Net environmental benefit: introducing a new LCA approach on wastewater treatment systems

D. Godin, C. Bouchard and P. A. Vanrolleghem

ABSTRACT

Life cycle assessment (LCA) allows evaluating the potential environmental impacts of a product or a service in relation to its function and over its life cycle. In past LCAs applied to wastewater treatment plants (WWTPs), the system function definition has received little attention despite its great importance. This has led to some limitations in LCA results interpretation. A new methodology to perform LCA on WWTPs is proposed to avoid those limitations. It is based on net environmental benefit (NEB) evaluation and requires assessing the potential impact of releasing wastewater without and with treatment besides assessing the impact of the WWTP's life cycle. The NEB allows showing the environmental trade-offs between avoided impact due to wastewater treatment and induced impact by the WWTP's life cycle. NEB is compared with a standard LCA through the case study of a small municipal WWTP consisting of facultative aerated lagoons. The NEB and standard LCA show similar results for impact categories solely related to the WWTP's life cycle but differ in categories where wastewater treatment environmental benefit is accounted for as NEB considers influent wastewater quality whereas standard LCA does not.

Key words | facultative aerated lagoons, greenhouse gas, heavy metals, life cycle assessment

D. Godin (corresponding author)
C. Bouchard
P. A. Vanrolleghem
Département de génie civil et de génie des eaux,
Université Laval,
1065 Avenue de la médecine,
Québec, G1V 0A6,
Canada
E-mail: *david.godin.1@ulaval.ca*

INTRODUCTION

Life cycle assessment (LCA) is a method which allows assessing the environmental impact of a product or a service in relation to its function and over its whole life cycle. The general LCA methodology is set by ISO 14040 standards (ISO 2006) dividing it in four steps: goal and scope definition, inventory analysis, impact assessment and interpretation. In LCA's first step, the function must be clearly defined considering that the whole environmental impact assessment is conducted in relation to the function and that it serves as a basis to settle two elements of prime importance for the assessment: the functional unit and the system boundaries. In past LCAs conducted on wastewater treatment plants (WWTPs) that tackled directly the function's definition, the main function of a WWTP was defined as the removal of pollutants present in water so that it reaches satisfactory values before release in natural water courses (Hospido *et al.* 2004; Gallego *et al.* 2008). However, this definition implies several limitations in the interpretation of LCA results because it allows neither taking into account the influent water quality nor considering the treatment efficiency, whereas these are two key criteria to assess the environmental performance of a WWTP. To compensate for these limitations, some LCAs took indirectly into account the influent quality by setting the functional unit in terms of person equivalents (p.e.) served by the WWTP (Tillman *et al.* 1998; Machado *et al.* 2007; Gallego *et al.* 2008) or more directly by defining some influent quality parameters (Foley *et al.* 2010). The aforementioned function's definition also associates the environmental impact of the pollution entering the WWTP solely to this element of the urban water cycle even though it does not generate this pollution itself. This last remark is confirmed by LCAs studying the whole urban water cycle which concluded that the WWTP is an important contributor to the global environmental load of the whole urban water system basically because of the residual load rejected in receiving water after treatment (Lundie *et al.* 2004; Lassaux *et al.* 2007). In a first attempt to avoid the aforementioned limitation regarding LCA function definition and interpretation, we developed a comparative LCA between a WWTP scenario and a null option scenario to explicitly assess the environmental trade-off provided by the WWTP. This trade-off is expressed by the net environmental benefit (NEB) concept introduced in this paper.

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METHODOLOGY

Sewer systems allow collecting and transporting wastewater away from human living areas. These sewer systems are most of the time connected to a WWTP partially removing the pollutant loads to protect the receiving aquatic ecosystems and, more or less directly, human health. In this context, the main function of the WWTP is to limit the environmental damage of wastewaters on the receiving water body. With the growing concern for environmental sustainability, the impacts generated by the WWTP itself should also be taken into account. Therefore, the question which arises is: what is the NEB provided by a WWTP? Answering this question requires conducting a comparative LCA between a null option and a wastewater treatment option, the null option consisting of rejecting wastewater directly into the receiving water body, i.e. without any treatment. The NEB therefore corresponds to the difference between avoided and induced potential environmental impacts (PI) of the WWTP and is computed through the following equation:

$$NEB = [PI_{NO} - PI_{TW}] - PI_{SLC} \tag{1}$$

where $[PI_{NO} - PI_{TW}]$ corresponds to the avoided potential impact due to wastewater treatment, i.e. the difference between the untreated wastewater impact, corresponding to the null option scenario (PI_{NO}), and the treated wastewater impact (PI_{TW}). The induced impact (PI_{SLC}) corresponds to the potential impact of the resources extracted and the emissions generated by the life cycle of the WWTP. This equation applies to each impact that is assessed through a LCA (greenhouse gas emissions, eutrophication, etc.) as detailed below. In comparison with the NEB approach, a standard LCA of a WWTP would provide results as the sum of PI_{TW} and PI_{SLC} which leads to associating the impact of incoming wastewater solely to the WWTP and does not allow assessing its environmental performance. It is worth mentioning that NEB has already been assessed in the field of waste treatment technology for municipal solid waste (Hellweg *et al.* 2005). However, the assessment for solid waste was simplified by comparing two treatment scenarios which excluded the assessment of the null option that was similar between scenarios. In our case, the null option scenario allows taking explicitly into account the influent wastewater quality. The null scenario sets a baseline and helps clarifying the advantages and disadvantages of the proposed WWTP (Reap *et al.* 2008). The null scenario was used in a LCA conducted on a WWTP by Roeleveld *et al.* (1997) to show its unsustainability on a country scale compared with other treatment scenarios. Foley *et al.* (2010) also included a no-treatment scenario in their study as this situation still occurs in developing countries.

Considering that NEB defines the main function of the WWTP as providing a net environmental benefit and requires conducting a comparative LCA between a null option and a WWTP scenario, the system boundaries of the LCA should be defined accordingly. Regarding the inventory, the same pollutants should be tracked in the influent, effluent and in the WWTP removal process to avoid skewing the assessment. If this is not done, the environmental trade-off will not be assessed accurately. This happened in past WWTP LCAs when the heavy metal content of treatment sludge was regarded but not the heavy metals content of the effluent (Emmerson *et al.* 1995; Hospido *et al.* 2004; Gallego *et al.* 2008). Excluding raw and treated wastewater heavy metal loads from the assessment leads to the conclusion that WWTPs with low heavy metal removal efficiency are environmentally superior, regarding toxicity related impact categories, compared with high heavy metal removal efficiency's WWTP leading to high metal loads in treatment sludge.

Case study application

The NEB approach is illustrated through the case study of a WWTP of a small city in the Province of Quebec, Canada. The treatment consists of four facultative aerated lagoons in series equipped with static aerators. The first three lagoons mainly aim at removing organic matter whereas the last one serves as a polishing pond. From May to November, phosphorus removal is achieved by dosing aluminum sulphate at the inlet of the last lagoon. The lagoons have a total operating volume of 215,700 m^3, an operating water depth of 4.6 m, occupy an area of 10 ha and serve 10,500 person equivalents. Treated wastewater is rejected in a tributary of the St-Lawrence River. The WWTP was designed considering that most of the organic and particulate loads come from food industries and more than half of the design flow of 8,750 m^3 comes from infiltration caused by the sewer being in a high level water table area. From a LCA perspective, the boundaries of the system are set according to the NEB approach. As shown in Figure 1, the case study is limited to the operational phase. As mentioned by Rodriguez-Garcia *et al.* (2011), although the construction phase was found responsible for 25 to 35% of the global warming potential (GWP) in past

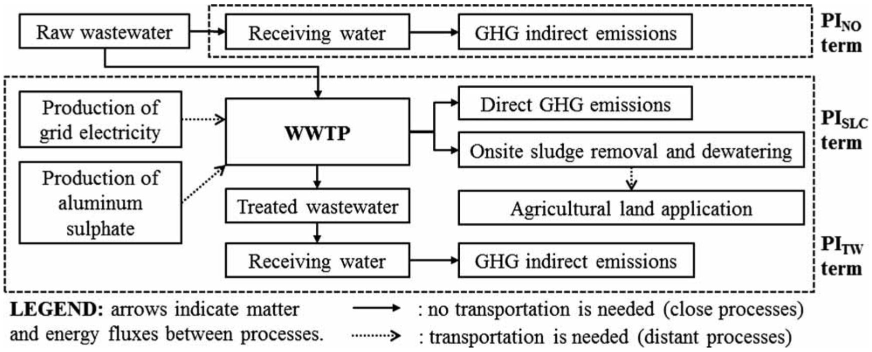

Figure 1 | System boundaries for the null option and for the treatment option.

LCA studies conducted on intensive technology systems (i.e. activated sludge and water recycling technology), the operational phase of these systems was far more relevant for the rest of the impact categories. Considering that lagoons are low technology systems that require far less resources to be built, the construction phase was not taken into account in this first application of the NEB concept.

The functional unit has been set as 1 m^3 of raw wastewater entering the WWTP.

WWTP operation and sludge production

Average organic matter and nutrients load corresponding to the functional unit are shown in Table 1. Characterization data for organic matter (COD, BOD_5), total suspended solids (TSS) and pH and effluent NH_4^+ come from the 2007, 2008 and 2009 WWTP regular monitoring data. Influent nutrient loads are estimated based on unit loads of 5 g N-Norg*$pers^{-1}$*d^{-1}, 5 g N-NH_4^+*$pers^{-1}$*d^{-1} and 2 g Ptot*$pers^{-1}$*d^{-1} (MDDEP 2011). The sludge nutrient content comes from sludge disposal reports. These measured sludge nitrogen and phosphorus loads are considered to allow estimating removal and effluent loads by assessing the mass balance over the system. The nitrogen mass balance is made over the system taking into account that the WWTP is not designed for nitrogen removal (i.e. the denitrification process is regarded as negligible). Nitrogen enters the system as Norg and NH_4^+ and is considering leaving in the effluent only as NH_4^+ and NO_3^- given the WWTP provides sufficient aeration for nitrification during summer. Due to the 10 years sludge retention, the assimilation of nitrogen corresponds to the Norg content of the sludge. The effluent organic nitrogen is neglected given the low TSS load of the effluent. Given the large uncertainty surrounding N_2O emissions that are shown in Table 4, the N_2O emissions have been excluded from the nitrogen mass balance. Values of Table 1 were calculated per m^3 of raw wastewater considering the total load and treated wastewater volume during the 2007–2009 period during which the mean daily flow was 7873 m^3*d^{-1}.

Table 1 | Average load corresponding to 1 m^3 of raw wastewater

Parameter	Units	Average load Influent	Effluent	Sludge	Data source
COD	g COD/m^3	480	57	–	Operating data from 2007, 2008 and 2009 WWTP monitoring reports.
BOD_5	g BOD_5/m^3	220	12	–	
TSS	g TSS/m^3	140	14	–	
pH	–		7.4	–	
Total phosphorus (Ptot)	g P/m^3	2.7	2.2	0.5	• Influent data estimated from unit load (MDDEP 2011).
Ammonium ion (NH_4^+)	g N-NH_4/m^3	6.6	9.3	0.1	• Effluent data estimated by mass balances except for NH_4^+ coming from monitoring reports.
Organic nitrogen (Norg)	g N/m^3	6.6	–	0.3	• Sludge disposal report.
Nitrate (NO_3^-)	g N-NO_3^-/m^3	–	3.5	–	

Table 2 | WWTP operation and sludge disposal inventory per m^3 of raw wastewater

Parameter	Value	Units
Energy and chemical associated with the operation of the system		
WWTP total energy input	0.778	kWh/m^3
Aluminum sulphate for phosphorus removal	0.021	kg $Al_2(SO_4)_3{\cdot}14H_2O/m^3$
Transport distance for aluminum sulphate	270	km
Sludge dewatering, transport and composting		
Sludge production	0.19	kg/m^3
Sludge mean dryness	23	%
Sludge production (dry matter, DM)	0.0445	kg DM/m^3
Energy input for sludge dewatering	0.0005	L $diesel/m^3$
Transport distance for sludge land farming	50	km

Sludge produced by the treatment process was removed after a build-up period of approximately ten years (1995–2005). A short time before this removal, the sludge volume of each lagoon was estimated through several in-situ thickness measurements whereas sludge nutrient and heavy metal content were determined from sludge samples analysis. Regarding sludge disposal, the energy consumption for mechanical dewatering was assessed from information provided by the private contractor and the transport mean distance between the WWTP and the agricultural land was estimated as 50 km. In Table 2, the WWTP global energy input and aluminum sulphate transport and consumption for phosphorus removal are presented along with the inventory data related to sludge disposal.

Heavy metals

Due to the importance of heavy metal effects in human toxicity and ecotoxicity impact categories shown in previous LCAs (Hospido *et al.* 2004; Gallego *et al.* 2008), the potential impact of adding heavy metals to agricultural soil through sludge spreading is considered in this study. To avoid disregarding potential impact of wastewater heavy metal content and by doing so skewing the NEB assessment, influent and effluent heavy metal data have been measured for the case study plant. This was done by analysing three 24 hour composite samples of the influent and three grab samples of the effluent during the year 2011. Because of the initial intensive mixing in the first two lagoons and the long retention time of the WWTP, the daily variability of the effluent quality is low and grab samples are considered representative of the daily mean effluent. Samples were taken during winter, summer and fall to capture seasonal variability. Influent and effluent loads have been assessed based on daily flow and measured concentration. Table 3 shows mean and standard deviation (STD) of the measured influent load. As many effluent measured concentrations showed values below the detection limit (DL), influent mean loads and sludge equivalent removal loads are used to assess effluent loads presented in Table 3 in order to obtain mass balancing. Sludge heavy metal loads of Table 3 were estimated in relation to the functional unit by multiplying the heavy metal concentrations of sludge in Table 3 by the assessed sludge production of Table 2. As the values of Table 3 were obtained by using both influent data coming from grab samples and sludge estimated heavy metal loads coming from long-term accumulation in the sludge, this only allows performing an approximated heavy metal mass balance over the WWTP. Considering the uncertainty of Life Cycle Impact Assessment (LCIA) regarding toxicity impact categories (Renou *et al.* 2008), this level of accuracy in the estimation of heavy metal concentrations seems suitable for the present study.

Greenhouse gas emissions

As shown in Figure 1, the direct greenhouse gas (GHG) emissions from the treatment process and the indirect GHG emissions generated in the receiving water bodies due to the rejection of treated or non-treated wastewater have been included in the case study. The indirect GHG emissions can be particularly important when large COD loads are emitted in the receiving waters causing potential methane emissions induced by methanogenic conditions (Foley *et al.* 2010), e.g. for the null option case. Table 4 shows GHG emissions and corresponding global warming potential expressed in CO_2 equivalent per m^3 of raw

Table 3 | Estimated heavy metal loads in relation to the functional unit

	Heavy metal loads					
	Influent (measured)			Sludge (measured)		Effluent (computed)
Parameter	Mean mg/m^3	STD	DL	mg /kg DM	mg/m^3	mg/m^3
Arsenic (As)	3.1	1.3	0.2	6.5	0.3	2.8
Cadmium (Cd)	0.2	0.1	0.2	1	0.04	0.16
Chromium (Cr)	1	1	0.5	46	2	0[a]
Cobalt (Co)	1.0	0.5	0.5	6	0.3	0.7
Copper (Cu)	24	5	1	170	8	16
Mercury (Hg)	0.4	0.1	0.2	2.5	0.1	0.3
Nickel (Ni)	3.6	1.5	1	29	1.3	2.4
Lead (Pb)	2.6	1.5	1	46	2.0	0.6
Selenium (Se)	1.5	1.2	1	0.7	0.03	1.47
Zinc (Zn)	61	14	5	320	14	47

[a]Fixed to zero as sludge load is higher than influent mean load.

wastewater. CH_4 and N_2O emission estimations have been computed based on IPPC Guidelines (Doorn *et al.* 2006) considering biogenic CH_4 emissions due to COD anaerobic degradation and N_2O emissions from nitrogen removal. One must note that biogenic CO_2 emissions have been excluded according to IPCC Guidelines (Doorn *et al.* 2006). Ranges for CH_4 emissions reflect that only 0 to 10% of removed COD is assumed to be anaerobically biodegraded in the lower part of the lagoons and that 0 to 20% of discharged COD is assumed to be ultimately anaerobically biodegraded downstream. The theoretical ratio of 0.25 kg CH_4 per g of COD is further applied to calculate CH_4 emissions (range and IPCC expert recommendation) that are shown in Table 4. Regarding COD emissions, the fraction of COD that is anaerobically degraded into CH_4 was removed from the total COD load to avoid double counting of its impact. Regarding N_2O emissions, the wide range of values for potential production shows the high uncertainty related to N_2O emission. The GWP is assessed using GWP factors for a time scale of 100 years (GWP100) of 22 kg CO_2 eq

Table 4 | Direct and indirect range of GHG emissions per m^3 of raw wastewater

	IPCC CH_4 emission factors			GWP100 (g CO_2 eq/m^3)	
CH_4 emission sources	**Units**	**Range**	**Recommended value**	**Range**	**Recommended value**
Effluent discharge without treatment	kg CH_4 per kg COD discharged	0 to 0.050	0.025	0 to 528	264
Effluent discharge with treatment				0 to 66	33
WWTP operation	kg CH_4 per kg COD removed	0 to 0.025	0.00125	0 to 242	121
	IPCC N_2O emission factors			**GWP100 (g CO_2 eq/m^3)**	
N_2O emission sources	**Units**	**Range**	**Recommended value**	**Range**	**Recommended value**
Effluent discharge without treatment	kg N_2O-N per kg N discharged	0.0005 to 0.25	0.0025	3 to 1550	15
Effluent discharge with treatment				3 to 1490	15
WWTP operation	g N_2O per person per year	2 to 8	3.2	2 to 9	4

per kg of biogenic CH_4 and 298 kg CO_2 eq per kg of N_2O. These GWP100 factors were taken from the LCIA method used for the case study and are discussed in the following section.

Impact assessment

The inventory and impact assessment phases were performed with the SimaPro 7.2 LCA software. Our inventory was completed using the Ecoinvent database Version 2.2 for secondary processes like transport and primary resource production for aluminum sulphate. The Quebec electricity mix where 95% of the electricity is generated from hydropower was also added to the database to assess the electricity input potential impact. The LCIA method ReCiPe hierarchist version 1.06 was retained for impact assessment. ReCiPe comes from the harmonization of the two widely recognized LCA methodologies CML and Eco-indicator 99 (Goedkoop *et al.* 2009). Still, we have modified ReCiPe to take into account the effect of oxygen depletion due to organic pollution by including it into the freshwater eutrophication impact category following the methodology of Heijungs *et al.* (1992). We chose to aggregate the impact up to the midpoint to track the trade-off between impacts generated by the WWTP's life cycle and the environmental gain related to wastewater pollutant removal.

RESULTS AND DISCUSSION

Results of the LCIA according to the NEB methodology are summarized in Table 5 which also presents the results that a standard LCA would provide. Only a limited number of impact categories among the 18 midpoint categories of ReCiPe are shown as examples. Globally the NEB shows a trade-off between impact generated by the WWTP's life cycle expressed as negative NEB and avoided impacts thanks to rejected wastewater quality improvement expressed as positive NEB values. More precisely, the positive NEB value for eutrophication is due to organic matter and nutrient removal. Its counterpart corresponds to the negative NEB values for metal and fossil depletion mainly caused by electricity consumption for aeration and production and transport of aluminum sulphate for phosphorus removal. Regarding toxicity due to heavy metals, the NEB approach points out the transfer of impacts from marine and freshwater aquatic ecotoxicity to human toxicity and terrestrial ecotoxicity. Heavy metals are actually mostly removed from wastewater but are rejected into the environment through agricultural land spreading of the treatment sludge. Regarding climate change, the large uncertainties surrounding CH_4 and N_2O emissions (see Table 4) lead to either positive or negative NEB. Positive NEB values in this category are partly caused by Quebec's electricity mix containing low fossil fuel primary energy sources.

Comparing the NEB and standard LCA results of Table 5, it appears that both methods show the same results for impact categories only containing the potential impact caused by the WWTP's life cycle. For all other impact categories, the pollutant removal potential benefit expressed by the difference between rejecting wastewater without and with treatment allows showing positive potential environmental impacts of treating wastewater that are not overcome by the potential impacts generated by the WWTP's life cycle. On the down side, the NEB approach is more data intensive than a standard LCA conducted on

Table 5 | LCIA results expressed as NEB and standard LCA

Impact categories (ReCiPe V1.06)		Units	IP_{NO}	IP_{TW}	IP_{SLC}	NEB (eq. 1)	Standard LCA
Climate change (GWP100)	Range min value	g CO_2 eq	3.0	3.0	2.1	−2.1	5.1
	Recommended value		279	48	144	87	192
	Range max value		2080	1560	270	250	1830
Freshwater eutrophication		g P eq	12.2	3.4	0.0	8.8	3.4
Marine eutrophication		g N eq	132	4.5	0.0	128	4.6
Freshwater ecotoxicity		g 1.4-DB eq	1.1	0.8	0.1	0.1	0.9
Marine ecotoxicity		g 1.4-DB eq	1.0	0.7	0.1	0.1	0.8
Human toxicity		g 1.4-DB eq	75	67	18	−10	85
Terrestrial ecotoxicity		g 1.4-DB eq	3E-04	2E-04	0.10	−0.10	0.10
Metal depletion		g Fe eq	–	–	0.2	−0.2	0.2
Fossil depletion		g oil eq	–	–	4.7	−4.7	4.7

a WWTP because one must gather data related to influent quality whereas this is not necessary for a standard LCA. However, the NEB approach takes advantage of this more extensive inventory to provide an overview of the environmental performance of the WWTP that cannot be obtained from a standard LCA. Data required by the NEB point to the actual data gap in the monitoring of the WWTP. Currently, most of the data on WWTP are gathered to ensure legal compliance of the effluent and sludge disposal process. A better characterization of the influent could improve the efficiency assessment of the WWTP and attract attention to source control of pollutants.

CONCLUSION

The NEB approach presented in this paper shows potential in helping to frame and interpret LCA in the field of wastewater treatment by taking into account more explicitly the function of the WWTP. The approach requires assessing the potential impacts associated with a direct discharge of wastewater into the environment, the potential impacts of the discharge of treated water and the potential impacts of the WWTP's life cycle by means of the LCA methodology. Compared with a standard LCA, the NEB approach allows a clearer definition of the main function of the WWTP by representing the trade-off between the environmental benefit associated with the improved water quality before discharge to the receiving water body and impacts generated by the WWTP's life cycle. This avoids seeing the WWTP as a major pollutant source of the integrated water system, as it appears in previous LCA studies, but rather recalls its pollution control function. However, it also points out what are the environmental impacts of the WWTP's cycle itself. The NEB is therefore a promising approach to more efficiently assess the global environmental impacts of a WWTP.

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