A comparison of municipal wastewater treatment plants for big centres of population in Galicia (Spain). *Int. J. Life Cycle Assess.* 13 (1): 57-64.

Hospido, A., Moreira, T., Martín, M., Rigola, M., Feijoo, G., 2005. Environmental evaluation of different treatment processes for sludge from urban wastewater treatments: anaerobic digestion versus thermal processes. *Int. J. Life Cycle Assess.* 10 (5): 336-345.

Huijbregts, M.A.J., Steinmann, Z.J.N., Elshout, P.M.F., Stam, G., Verones, F., Vieira, M., Zijp, M., Hollander, A., van Zelm, R., 2016. ReCiPe 2016: A harmonized life cycle impact assessment method at midpoint and endpoint level. Report I: Characterization. RIVM Report 2016-0104. National Institute for Public Health and the Environment, Bilthoven, the Netherlands.

Humbert, S., De Schryver, A., Bengoa, X., Margni, M., Jolliet, O., 2012. IMPACT 2002+: User Guide. Draft for Version Q2.21 (version adapted by Quantis). Available at: http://www.quantis-intl.com/pdf/IMPACT2002_UserGuide_for_vQ2.21.pdf (last access: 09/08/2018).

IKE, 2014. Full-featured LCA Software and Database Package – eBalance. Available at: <http://www.ike-global.com/products-2/lca-software-ebalance> (last access: 09/08/2018)

IPCC, 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories, vol. 5. Intergovernmental Panel for Climate Change, Geneva, Switzerland.

IPCC, 2013. Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, USA.

ISO 14040:2006, 2006. Environmental management — Life cycle assessment — Principles and framework. International Organization for Standardization, Geneva, Switzerland.

ISO 14044:2006, 2006. Environmental management — Life cycle assessment — Requirements and guidelines. International Organization for Standardization, Geneva, Switzerland.

Jung, Y.T., Narayanan, N.C., Cheng, Y.L., 2018. Cost comparison of centralized and decentralized wastewater management systems using optimization model. *J. Environ. Manage.* 213, 90–97.

Kalbar, P.P., Karmakar, S., Asolekar, S.R., 2013. Assessment of wastewater treatment technologies: life cycle approach. *Water Environ. J.* 27: 261–268.

Kalbar, P.P., Karmakar, S., Asolekar, S.R., 2014. Life cycle-based environmental assessment of municipal wastewater treatment plant in India. *Int. J. Environ. Waste Manag.* 14 (1): 84–97.

Kamble, S.J., Chakravarthy, Y., Singh, A., Chubilleau, C., Starkl, M., Bawa, I., 2017. A soil biotechnology system for wastewater treatment: technical, hygiene, environmental LCA and economic aspects. *Environ. Sci. Pollut. R.* 24 (15): 13315-13334.

Kampschreur, M.J., Temmink, H., Kleerebezem, R., Jetten, M.S.M., van Loosdrecht, M.C.M., 2009. Nitrous oxide emission during wastewater treatment. *Water Res.* 43 (17): 4093-4103.

Kivaisi, A.K., 2001. The potential for constructed wetlands for wastewater treatment and reuse in developing countries: a review. *Ecological Engineering*. 16.: 545-560.

Koellner, T., de Baan, L., Beck, T., Brandao, M., Civit, B., Goedkoop, M., Margni, M., Mila i Canals, L., Müller-Wenk, R., Weidema, B., Wittstock, B., 2013. Principles for life cycle inventories of land use on a global scale. *Int. J. Life Cycle Assess.* 18: 1203-1215.

Kulak, M., Shah, N., Sawant, N., Unger, N., King, H., 2017. Technology choices in scaling up sanitation can significantly affect greenhouse gas emissions and the fertiliser gap in India. *J. Water Sanit. Hygi. Develop.* 7 (3): 466-476.

Laitinen, J., Moliis, K., Surakka, M., 2017. Resource efficient wastewater treatment in a developing area—Climate change impacts and economic feasibility. *Ecol. Eng.* 103: 217-225.

Lam, L., Kurisu, K., Hanaki, K., 2015. Comparative environmental impacts of source-separation systems for domestic wastewater management in rural China. *J. Clean. Prod.* 104: 185-198.

Li, Y., Luo, X., Huang, X., Wang, D., Zhang, W., 2013. Life Cycle Assessment of a municipal wastewater treatment plant: A case study in Suzhou, China. *J. Clean. Prod.* 57: 221-227.

Lesage, P., Samson, R., 2016. The Quebec Life Cycle Inventory Database Project: Using the ecoinvent database to generate, review, integrate, and host regional LCI data. *Int J LCA*, 21 (9): 1282-1289.

Libralato, G., Volpi Ghirardini, A., Avezzi, F., 2012. To centralise or to decentralise: An overview of the most recent trends in wastewater treatment management. *J. Environ. Manage.* 94, 61–68.

- Limphitakphong, N., Pharino, C., Kanchanapiya, P., 2016. Environmental impact assessment of centralized municipal wastewater management in Thailand. *Int. J. Life Cycle Assess.* 21 (12): 1789–1798.
- Lippiatt, B.C., 2007. BEES 4.0. Building for Environmental and Economic Sustainability Technical Manual and User Guide. National Institute of Standards and Technology (NIST), Gaithersburg, USA.
- Liu, Y., Nie, Z., Sun, B., Wang, Z., Gong, X., 2010. Development of Chinese characterization factors for land use in life cycle impact assessment. *Sci. China Tech. Sci.*, 53: 1483–1488.
- Lofrano, G., Brown, J., 2010. Wastewater management through the ages: A history of mankind. *Sci. Total Environ.* 408 (22): 5254–64.
- Lorenzo-Toja, Y., Alfonsín, C., Amores, M.J., Aldea, X., Marin, D., Moreira, M.T., Feijoo, G., 2016. Beyond the conventional life cycle inventory in wastewater treatment plants. *Sci. Total Environ.* 553, 71–82.
- Lu, B., Du, X., Huang, S., 2017. The economic and environmental implications of wastewater management policy in China: From the LCA perspective. *J. Clean. Prod.* 142: 3544–3557.
- Lundin, M., Bengtsson, M., Molander, S., 2000. Life Cycle Assessment of Wastewater systems: influence of system boundaries and scale on calculated environmental loads. *Environ. Sci. Technol.* 34 (1): 180–186.
- Luo, Y., Guo, W., Ngo, H.H., Nghiem, L.D., Hai, F.I., Zhang, J., Liang, S., Wang, X.C., 2014. A review on the occurrence of micropollutants in the aquatic environment and their fate and removal during wastewater treatment. *Science of the Total Environment* 473–474: 619–641.
- Lutterbeck, C.A., Kist, L.T., Lopez, D.R., Zerwes, F.V., Machado, E.L., 2017. Life cycle assessment of integrated wastewater treatment systems with constructed wetlands in rural areas. *J. Clean. Prod.* 148: 527–536.

Machado, A.P., Urbano, L., Brito, A.G., Janknecht, P., Salas, J.J., Nogueira, R., 2007. Life cycle assessment of wastewater treatment options for small and decentralized communities. *Water Sci. Technol.* 56 (3): 15–22.

Machado, K.C., Grassi, M.T., Vidal, C., Pescara, I.C., Jardim, W.F., Fernandes, A.N., Sodré, F.F., Almeida, F. V., Santana, J.S., Canela, M.C., Nunes, C.R.O., Bichinho, K.M., Severo, F.J.R., 2016. A preliminary nationwide survey of the presence of emerging contaminants in drinking and source waters in Brazil. *Sci. Total Environ.* 572, 138–146.

Mahgoub, M., Van Der Steen, N.P., Abu-Zeid, K., Vairavamoorthy, K., 2010. Towards sustainability in urban water: A life cycle analysis of the urban water system of Alexandria City, Egypt. *J Clean Prod.* 18 (10–11): 1100–1106.

Maia de Souza, D., Flynn D.F.B., DeClerck, F., Rosenbaum R. K., de Melo Lisboa, H., Koellner, T., 2013. Land use impacts on biodiversity in LCA: proposal of characterization factors based on functional diversity. *Int. J. Life Cycle Assess.* 18, 1231–1242.

Mander, Ü., Gabriela, D., Ebie, Y., Towprayoon, S., Chiemshaisri, C., Furlan Nogueira, S., Jamsranjav, B., Kasak, K., Truu, J., Tournebize, J., Mitsch, W.J., 2014. Greenhouse gas emission in constructed wetlands for wastewater treatment: a review. *Ecol. Eng.* 66: 19–35.

Mannina, G., Ekama, G., Caniani, D., Cosenza, A., Esposito, G., Gori, R., Garrido-Baserba, M., Rosso, D., Olsson, G., 2016. Greenhouse gases from wastewater treatment - A review of modelling tools. *Sci. Total Environ.* 551–552, 254–270.

Mara, D., 2004. Domestic wastewater treatment in developing countries. Earthscan, London, UK

Marlow, D.R., Moglia, M., Cook, S., Beale, D.J., 2013. Towards sustainable urban water management: A critical reassessment. *Water Res.* 47, 7150–7161.

Massoud, M.A., Tarhini, A., Nasr J.A., 2009. Decentralized approaches to wastewater treatment and management: Applicability in developing countries. *J. Environ. Manage.* 90 (1): 652–59.

Maucieri, C., Barbera, A.C., Vymazal, J., Borin, M., 2017. A review on the main affecting factors of greenhouse gases emission in constructed wetlands. *Agric. For. Meteorol.* 236: 175–193.

Meneses-Jácome, A., Osorio-Molina, A., Parra-Saldívar, R., Gallego-Suárez, D., Velásquez-Arredondo, H.I., Ruiz-Colorado, A.A., 2015. LCA applied to elucidate opportunities for biogas from wastewaters in Colombia. *Water Sci. Technol.* 71 (2): 211–219.

Metcalf and Eddy, 1991. *Wastewater Engineering: Treatment, Disposal and Reuse*, McGraw-Hill Education, Sydney.

Mila i Canals, L., Bauer, C., Depestele, J., Dubreuil, A., Freiermuth Knuchel, R., Gaillard, G., Michelsen, O., Müller-Wenk, R., Rydgren, B., 2007. Key elements in a framework for land use impact assessment within LCA. *Int. J. Life Cycle Assess.* 12, 5–15.

Niero, M., Pizzol, M., Bruun, H.G., Thomsen, M., 2014. Comparative life cycle assessment of wastewater treatment in Denmark including sensitivity and uncertainty analysis. *J. Clean. Prod.* 68, 25–35.

Norris, G., 2002. Impact characterization in the tool for the reduction and assessment of chemical and other environmental impacts. Methods for acidification, eutrophication and ozone formation. *J. Indust. Ecol.* 6 (3–4): 79–101.

Oldenkamp, R., Huijbregts, M.A.J., Ragas, A.M.J., 2016. The influence of uncertainty and location-specific conditions on the environmental prioritisation of human pharmaceuticals in Europe. *Environ. Int.* 91, 301–311.

OECD, 2018. DAC List of ODA. Effective for reporting on 2018, 2019 and 2020 flows. Available at: http://www.oecd.org/dac/financing-sustainable-development/development-finance-standards/DAC_List_ODA_Recipients2018to2020_flows_En.pdf (last access: 09/08/2018).

Ontiveros, G.A., Campanella, E.A., 2013. Environmental performance of biological nutrient removal processes from a life cycle perspective. *Bioresour. Technol.* 150: 506–512.

OpenLCA, 2018. The open source Life Cycle and Sustainability Assessment software. Available at: <http://www.openlca.org/> (last access: 09/08/2018).

Opher, T., Friedler, E., 2016. Comparative LCA of decentralized wastewater treatment alternatives for non-potable urban reuse. *J. Environ. Manage.* 182: 464–76.

Palomares-Rodríguez, C., Martínez-Guido, S.I., Apolinar-Cortés, J., del Carmen Chávez-Parga, M., García-Castillo, C.C., Ponce-Ortega, J.M., 2017. Environmental, Technical, and Economic Evaluation of a New Treatment for Wastewater from Slaughterhouses. *Int. J. Environ. Res.* 11 (4): 535-545.

Pan, T., Zhu, X.-D., Ye, Y.-P., 2011. Estimate of life-cycle greenhouse gas emissions from a vertical subsurface flow constructed wetland and conventional wastewater treatment plants: A case study in China. *Ecol. Eng.* 37 (2): 248-254.

Patouillard, L., Bulle, C., Querleu, C., Maxime, D., Osset, P., Margni, M., 2018. Critical review and practical recommendations to integrate the spatial dimension into life cycle assessment. *J. Clean. Prod.* 177: 398-412.

Pillay, S.D., 2006. An environmental life cycle assessment of the provision of recycled water in Durban. PhD thesis, University of Kwazulu-Natal, Durban, South Africa.

Pizzol, M., Laurent, A., Sala, S., Weidema, B., Verones, F., Koffler, C., 2017. Normalisation and weighting in life cycle assessment: quo vadis? *Int. J. Life Cycle Assess.* 22 (6): 853-866.

Polruang, S., Sirivithayapakorn, S., Prateep Na Talang, R., 2018. A comparative life cycle assessment of municipal wastewater treatment plants in Thailand under variable power schemes and effluent management programs. *J. Clean. Prod.* 172: 635-648.

PRé Consultants, 2018. SimaPro Faculty. Available at: <https://simapro.com/licences/faculty/> (last access: 09/08/2018).

Raghuvanshi, S., Bhakar, V., Sowmya, C., Sangwan, K.S., 2017. Waste Water Treatment Plant Life Cycle Assessment: Treatment Process to Reuse of Water. *Procedia CIRP* 61: 761-766.

Ren, J., Liang, H., 2017. Multi-criteria group decision-making based sustainability measurement of wastewater treatment processes. *Environ. Impact Assess. Rev.* 65, 91–99.

Risch, E., Gutierrez, O., Roux, P., Boutin, C., Corominas, L., 2015. Life cycle assessment of urban wastewater systems: Quantifying the relative contribution of sewer systems. *Water Res.* 77: 35–48.

Rodríguez-García, G., Frison, N., Vazquez-Padin, J.R., Hospido, A., Garrido, J.M., Fatone, F., Bolzonella, D., Moreira, M., Feijoo, G., 2014. Life cycle assessment of nutrient removal technologies for the treatment of anaerobic digestion supernatant and its integration in a wastewater treatment plant. *Sci. Total Environ.* 490, 871-879.

Rizzo, L., Manaia, C., Merlin, C., Schwartz, T., Dagot, C., Ploy, M.C., Michael, I., Fatta-Kassinos, D., 2013. Urban wastewater treatment plants as hotspots for antibiotic resistant bacteria and genes spread into the environment: A review. *Sci. Total Environ.* 447, 345–360.

Rodríguez-García, G., Hospido, A., Bagley, D.M., Moreira, M.T., Feijoo, G., 2012. A methodology to estimate greenhouse gases emissions in life cycle inventories of wastewater treatment plants. *Environ. Impact Assess. Rev.* 37: 37-46.

Rodríguez-García, G., Molinos-Senante, M., Hospido, A., Hernández-Sancho, F., Moreira, M. T., Feijoo, G., 2011. Environmental and economic profile of six typologies of wastewater treatment plants. *Water Res.* 45 (18): 5997–6010.

Roefs, I., Meulman, B., Vreeburg, J.H.G., Spiller, M., 2017. Centralised, decentralised or hybrid sanitation systems? Economic evaluation under urban development uncertainty and phased expansion. *Water Res.* 109: 274-286.

Rolim, S., 2000. Stabilization Ponds System (In Spanish). McGraw Hill Interamericana, Bogotá, Colombia.

Roushdi, M., El-Hawary, A., Mahgoub, M., 2012. Environmental improvement of Alexandria's wastewater treatment plants using life cycle assessment approach. *Global Nest J.* 14 (4): 450-459.

Seppälä, J., Knuuttila, S., Silvo, K., 2004. Eutrophication of aquatic ecosystems — a new method for calculating the potential contributions of nitrogen and phosphorus. *Int. J. Life Cycle Assess.* 9: 90–100.

Sharma, A., Burn, S., Gardner, T., Gregory, A., 2010. Role of decentralised systems in the transition of urban water systems. *Water Sci. Technol. Water Supply* 10, 577–583.

Sharma, B., Sarkar, A., Singh, P., Singh, R.P., 2017. Agricultural utilization of biosolids: A review on potential effects on soil and plant grown. *Waste Manag.* 64, 117–132.

Shifrin, N., 2005. Pollution Management in the Twentieth Century. *Journal of Environmental Engineering.* 131(5): 676-691.

Singh, A., Kamble, S.J., Sawant, M., Chakravarthy, Y., Kazmi, A., Aymerich, E., Starkl, M., Ghangrekar, M., Philip, L., 2018. Technical, hygiene, economic, and life cycle assessment of full-scale moving bed biofilm reactors for wastewater treatment in India. *Environ. Sci. Pollut. Res. Int.* 25 (3): 2552-2569.

Singh, N.K., Singh, R.P., Kazmi, A.A., 2017. Environmental impact assessment of a package type IFAS reactor during construction and operational phases: A life cycle approach. *Water Sci. Technol.* 75 (10): 2246-2256.

Singh, P., Kansal, A., 2018. Energy and GHG accounting for wastewater infrastructure. *Resour. Conserv. Recycl.* 128: 499-507.

Singh, P., Kansal, A., Carliell-Marquet, C., 2016. Energy and carbon footprints of sewage treatment methods. *J. Environ. Manage.* 165: 22-30.

Slagstad, H., Brattebo, H., 2014. Life cycle assessment of the water and wastewater system in Trondheim, Norway e a case study. *Urban Water J.* 11(4): 323-334.

Søvik, A.K., Augustin, J., Heikkinen, K., Huttunen, J.T., Necki, J.M., Karjalainen, S.M., Kløve, B., Liikanen, A., Mander, U., Puustinen, M., Teiter, S., Wachniew, P., 2006. Emission of the greenhouse gases nitrous oxide and methane from constructed wetlands in Europe. *J Environ Qual* 35: 2360-2373.

Suh, S., Lippiatt, B.C., 2012. Framework for hybrid life cycle inventory databases: a case study on the Building for Environmental and Economic Sustainability (BEES) database. *Int. J. Life Cycle Assess.* 17, 604-612.

Sun, J., Luo, Q., Wang, D., Wang, Z., 2015. Occurrences of pharmaceuticals in drinking water sources of major river watersheds, China. *Ecotoxicol. Environ. Saf.* 117, 132–140.

Tarpani, R.R.Z., Miralles-Cuevas, S., Gallego-Schmid, A., Cabrera-Reina, A., Cornejo-Ponce, L., 2019. Environmental assessment of multi effect distillation of brackish water with high arsenic and boron levels. Submitted to *J. Clean. Prod.*

Tiedeken, E.J., Tahar, A., McHugh, B., Rowan, N.J., 2017. Monitoring, sources, receptors, and control measures for three European Union watch list substances of emerging concern in receiving waters – A 20 year systematic review. *Sci. Total Environ.* 574, 1140–1163.

Thinkstep, 2017. Extension database XXI: India for Gabi software. Available at: <http://www.gabi-software.com/support/gabi/gabi-database-2017-lci-documentation/extension-database-xxi-india/> (last access: 09/08/2018).

Tong, L., Liu, X., Liu, X., Yuan, Z., Zhang, Q., 2013. Life cycle assessment of water reuse systems in an industrial park. *J. Environ. Manage.* 15 (129): 471-478.

Tran, N.H., Reinhard, M., Gin, K.Y.H., 2018. Occurrence and fate of emerging contaminants in municipal wastewater treatment plants from different geographical regions-a review. *Water Res.* 133, 182–207.

Tsadilas, C.D., Mitsios, I.K., Golia, E., 2005. Influence of biosolids application on some soil physical properties. *Commun. Soil Sci. Plant Anal.* 36 (4-6): 709-716.

Uggetti, E., Ferrer, I., Arias, C., Brix, H., García, J., 2012. Carbon footprint of sludge treatment reed beds. *Ecol. Eng.* 44: 298-302.

UN, 2017. The Sustainable Development Goals Report 2017. United Nations, New York.

UNDP, 2016. Human Developing Report. New York, USA: United Nations Developing Countries.

UNEP, 2016. Opportunities for National Life Cycle Network Creation and Expansion Around the World. United Nations Environment Programme, Paris.

UNICEF, WHO, 2017. Progress on Drinking Water, Sanitation and Hygiene: 2017 Update and SDG Baselines. Geneva, Switzerland: World Health Organization.

UN-Water, 2017a. The United Nations World Water Development Report 2017. Wastewater: The Untapped Resource. UNESCO, Paris.

UN-Water, 2017b. Integrated Monitoring Guide for Sustainable Development Goal 6 on Water and Sanitation – Targets and global indicators. UNESCO, Paris.

Vera, L., Sun, W., Iftikhar, M., Liu, J., 2015. LCA based comparative study of a microbial oil production starch wastewater treatment plant and its improvements with the combination of CHP system in Shandong, China. *Resour. Conserv. Recycl.* 96: 1-10.

Verones, F., Hellweg, S., Azevedo, L.B., Chaudhary, A., Cosme, N., Fantke, P., Goedkoop, M., Hauschild, M.Z., Laurent, A., Mutel, C.L., Pfister, S., Ponsioen, T., Steinmann, Z.J.N., van Zelm, R., Vieira, M., Huijbregts, M. A. J., 2016. LC-impact. Version 0.5. A spatially differentiated life cycle impact assessment approach. Available at: http://www.lc-impact.eu/downloads/documents/LC-Impact_report_SEPT2016_20160927.pdf (last access: 09/08/2018).

Wang, D., Guo, F., Wu, Y., Li, Z., Wu, G., 2018. Technical, economic and environmental assessment of coagulation/filtration tertiary treatment processes in full-scale wastewater treatment plants. *J. Clean. Prod.* 170: 1185-1194.

Wang, X., Liu, J., Ren, N.-Q., Duan, Z., 2012. Environmental profile of typical anaerobic/anoxic/oxic wastewater treatment systems meeting increasingly stringent treatment standards from a life cycle perspective. *Bioresour. Technol.* 126: 31-40.

Wang, X.-H., Wang, X., Huppes, G., Heijungs, R., Ren, N.-Q., 2015. Environmental implications of increasingly stringent sewage discharge standards in municipal wastewater treatment plants: Case study of a cool area of China. *J. Clean. Prod.* 94: 278-283.

Wang, X., Chen, T., Ge, Y., Jia, Y., 2008. Studies on land application of sewage sludge and its limiting factors. *J. Hazard. Mater.* 160 (2-3): 554-558.

WB, 2018. World Development Indicators: Electricity production, sources, and access. Available at: <http://wdi.worldbank.org/tables> (last access: 09/08/2018)

Weidema, B.P., Wesnæs, M.S., 1996. Data quality management for life cycle inventories-an example of using data quality indicators. *J. Clean Prod.* 4 (3-4): 167-174.

Wu, C., Huang, X., Witter, J.D., Spongberg, A.L., Wang, K., Wang, D., Liu, J., 2014. Occurrence of pharmaceuticals and personal care products and associated environmental risks in the central and lower Yangtze river, China. *Ecotoxicol. Environ. Saf.* 106, 19-26.

Wu, H., Zhang, J., Ngo, H.H., Guo, W., Hu, Z., Liang, S., Fan, J., Liu, H., 2015. A review on the sustainability of constructed wetlands for wastewater treatment: Design and operation. *Bioresour. Technol.* 175, 594-601.

Yildirim, M., Topkaya, B., 2012. Assessing environmental impacts of wastewater treatment alternatives for small-scale communities. *Clean Soil Air Water* 40: 171-178.

Yin, L., Wang, B., Yuan, H., Deng, S., Huang, J., Wang, Y., Yu, G., 2017. Pay special attention to the transformation products of PPCPs in environment. *Emerg. Contam.* 3, 69–75.

Yoshida, H., ten Hoeve, M., Christensen, T.H., Bruun, S., Jensen, L.S., Scheutz, C., 2018. Life cycle assessment of sewage sludge management options including long-term impacts after land application. *J. Clean. Prod.* 174, 538–547.

Zang, Y., Li, Y., Wang, C., Zhang, W., Xiong, W., 2015. Towards more accurate life cycle assessment of biological wastewater treatment plants: A review. *J. Clean. Prod.* 107: 676–692.

Zepon Tarpani, R.R., Azapagic, A., 2018. Life cycle environmental impacts of advanced wastewater treatment techniques for removal of pharmaceuticals and personal care products (PPCPs). *J. Environ. Manage.* 215, 258–272.

Zhang, Q.H., Wang, X.C., Xiong, J.Q., Chen, R., Cao, B., 2010. Application of life cycle assessment for an evaluation of wastewater treatment and reuse project - Case study of Xi'an, China. *Bioresour. Technol.* 101 (5): 1421–1425.

Zhang, Z., Wilson, F., 2000. Life-cycle assessment of a sewage-treatment plant in South-East Asia. *Water Environ. J.* 14 (1): 51–56.

Zhang, D.Q., Jinadasa, K.B.S.N., Gersberg, R.M., Liu, Y., Ng, W.J., Tan, S.K., 2014. Application of constructed wetlands for wastewater treatment in developing countries - A review of recent developments (2000–2013). *J. Environ. Manage.* 141, 116–131.

Zhao, X., Yang, J., Ma, F., 2018. Set organic pollution as an impact category to achieve more comprehensive evaluation of life cycle assessment in wastewater-related issues. *Environ. Sci. Pollut. R* 25 (6): 5960–5968.

Zhao, X., Yang, J., Zhang, X., Wang, L., Ma, F., 2017. Evaluation of bioaugmentation using multiple life cycle assessment approaches: A case study of constructed wetland. *Bioresour. Technol.* 244: 407–415.

Water Research 153 (2019) 63-79. <https://doi.org/10.1016/j.watres.2019.01.010>

Zhu, L., Liu, B., Wang, F., Bi, J., 2013. Raising discharge standards leads to environmental problem shifting in China. *Water Sci. Technol.* 68 (12): 2605-2612.