Life cycle analysis and sustainability assessment of advanced wastewater treatment technologies

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Abstract

Purpose – Wastewater treatment plants (WWTPs) have long-time environmental impacts. The purpose of this paper is to assess the environmental footprint of two advanced wastewater treatment (WWT) technologies in a life cycle and sustainability perspective and identify the improvement alternatives.

Design/methodology/approach – In this study life cycle-based environmental assessment of two advanced WWT technologies (moving bed biofilm reactor (MBBR) and sequencing batch reactor (SBR)) has been carried out to compare different technological options. Life cycle impacts were computed using GaBi software employing the CML 2 (2010) methodology. Primary data were collected and analysed through surveys and on-site visits to WWTPs. The present study attempts to achieve significantly transparent results using life cycle assessment (LCA) in limited availability of data.

Findings – The results of both direct measurements in the studied wastewater systems and the LCA support the fact that advanced treatment has the best environmental performance. The results show that the operation phase contributes to nearly 99 per cent for the impacts of the plant. The study identified emissions associated with electricity production required to operate the WWTPs, chemical usage, emissions to water from treated effluent and heavy metal emissions from waste sludge applied to land are the major contributors for overall environmental impacts. SBR is found to be the best option for WWT as compared to MBBR in the urban context. In order to improve the overall environmental performance, the wastewater recovery, that is, reusable water should be improved. Further, sludge utilisation for energy recovery should be considered. The results of the study show that the avoided impacts of energy recovery can be even greater than direct impacts of greenhouse gas emissions from the wastewater system. Therefore, measures which combine reusing wastewater with energy generation should be preferred. The study highlights the major shortcoming, i.e., the lack of national life cycle inventories and databases in India limiting the wide application of LCA in the context of environmental decision making.

Research limitations/implications – The results of this study express only the environmental impacts of the operation phase of WWT system and sludge management options. Therefore, it is recommended that further LCAs studies should be carried out to investigate construction and demolition phase and also there is need to reconsider the toxicological- and pathogen-related impact categories. The results obtained through this type of LCA studies can be used in the decision-making framework for selection of appropriate WWT technology by considering LCA results as one of the attributes.

Practical implications – The results of LCA modelling show that though the environmental impacts associated with advanced technologies are high, these technologies produce the good reusable quality of effluent. In areas where water is scarce, governments should promote reusing wastewater by providing additional treatment under safe conditions as much as possible with advanced WWT. The LCA model for WWT and management planning can be used for the environmental assessment of WWT technologies.

Originality/value – The current work provides a site-specific data on sustainable WWT and management. The study contributes to the development of the regional reference input data for LCA (inventory development) in the domain of wastewater management.

Keywords Sustainability assessment, Life cycle analysis, Moving bed biofilm reactor, Sequencing batch reactor, Wastewater treatment technologies

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1. Introduction

Sustainability is one of the main concerns globally, which is especially true when considering indispensable, broad spectrum commodities such as water (Beery and Repke, 2010). With the increasing world population, as well as industrial and agriculture activities, countries worldwide face growing global water stress, both in terms of water scarcity and deteriorated quality (Zhou *et al.*, 2011). Appropriate treatment of wastewater and further its reuse can help to solve the problem of water scarcity as well as save valuable resources by reducing the use of freshwater.

Although there has been considerable technological advancement, wastewater treatment (WWT) is one of the major issues faced by the developing countries. The main function of a wastewater treatment plant (WWTP) is to produce clean effluent by removing nutrients, metals and organic pollutants present in the influent. Nowadays, WWT has multiple functions and produces both clean effluents and sludge, which is increasingly seen as a resource rather than a waste product and can be used for nutrients and energy recovery. Thus, technological as well as management choices influence the performance of WWTPs. To improve this performance, the trade-offs related to the different choices have to be identified and assessed.

Many advanced technologies such as sequencing batch reactors (SBRs) and membrane bioreactors (MBRs) have been developed for WWT. Generally, the choice of "best" WWT technology is based first and foremost on economic and technical constraints (Bonton *et al.*, 2012). However, climate change, energy crisis, social aspects and other environmental problems coming into focus make this choice more complicated. To provide the foundation for a better choice, information on environmental aspects of different systems is thus needed (Finnveden *et al.*, 2009). Therefore, selection of appropriate technologies for WWT is a prime challenge faced by the decision makers. The decision makers, however, do not have a rational framework to compare WWT technologies. In this context, life cycle assessment (LCA) can determine what choices provide the best environmental performance.

The environmental footprint of a given WWTP depends on the choice of technology because any given treatment technology has a characteristic consumption of resources, energy and chemicals. The footprint is also determined by the treatment objective (e.g. disposal of treated wastewater after mere compliance with the prescribed regulatory norms or production of high-quality water for recycling and reuse applications). There is very limited knowledge available on the performance of existing technologies and evaluation of those systems is very timely in order to derive sound conclusions and recommendations for future wastewater management strategies in India. Further, new technologies that should be introduced in India should be carefully selected, taking into account the already existing experiences.

Therefore, in this study, environmental footprints of two advanced WWT technologies namely: moving bed biofilm reactor (MBBR) and SBR are estimated and have been compared using the life cycle approach. LCA will help to evaluate and to quantify the potential environmental impacts (footprint) due to respective processes and to compare the processes using LCA metrics for each environmental effect.

The remainder of the paper is organised as follows: the following section presents the brief literature review. Section 3 presents the methodology adopted in the study. The results are discussed in Section 4. Finally, Section 5 presents the concluding remarks.

2. Literature review

LCA is becoming increasingly popular amongst researchers in WWT field nowadays because of its holistic approach. LCA has proved as a useful tool for computation of the environmental footprint of a given WWT technology (Hospido *et al.*, 2004; Gallego *et al.*, 2008; Li *et al.*, 2013).

In germane literature, LCA has been mainly used to identify improvement alternatives for a single plant (Hospido *et al.*, 2004; Pasqualino *et al.*, 2009). Previously, several LCA studies examined different aspects of WWT systems such as evaluating the environmental impacts of a single plant (Clauson-Kaas *et al.*, 2001; Pasqualino *et al.*, 2009; Bravo and Ferrer, 2011; Venkatesh and Brattebo, 2011), plant operation, optimisation and modifications (Vidal *et al.*, 2002; Wu *et al.*, 2010), comparison of different competing technologies (Gallego *et al.*, 2008; Meneses *et al.*, 2010; Rodriguez-Garcia *et al.*, 2011; Coats *et al.*, 2011; Ontiveros and Campanella, 2013), WWT modelling (Foley, De Haas, Hartley and Lant, 2010; Foley, Rozendal, Hertle, Lant and Rabaey, 2010; Wang *et al.*, 2012; Rodriguez-Garcia *et al.*, 2012) and significance and dominance of system boundaries on calculated environmental impacts (Lundin *et al.*, 2000). In order to improve performance, few LCA studies compared the performance of a single system with different configurations (Mels *et al.*, 1999; Clauson-Kaas *et al.*, 2004). Foley, De Haas, Hartley and Lant (2010) investigated multiple biological nutrient removal configurations to find the best option, while few others evaluated the single nutrient removal strategy. Ortiz *et al.* (2007) applied LCA to investigate the WWT technology that gives the lowest environmental load employing three different evaluation methods – CML 2 baseline 2000, Eco-Points 97 and Eco-Indicator 99. The results of the study showed that tertiary treatment does not increase significantly the environmental loads but provide new uses for that treated water, thus justifying the intensive use of water reuse techniques in water scarce areas. In some of the LCA studies, assessment and comparison of multiple conventional systems have also been addressed (Gallego *et al.*, 2008; Hospido *et al.*, 2008; Rodriguez-Garcia *et al.*, 2011). LCA has often been used as the comparison tool to assess the differences between conventional and ecological or natural systems and to verify if the latter have lower impacts.

With the increasing need of fresh water and growing, water scarcity problem reclamation of wastewater and its reuse has received much attention in the recent years because of its potential to conserve freshwater and reduce pollutant emissions. However, advanced treatment of wastewater for reuse adds to increased consumption of chemicals, materials and energy, and hence, cost. Nevertheless, despite these drawbacks, advanced treatment technologies play an important role in reducing overall risk and toxicity and are more efficacious in pollutant removal. Consequently, innovative WWT technologies are continuously being developed and compared with conventional technologies using LCA; for instance, microbial electrolysis cells and microbial fuel cells (MFC) (Foley, Rozendal, Hertle, Lant and Rabaey, 2010), technologies based on advanced oxidation processes (Muñoz *et al.*, 2005), MBRs (Hoibye *et al.*, 2008, Wenzel *et al.*, 2008; Foley, De Haas, Hartley and Lant, 2010; Hospido *et al.*, 2012; Remy and Jekel, 2012), ultrafiltration (Ortiz *et al.*, 2007), nanofiltration, and reverse osmosis (Rodriguez-Garcia *et al.*, 2011; Amores *et al.*, 2013; Tong *et al.*, 2013; Alfonsín *et al.*, 2014).

Rahman *et al.* (2016) evaluated three levels of treatment for nutrient removal (N and P) using 27 representative WWT process configurations. Impacts were assessed across multiple environmental and health impacts using LCA employing the tool for the reduction and assessment of chemical and other environmental impacts (TRACI) impact-assessment method. The results showed that although advanced technologies achieve high-level nutrient removal significantly resulting in decreased local eutrophication potential (EP), however, the chemicals and electricity used for these advanced treatments, simultaneously increased eutrophication indirectly and contribute to other potential environmental and health impacts including human and ecotoxicity, global warming potential, ozone depletion and acidification. Fang *et al.* (2016) applied LCA to research and develop a biochemical system for wastewater resource recovery. The key environmental concerns obtained through the LCA were linked to increased human toxicity (HT) impacts from the chosen end use of wastewater recovery products.

Holloway *et al.* (2016) applied LCA to study two potable reuse treatment schemes: a full-advanced treatment (FAT) approach and a hybrid ultrafiltration osmotic membrane bioreactor (UFO-MBR). Results from the LCA illustrated that the energy use and environmental impacts of FAT are lower than those of UFO-MBR treatment. Further, utilising simulation of process optimisation, the environmental impacts of UFO-MBR were brought much closer to those of FAT. Morrison *et al.* (2016) developed a method to account for the upstream and downstream life cycle impacts of WWT at the plant and individual building level using economic input-output LCA, ecologically based LCA, and energy analysis for measuring upstream impacts and process-based method for downstream impact. Piao *et al.* (2016) evaluated several WWTP processes including an integrated sludge management system and waste sludge disposal methods applying LCA and economic efficiency analysis (EEA). The results demonstrated that application of LCA and EEA would be a useful tool for optimising an integrated WWT-sludge-management system. Similarly, Buonocore *et al.* (2016) applied LCA to compare the environmental performance of different scenarios for wastewater and sludge disposal in a WWT plant.

Kamble, Chakravarthy, Singh, Chubilleau, Starkl and Bawa (2017) applied LCA to soil-biotechnology (natural WWT technology) based WWTP to assess the environmental impacts associated with the construction and operation phase of the plant. The results of the study indicated that natural treatment technologies are the best options to treat wastewater due to its low cost, low energy demand, simple operation, minimum maintenance, low noise and free of odour. Garfí *et al.* (2017) assessed the environmental impact of three alternatives for WWT in small communities. An LCA was carried out comparing a conventional WWTP (i.e. activated sludge system) with two nature-based technologies (i.e. hybrid constructed a wetland and high rate algal pond systems). Moreover, an economic evaluation was also conducted. The results showed that the natural WWT solutions were the most environmentally friendly options, while the conventional WWTP presented the worst results due to the high electricity and chemicals consumption. Bai *et al.* (2017) investigated whether the environmental impact assessment of WWTPs varies with different LCA methods using a generic LCA method, CML and a China-specific method, e-Balance. The study specifically examined the environmental impacts and compared four effluent treatment levels: no treatment, basic treatment, intermediate treatment and tertiary treatment. The LCA results revealed great variation between the no treatment scenarios and between the tertiary treatment scenarios. Lutterbeck *et al.* (2017) investigated the performance of a WWT system with constructed wetland in a rural scenario using LCA. The results of the study showed that the application of LCA can give valuable insights for setting the best configurations for a WWT system in rural areas by identifying the most critical parameters and by the evaluation of actions which might reduce the environmental impacts. Corbella *et al.* (2017) assessed the environmental impact of MFCs implemented in constructed wetlands using the CML-IA baseline method. The study results showed that the environmental impacts of the system under study were higher and the cost was around 1.5 times more expensive than the conventional constructed wetland system. Kamble, Singh and Kharat (2017) developed hybrid LCA-based fuzzy multi-criteria decision-making model for the evaluation and selection of appropriate municipal WTT technology. The results of LCA were used to formulate sustainability and environmental criteria for selecting the most appropriate technology. However, the single LCA studies and the comparative LCA studies between different treatments plants present in the literature are still insufficient for business, public administrators and policy makers.

Moreover, with the increasing use of LCA, several LCA studies on WWT have also reported problems associated with data availability and data quality in the life cycle inventory (LCI) phase (Corominas *et al.*, 2013), and used secondary data to model effluent emissions (Foley, De Haas, Hartley and Lant, 2010).

3. Methodology

In the Indian context, there are no national databases available for carrying out LCA and it becomes rather difficult to generate a material or emission inventory. The standard methodology as prescribed in ISO 14040-44 series (International Organization for Standardization, 2006a, b), is employed for conducting the LCA in this study. The ISO 14040-44 (International Organization for Standardization, 2006a, b) standard determines four stages for LCA studies as follows:

(1) Goal and scope definition

Goal and scope definition constitutes the first phase of an LCA and aims at defining the boundaries of the study and the quality of data used. A functional unit (FU), which represents the function of the system under study, must be also established in this phase.

(2) LCI

In the second stage, LCI is performed, which involves data collection and interpretation of inputs and outputs. The allocation procedure is also conducted during the LCI phase, which consists of distributing input and output flows among the process.

(3) Life cycle impact assessment (LCIA)

LCIA represents the third phase and its purpose is to convert LCI data into potential impacts associated with products and processes. LCIA includes two mandatory steps (i.e. classification and characterisation) and other optional elements, such as normalisation and weighting.

(4) Interpretation of results

Finally, the interpretation of the results allows identifying the hot spots of the process as well as recommending options to reduce the environmental burdens.

3.1 Description of WWTPs under the scope of this study

3.1.1 MBBR. MBBR technology employs thousands of polyethylene biofilm carriers operating in mixed motion within an aerated WWT basin. Each individual biocarrier increases productivity through providing protected surface area to support the growth of heterotrophic and autotrophic bacteria within its cells. It is this high-density population of bacteria that achieves high rate of biodegradation within the system, while also offering process reliability and ease of operation, thus, greatly reducing the organic load of the effluent. In the current study, a small scale 2 MLD MBBR plant was selected for the life cycle analysis. MBBR technology provides cost-effective treatment with minimal maintenance since MBBR processes self-maintain an optimum level of productive biofilm. Additionally, the biofilm attached to the mobile biocarriers within the system automatically responds to load fluctuations. The MBBR plant under study with system boundaries is shown in Figure 1.

3.1.2 SBR. SBR technology is emerging as a promising advanced technology in India. It is most suitable in the urban settlements owing to its lower land requirement compared with conventional systems. Also, SBR can achieve good nutrient removal with minor design changes. In an urban area where surface water bodies are under stress and cannot take any more nutrient load, SBR is the best possible solution. This brings the need to compare this new system with conventional systems and other competing technologies. Figure 2 shows the treatment scheme of typical SBR plant under study. In this study, a small scale, 3 MLD (million litres per day) capacity SBR plant designed for greater organic as well as nutrient removal was selected.

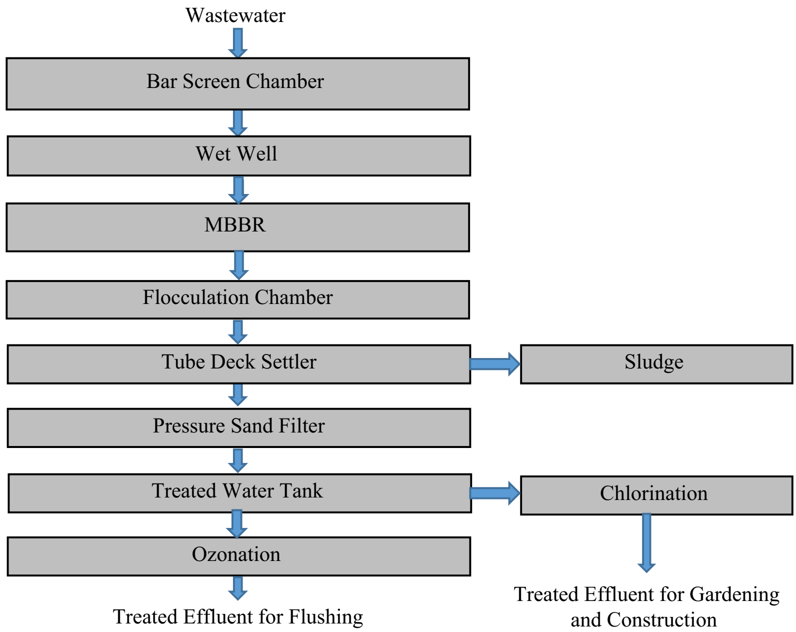

Figure 1.
Flow diagram of MBBR plant

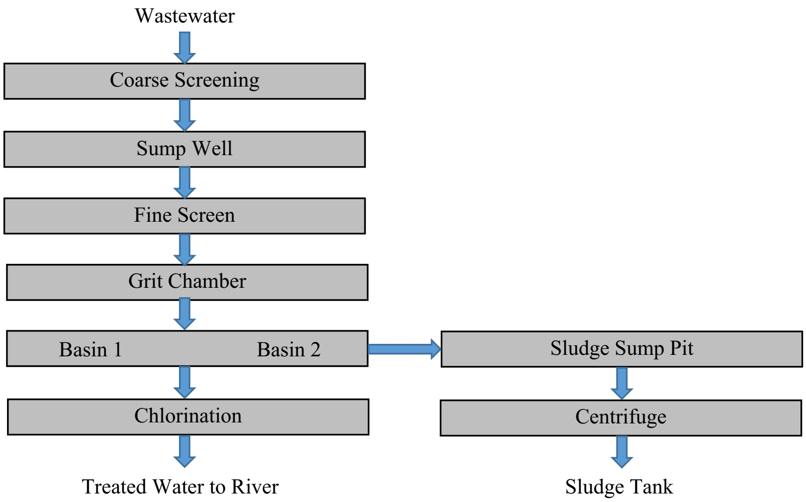

Figure 2.
Flow diagram of SBR plant

Table I shows the water quality parameters for both the plants, while Figure 3 shows the graph presenting a comparison of the parameters.

3.2 Assumptions made for LCA and technology assessment

In order to perform technology assessment, it is essential to bring all the technologies on a common platform, so that there can be parity in comparison. On the basis of expert's opinion

Parameters	Units	MBBR Influent	MBBR Effluent	SBR Influent	SBR Effluent
pH		6.9	7.6	7.351	7.635
Turbidity	NTU	78	11.0	111.9	7.75
Conductivity	μS/cm	542	908.4	767.67	608
TSS	mg/ltr	141	43	308.5	25.267
VSS	mg/ltr	82	12	154.3	12.333
BOD	mg/ltr	111	13	153.5	10.433
COD	mg/ltr	288	47	439.5	42.333
TKN	mg/ltr	37	31.4	37.98	7.0833
N-NH_4^+	mg/ltr	30	26.3	36.07	2.925
PO_4^3	mg/ltr	3.2	0.8	3.12	1.7967
T-P	mg/ltr	–	–	4.417	2.3833
Total alkalinity	$CaCO_3$ mg/ltr	161	241.8	321.83	257

Table I.
Wastewater quality parameters of MBBR and SBR plants

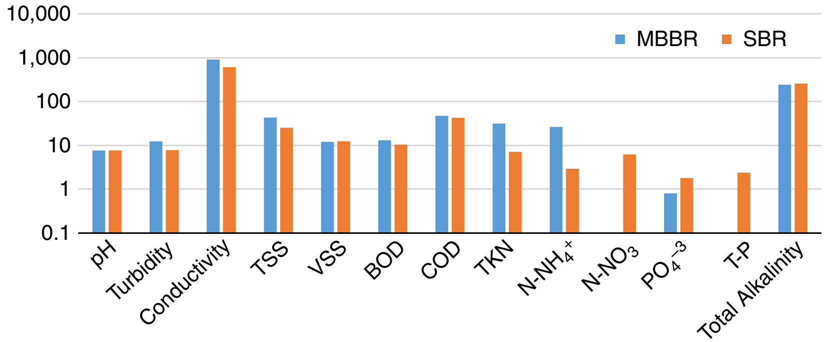

Figure 3.
Effluent quality parameters

and depending on the aim of the research work, the following assumptions were made in this study for technology comparison:

(1) Energy consumption for the pumping of sewage to the plant was not considered because of variation in the pumping distances at each location, which may affect results of the study.

(2) The inlet biochemical oxygen demand (BOD) for both the treatment plants was assumed to be 200 mg/L, which is the average BOD_5 value in India (CPCB, 2009).

(3) Globally it is accepted and proven in many LCA studies that primary processes have major contribution in the impacts over a life cycle and secondary processes such as construction of plant and production of chemicals, etc. and demolition phase is estimated to contribute less than 1 per cent of the total impacts of the treatment plant (Emmerson *et al.*, 1995; Gaterell *et al.*, 2005; Hospido *et al.*, 2008).

(4) A similar approach is used in this study where only operation phase (primary processes – electricity production, emissions to air, water and soil from treatment plant) is considered, as the main focus is on technology assessment.

(5) Chemical production has not been included in the analysis as the production of chemicals will not influence the results of this comparative assessment study (Gaterell *et al.*, 2005; Kalbar *et al.*, 2014).

(6) In this study, the best suitable sludge management options for each of the technology were evaluated, and accordingly, system boundaries were decided. This assumption makes technology assessment truly unbiased, and due credit is given to intrinsic properties of the technology.

(7) Sludge transportation distance was assumed to be 50 kms for both the WWTPs.

3.2.1 Goal and scope. The goal of this LCA study was to evaluate the environmental footprints (impacts) of two WWTPs which operate using different technologies. This was carried out with the help of GaBi software. The scope of the study includes the operation and maintenance (O&M) phase of WWTPs.

3.2.2 System boundaries. System boundaries considered for LCA affect largely on the final results and hence shall be judiciously selected (Tillman *et al.*, 1994). Studies on LCA of WWTPs have shown that construction and demolition phases of WWTPs have negligible impacts (about 1 per cent of impacts compared with overall life cycle impacts of the WWTP) compared with the operation phase (Emmerson *et al.*, 1995; Tillman *et al.*, 1998; Lundin *et al.*, 2000; Karrman and Jonsson, 2001; Gaterell *et al.*, 2005; Machado *et al.*, 2007). Therefore, this study focussed on the O&M phase of both the WWTPs in accordance with the final aim of the research work. In consequence, the assessment was carried out considering the environmental impact associated with the operational phase of the primary treatment, secondary treatment, tertiary treatment and the sludge treatment.

Energy and chemicals required for operation of the plant and emissions during the O&M phase were taken into account in this study. Process emissions that are biogenic in nature (i.e. CO_2 associated with microbial activity in the treatment reactors) are excluded from the analyses because they belong to the short CO_2 cycle and do not contribute to climatic change (Hospido *et al.*, 2008; Coats *et al.*, 2011). This approach is in agreement with similar studies on WWTPs assessment (Gallego *et al.*, 2008; Hospido *et al.*, 2008).

3.2.3 FU. In the current study, 1 m^3 of treated wastewater was chosen as FU which is most commonly used FU in similar studies (Clauson-Kaas *et al.*, 2001; Hospido *et al.*, 2004; Pasqualino *et al.*, 2009; Venkatesh and Brattebo, 2011; Roushdi *et al.*, 2012; Rodriguez-Garcia *et al.*, 2011; Corominas *et al.*, 2013; Niero *et al.*, 2014). This FU will be helpful for making a comparison of different WWTP using different technologies.

3.2.4 LCI. Following the goal and scope definition, LCI analysis was conducted regarding mainly materials and energy consumption in the inputs items and outputs items (Table II). Life cycle inventories were generated based on several on-site visits to WWTPs. Energy and chemicals are primarily used in the O&M phase. In this study, it is assumed that all the power generation is from coal-based thermal power plants. The data of energy consumption and chemicals used in the operation of plants per day were collected and later converted as per the FU (Table II).

3.2.5 LCIA. Impact assessment is an important step in measuring the environmental impacts in LCA. GaBi software comes with a large number of standard impact assessment methods. In this study, CML 2001 (November – 2010) method was used for LCIA using GaBi v. 6.2 software.

The impact assessment phase of the LCA is comprised of mandatory elements, namely, selection of impact categories, classification (assignment of the inventory data to the chosen impact category), characterisation (calculation of impact categories using characterisation factors), as well as optional elements, namely, normalisation (calculation of category indicator results relative to reference value(s)) and grouping and/or weighting the results (Pennington *et al.*, 2004). The current study does not include the optional elements (normalisation and grouping and/or weighting) because there are no reference values available due to the lack of LCA studies in the Indian context.

Sr. no.	Parameter	Unit	MBBR	SBR
1	Electricity consumption	MJ	2.51	0.913
2	Chemical consumption			
2.1	Alum	kg	0.00553	–
2.2	Lime	kg	0.0502	–
2.3	Sodium hypochlorite	kg	0.052	0.02
3	Emissions to air		11.6	4.26
3.1	CO_2	kg	1.19	0.43
3.2	SO_2	g	0.00642	0.00225
3.3	NO_X	g	0.0046	0.00167
3.4	CO	g	0.000648	0.000234
3.5	Heavy metals	g	7.42E-006	2.69E-006
3.5.1	Zinc	g	2.41E-006	8.76E-007
3.5.2	Tin	g	2.41E-007	8.75E-008
3.5.3	Nickel	g	3.92E-007	1.4E-007
3.5.4	Lead	g	9.63E-007	3.5E-007
3.5.5	Copper	g	1.24E-007	4.49E-008
3.5.6	Cobalt	g	7.41E-008	2.69E-008
3.5.7	Chromium	g	1.92E-010	6.77E-011
3.5.8	Cadmium	g	6.82E-008	2.48E-008
3.5.9	Arsenic	g	4.24E-007	1.54E-007
4	Emissions to water		2.86E003	1.04E003
4.1	COD	kg	0.000511	0.000188
4.2	N-Total	kg	2.71E-009	9.85E-010
4.3	P-Total	kg	9.1E-007	3.49E-007
4.4	Heavy metals	g	0.000761	0.000278
4.4.1	Zinc	g	7.99E-008	2.88E-008
4.4.2	Tin	g	9.71E-015	3.51E-051
4.4.3	Nickel	g	7.69E-008	2.37-008
4.4.4	Lead	g	4.11E-008	1.26E-008
4.4.5	Copper	g	6.66E-008	1.19e-008
4.4.6	Cobalt	g	3.15E-010	1.21E-010
4.4.7	Chromium	g	4.05E-009	1.51E-009
4.4.8	Cadmium	g	1.76E-008	2.95E-009
4.4.9	Arsenic	g	4.68E-008	9E-009
5	Emissions to soil		4.47E-006	1.63E-006
5.1	Heavy metals	g	4.47E-006	1.63E-006
5.1.1	Zinc	g	1.65E-006	6.02E-007
5.1.2	Nickel	g	1.21E-008	4.55E-009
5.1.3	Lead	g	1.08E-007	3.96E-008
5.1.4	Copper	g	5.94E-007	2.16E-007
5.1.5	Chromium	g	2.07E-006	7.55E-007
5.1.6	Cadmium	g	1.38E-008	5.05E-009
5.1.7	Mercury	g	2.08E-008	7.57E-009

Table II. Summary of the O and M phase life cycle inventory for inputs and outputs of wastewater treatment plants (FU – 1 m^3 of wastewater treated)

Impact categories in this study were selected based on data availability and significance of a particular impact category with respect to the goal of the study. Life cycle impacts were computed using CML 2001 methodology, developed by Centre of Environmental Science (CML), University of Leiden, the Netherlands, which gives a separate score for each type of environmental impact. Thus, the potential impacts of each category were estimated.

Eight impact categories, namely, global warming potential (GWP), ozone layer depletion potential (ODP), EP, ecotoxicity (terrestrial, marine, freshwater), HT, photochemical ozone creation potential (POCP), acidification potential (AP) and Abiotic resources depletion potential (ADP – elements, fossil) were considered. The description of different impact categories considered is presented in Table III.

Impact category/ indicator	Units	Description
Acidification potential (AP)	kg SO_2 eq	Contribution from substances that produce sulphuric acid when they are in contact with water. When these substances are present in the environment they produce acid rain, causing terrestrial and aquatic species to degrade
Global warming potential (GWP)	kg CO_2 eq	The contribution of the various emissions that causes an increase in global warming. The most important substances are CO_2, CH_4, N_2O, and the halogenated hydrocarbons
Eutrophication (EP)	kg PO_4 eq	The contribution of the various emissions to the accumulation of nutrients in the environment. When nutrients accumulate in aquatic ecosystems, plant growth increases and deplete oxygen levels
Photochemical oxidation (PHO)	kg formed ozone	The contribution of the various emissions to the formation of photo-oxidant substances (particularly ozone and peroxyacetyl nitrate) through the photochemical oxidation of volatile organic substances and carbon monoxide
Depletion of abiotic resources (DAR)	kg antimony eq	The contribution of the various emissions to the extraction of resources, including their availability, energy content, concentration, and rate of use
Ozone depletion potential (ODP)	kg CFC-11 eq	The contribution of substances that deplete the ozone stratospheric layer. The most important substances are chlorated and bromated halocarbons, particularly trichlorofluoromethane (CFC-11, also known as freon-11)
Ecotoxicity potential (ETP)	kg 1,4-DCB eq	Combined result of freshwater aquatic and sediment ecotoxicity, marine aquatic and sediment ecotoxicity, human toxicity, and terrestrial ecotoxicity. These substances affect the health of humans, flora, and fauna in the different environments. The most important substances are heavy metals, persistent organic pollutants and volatile organic compounds

Table III.
Description of impact categories

4. Results and discussion

The results obtained from the study are presented and discussed in the subsequent sections. The results of LCIA of WWTPs for different selected impact categories are presented in Table IV. The comparative assessment of both the plants is shown in Figure 4.

LCA results of the MBBR and SBR technologies are based on data collected from actual plants. Table II shows the LCI for both the WWTPs, while Table IV shows the results of life cycle impacts in various categories of WWTPs for the O&M phase.

As stated earlier, the LCIs in this study address the primary, secondary, tertiary and sludge treatment processes. The inventory, however, does not incorporate pumping of

Impact category	Unit	MBBR	SBR
Abiotic depletion (ADP elements)	Kg sb-Equiv	1.48E-006	5.75E-007
Abiotic depletion (ADP fossil)	MJ	13.9	5.06
Acidification potential (AP)	Kg SO_2-Equiv	0.0102	0.00362
Eutrophication potential (EP)	Kg phosphate-Equiv	0.000631	0.000229
Freshwater aquatic ecotoxicity potential (FAETP)	Kg DCB-Equiv	0.00348	0.0012
Global warming potential (GWP 100 years)	Kg CO_2-Equiv	1.24	0.448
Human toxicity potential (HTP)	Kg DCB-Equiv	0.387	0.14
Marine aquatic ecotoxicity potential (MAETP)	Kg DCB-Equiv	1.79E003	651
Ozone layer depletion potential (ODP)	Kg R11-Equiv	3.83E-011	1.32E-001
Photochemical ozone creation potential (POCP)	Kg Ethene-Equiv	0.000526	0.000185
Terrestrial ecotoxicity potential (TEP)	Kg DCB-Equiv	0.0172	0.00626

Table IV.
LCIA results of MBBR and SBR plants for various impact categories

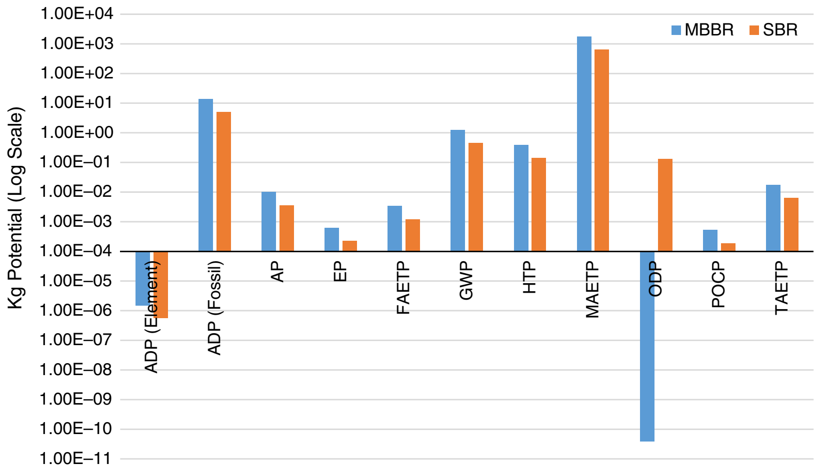

Figure 4.
Comparative results of LCIA of both the WWTPs

sewage to the primary treatment facility. The results are reliable owing to the primary data collection exercise carried out for generating LCI.

Impacts from both the treatment system are mainly caused by the use of electricity required to operate the WWTPs, emissions to water from treated effluent and heavy metal emissions from waste sludge are identified as main contributors for overall environmental impacts of WWTPs.

4.1 Energy consumption

The total energy consumption over the life cycle of the plant has been found to be 0.70 kWh/m^3 for MBBR and 0.25 kWh/m^3 for SBR. The results obtained are comparable with the results (0.1 kWh/m^3-1.5 kWh/m^3) reported by Pasqualino *et al.* (2011), Rodriguez-Garcia *et al.* (2011), Amores *et al.* (2013) and Niero *et al.* (2014).

4.2 GWP

Energy consumption for the operation of WWTPs is found to be the largest contributing parameter for CO_2 emissions and GWP. The GWP for MBBR (1.24) is found to be higher than SBR (0.448), as MBBR is more energy consuming than SBR. Further, the CO_2 emission from transportation is found to be negligible, i.e., less than 1 per cent of the total impact.

4.3 AP and ADP

AP is mainly because of SO_2 and NO_x emissions from coal combustion, which generates electricity for operating the plants (MBBR: 0.0102 and SBR: 0.00362). Coal consumption has also a major contribution in ADP (fossil). MBBR (13.9) is found to have the higher ADP (fossil) as compared with SBR (5.06).

4.4 Eutrophication (EP)

For eutrophication, the dominant factors are the total nitrogen and total phosphorus and to a lower extent COD contained in the effluent. The SBR (0.00029) has low EP value as compared to MBBR (0.000631), which matches with values (for nutrient removing systems) reported by Gallego *et al.* (2008). This lowest EP of SBR is because of its intrinsic nutrient removal (<80 per cent) capacity. It is important to note that reduction in the eutrophying load of the effluent was attained in both the WWTPs such that the effluent meets the standards defined by existing regulations.

4.5 ODP

The ODP for MBBR is found to be very negligible and for SBR it is found to be (1.32E-001). Emission of gases that reduce the ozone layer (principally CFC-11, CFC-12 and Halon 1301) is minimal.

4.6 POCP

Among the impact categories typically considered by LCA studies, Lundie *et al.* (2004) listed POCP as one of the categories of relevance to the water industry. According to Karrman and Jonsson (2001), 100 g of ethene equivalents are emitted per person equivalent when using a conventional system for WWT. The values reported here are far smaller, being 0.000526 for MBBR and 0.000185 for SBR.

4.7 Toxicity potentials (ecotoxicity (FAETP, MAETP, TE) and human toxicity potential)

Toxicity potentials are measured in terms of kilogram equivalent of dichlorobenzene (DCB). Ecotoxicity potential is mostly dependent on the heavy metals released in the water and soil environment from the WWTP, for which in the considered WWTPs there is no special provision for heavy metal removal, however, some removal takes place through the physico-chemical and biological processes of the WWTPs. Further, the disposal of sludge containing heavy metals contributed substantially to the ecotoxicity (FAETP, MAETP and TE) and HT impact categories. During the field study, it was observed that the treated sludge is sent for land application, which is the main cause of the TE, dominated by the presence of heavy metals in the sludge being Zn, Pb and Cu as the main contributors. This contribution is directly dependent on the sludge production.

TE for MBBR is found to be 0.0172 and for SBR it is 0.00626. In the current study, FWAT and MAET are primarily because of heavy metals released from the treated wastewater into natural water bodies. FAETP for MBBR is 0.00348, which is found to be higher than SBR, 0.0012. The significant difference is observed for MAETP, for MBBR (1.79E003) it is found to be very high as compared to SBR (651). The study reveals that MAETP contributes most to the overall impacts accounting for more than 97 per cent, the result is in agreement with the results of earlier studies (Kalbar *et al.*, 2014; Zhao *et al.*, 2015). This result is not surprising considering that both the technologies are not designed to remove heavy metals. HT is because of the release of heavy metals in water, air and the soil environment (MBBR-0.387, SBR-0.14).

These results suggest that toxicity potentials by themselves do not tell the whole story of impacts; the environment in which the treated effluent is released (species diversity, sensitive population, etc.) has a distinct role in defining the actual impact. Thus, the results highlight the need to revisit the characterisation models dealing with toxicological impact categories (Larsen *et al.*, 2010).

Removal efficiencies of the SBR are greater than MBBR and hence the effluent quality is typically better. In sum, it can be said that the impact of WWTP is more dependent on design and how a plant is operated. Even for similar technologies, there can be a huge difference in the performance depending upon the operation of the plant; this fact has already been reported by Gallego *et al.* (2008).

The results of LCIA show that any given technology may perform well in one particular impact category and poorly in another impact category, and hence, it is difficult to compare the technologies based only on the impact categories. Further, there are many issues like the effect of scale, operating conditions, technology design, the capability of technology to remove particular pollutants and regional as well as local priorities to be resolved before judging the overall performance of any technology. LCIA methodologies like CML 2001 (mid-point approach) and Eco-indicator 99 (end-point approach) are not designed to capture such technological or socioeconomic-specific issues (Bare *et al.*, 2000). Also, local and regional priorities cannot be accommodated in the present LCIA methodologies.

5. Conclusions

The life cycle approach was applied in this study to obtain environmental footprint of the WWTPs. The results obtained are comparable with published literature. This study illustrates that the LCA can be applied productively to obtain transparent results with the help of detailed primary data collection pertaining to a specific case study to add to Indian inventory. Selection of appropriate WWT technologies is essential for developing countries like India to be able to manage waste in a sustainable manner. The results of LCA can be used in the decision-making framework developed for the selection of appropriate WWT technology by considering LCA results as one of the attributes along with other attributes. Finally, it is necessary for a country like India to create its own LCIs to improve the environmental decision-making process and the present study is the nascent step towards this goal.

The LCA approach employed in the current study can be generally applied for comparing the life cycle impacts associated with various WWTPs operating using different WWT technologies. The LCA methodology used in the current study is as per the ISO 14040 series (International Organization for Standardization, 2006a, b). However, in order to perform other technology assessment and their comparison, it is essential to bring all the technologies on a common platform, so that there can be parity in comparison. Few fundamental assumptions need to be made for this purpose as discussed below.

The inputs/outputs, that is, the average values should be calculated using data from different treatment facilities employing different WWT technologies. This approach allows taking the variability of the WWT process into account. System boundaries for the LCA affect largely on the final results and hence shall be judiciously selected. Selection of FU should be made keeping in mind the variability of WWT technologies. Further, the choice of the LCIA method should be done keeping in mind the aim of conducting LCA as it has an influence on the final results, that is, whether the aim is to achieve mid-point impacts or the end-point impacts. All the WWT technologies should be assessed and compared on an equivalent basis. For example, if three technologies evaluated use secondary treatment, then it is necessary to transform the tertiary treatment of fourth WWT technology to an equivalent secondary treatment system so that all the four technologies can be compared.

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