Identifying energy and carbon footprint optimization potentials of a sludge treatment line with Life Cycle Assessment
C. Remy, B. Lesjean and J. Waschnewski

ABSTRACT
This study exemplifies the use of Life Cycle Assessment (LCA) as a tool to quantify the environmental impacts of processes for wastewater treatment. In a case study, the sludge treatment line of a large wastewater treatment plant (WWTP) is analysed in terms of cumulative energy demand and the emission of greenhouse gases (carbon footprint). Sludge treatment consists of anaerobic digestion, dewatering, drying, and disposal of stabilized sludge in mono- or co-incineration in power plants or cement kilns. All relevant forms of energy demand (electricity, heat, chemicals, fossil fuels, transport) and greenhouse gas emissions (fossil CO₂, CH₄, N₂O) are accounted in the assessment, including the treatment of return liquor from dewatering in the WWTP. Results show that the existing process is positive in energy balance (–162 MJ/PECOD *a) and carbon footprint (–11.6 kg CO₂-eq/PECOD *a) by supplying secondary products such as electricity from biogas production or mono-incineration and substituting fossil fuels in co-incineration. However, disposal routes for stabilized sludge differ considerably in their energy and greenhouse gas profiles. In total, LCA proves to be a suitable tool to support future investment decisions with information of environmental relevance on the impact of wastewater treatment, but also urban water systems in general.

Key words | energy optimization, global warming, Life Cycle Assessment, sludge treatment, sustainability

INTRODUCTION
Wastewater treatment plants (WWTPs) are well known to be a major energy user of urban water management infrastructure. In fact, WWTPs represent the greatest single consumer of electrical energy within the public sector in Germany, exceeding the energy demand of street lighting, schools, hospitals and other municipal facilities (Haberkern et al. 2008). Driven by the political will to cut costs and reduce the overall energy demand of municipalities, increased efforts have been made in recent years to improve the energy efficiency of the wastewater treatment process. This includes optimization measures to increase energy efficiency of processes, modifications in process design, and increased recovery of energy via anaerobic sludge digestion. The final vision of the sewage treatment plant of the near future would be an energy self-sufficient or even energy-positive process which complies with rising standards in effluent quality and environmental impact (Svardal & Kroiss 2011).

However, most optimization measures only target the decrease in direct electrical energy demand, i.e. the reduction of the amount of electricity consumed. Indirect effects of the optimization can thus be overlooked and may negatively influence the overall energy balance. For example, the addition of energy-rich fatty co-substrates to the sludge digestion process increases the production of valuable biogas, but may also have a negative impact on sludge dewaterability (DWA 2008). This may lead to an increased demand of polymer for the dewatering process and higher water content in dewatered sludge, affecting the energy recovery potential during disposal in incineration processes. For an overall evaluation of optimization measures, a comprehensive assessment of all direct and indirect effects should be targeted to provide reliable information and decision support for choosing the most preferable option.
An adequate tool for this task is the methodology of Life Cycle Assessment (LCA) as described in ISO 14040 and 14044 (ISO 14040 2006; ISO 14044 2006). Originally developed for the assessment of products, LCA is increasingly used for the evaluation of services (Schnoor 2009), and has already been applied to processes of potable water production (e.g. Friedrich 2002; Stokes & Horvath 2006; Lassaux et al. 2007; Vince et al. 2008) and wastewater treatment (e.g. Tillman et al. 1998; Hospido et al. 2004, 2008; Hoibye et al. 2008; Remy & Jekel 2008, 2012; Wenzel et al. 2008) or even the entire water cycle of a city (Lundie et al. 2004). Most studies found that electrical and thermal energy demand plays a major role in the environmental impact of WWTP, besides the direct emissions of nutrients into receiving waters which depend on the effluent quality. Additionally, sludge disposal can substantially affect the environmental profile if heavy metals and other pollutants bound in the sludge are accounted for in the LCA (Hospido et al. 2010). For different wastewater treatment schemes, definitions of system boundaries and system scale have been found to influence the comparison of different wastewater schemes heavily (Lundin et al. 2000).

Through a step-by-step procedure, LCA can systematically evaluate different scenarios for process operation or technology upgrade by stating goal and scope of the study, setting up an inventory with process data, and evaluate this system with a set of environmental indicators. Using the life cycle perspective, LCA takes into account all related upstream and downstream processes of WWTPs, such as production of electricity and chemicals or the disposal of sludges, and highlights the share of direct (=on-site) and indirect emissions of the system operation. Thus, it is well suited to show the trade-offs of environmental protection technologies such as WWTP, shifting the environmental burden from the direct emissions (which are reduced) at the cost of indirect emissions for energy and materials required during their operation and for the disposal of separated pollutants. On the other hand, LCA gives no information on the actual environmental damage of a certain process coming into effect, because its environmental indicators are based on global or at best regional characterization factors calculating only the potential for causing an environmental impact. Hence, the dynamic operation of WWTP due to variations of the influent wastewater throughout the day (morning peak, rain events) and the seasons and the related impacts on the local aquatic environment (e.g. short-term oxygen deficiency) may not be adequately reflected by LCA. However, LCA is well suited to analyse existing processes based on average performance data and to compare different technology options in terms of resource demand and related emissions as long-term environmental loads.

The present work will show the applicability of this tool for evaluating different options for process optimization in a large WWTP in Berlin. The focus of this study is on the process for sludge treatment and disposal, analysing the existing treatment line and disposal options in their impacts on cumulative energy demand (CED) and the emission of greenhouse gases (GHG), also known as carbon footprint.

METHODS

The present study closely follows the guidelines of ISO 14040/44. The goal of the study is to quantify the environmental impacts of the existing sludge treatment line (baseline scenario) and to reveal potentials for optimization. The primary function of the investigated system is the disposal of sewage sludge from sedimentation (primary sludge) and from the activated sludge process (excess sludge). Secondary functions such as the supply of electricity and heat (from biogas combustion in combined heat and power (CHP) plant), fertilizer (from production of magnesium ammonium phosphate (MAP)) and the substitution of fossil fuels in power plants or cement kilns are accounted for by crediting the respective products as avoided burden. The functional unit (FU) is defined in relation to the organic load in the influent of the WWTP, which represents an important parameter for the energy recovery potential within the process. The WWTP treats a daily mean flow of 191,000 m³ wastewater with an organic load of 944 mg/L of chemical oxygen demand (COD), equaling 1.5 Mio population equivalents (PE) assuming a mean COD load of 120 g/(PE · d) (ATV 2000). Consequently, the FU is defined as the amount of sludge generated per PE COD and year. This relation enables a fair comparison with other WWTPs on the basis of comparable organic loads (and not volume, for example). Reference flows define the quantity and quality of input primary and excess sludge according to annual mean data (Table 1).

Process description of sludge treatment line

Within the system boundaries of the study, all relevant processes of sludge treatment and disposal are considered (Figure 1). Excess sludge from the activated sludge tank is thickened by centrifuges with polymer dosage. Primary sludge from settling of raw wastewater is mixed with
excess sludge and fed into the digestors. Mixed sludge is screened for larger particles and preheated before entering the digestors which are operated in mesophilic mode (38°C). Additionally, digestors are fed with external fats to utilize excess digestion capacity. Sludge retention time within the digestors amounts to 18–20 days (Table 2). The generated biogas is purified (H₂S removal by biological process) and stored before combustion in CHP units (electrical/thermal efficiency: 36/44%, CH₄ slip 0.75%). Digested sludge is fed to a separate reactor, where MAP is precipitated by addition of MgCl₂ and pH increase by stripping of CO₂ with air (residual dissolved CH₄ in digested sludge (18 mg/L) is stripped to atmosphere). Precipitated MAP is separated by gravity and is sold as high-quality fertilizer. Digested sludge (∼4% total solids (TS)) is dewatered by centrifuges with polymer addition (final TS: 27%) prior to disposal. Sludge liquors from thickening and dewatering are recycled back to the WWTP inlet. These liquors contain a considerable load of N and COD, so this LCA accounts for the associated effects with a simplified model estimating energy demand (0.42 kWh/kg CODₐₘᵢₜ or 2.8 kWh/kg Nₐₘᵢₜ) and GHG emissions of denitrification (6 g N₂O-N/kg Nₐₘᵢₜ (Wicht 1996)) in the activated sludge process (Table 3).

Final disposal of the dewatered sludge targets energy recovery via incineration. Currently, a mix of four different routes of sludge disposal is used at the WWTP: (1) mono-incineration using additional fuel oil (35 km transport, 16% of stabilized sludge); (2) incineration in coal power plant (190 km, 50%); (3) drying on-site and incineration in coal power plant (386 km, 6%); and (4) drying on-site and incineration in cement kiln (90 km, 28%). On-site drying is done in drum dryers (final TS: 93%) fuelled by natural gas or biogas from digestion (276 m³ gas/tTS; ton total solids). The distribution of the total sludge volume between the four disposal routes depends on available capacities and economic

Table 1 | Quantity and quality of primary and excess sludge in Berlin WWTP and co-substrates

<table>
<thead>
<tr>
<th></th>
<th>Primary sludge (3% TS)</th>
<th>Excess sludge (1.2% TS)</th>
<th>External fats (5% TS)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Volume [L/(PE * a)]</td>
<td>246</td>
<td>1,178</td>
<td>24</td>
</tr>
<tr>
<td>TS [kg/(PE * a)]</td>
<td>12.2</td>
<td>14.4</td>
<td>1.2</td>
</tr>
<tr>
<td>Organic matter [kg/(PE * a)]</td>
<td>10.1</td>
<td>11.4</td>
<td>1.1</td>
</tr>
<tr>
<td>COD [kg/(PE * a)]</td>
<td>14.2</td>
<td>16</td>
<td>1.6</td>
</tr>
<tr>
<td>N [kg/(PE * a)]</td>
<td>0.43</td>
<td>1</td>
<td>0.02</td>
</tr>
<tr>
<td>P [kg/(PE * a)]</td>
<td>0.15</td>
<td>0.53</td>
<td>0.02</td>
</tr>
</tbody>
</table>

*aQuality estimated.*
conditions. At this stage, the study is restricted to the analysis of the status quo of sludge disposal at this plant: therefore, alternative routes for sludge disposal such as solar drying or disposal in agriculture are not investigated here. In fact, the heavy metal content (e.g. Cu) of the digested sludge prohibits the disposal in agriculture according to German regulations for sewage sludge disposal (AbfKlär 1992).

Environmental impacts of the background systems associated with the sludge treatment line are accounted for by using the LCI database Ecoinvent v2.2 (Ecoinvent 2007). Background processes include the supply of electricity (assuming the energy mix of Germany 2010), transport processes (truck with 16–32t, Euro4), the supply of natural gas (in Germany, high pressure), and the production and transport of chemicals such as polymers (assumed as acetonitrile) and MgCl₂ (Table 3). The disposal of sludge in incineration is modelled based on the heating value of the sludge and representative emission data from incineration processes (mono-incineration of sewage sludge, power plant, or cement kiln).

**Process data for Life Cycle Inventory**

Process data for the inventory is compiled from full-scale operational data of the large WWTP in Berlin for the year

<table>
<thead>
<tr>
<th>Table 2</th>
<th>Operating parameters of mesophilic digestion process</th>
</tr>
</thead>
<tbody>
<tr>
<td>Retention time</td>
<td>Mixed sludge (6% TS) 18–20</td>
</tr>
<tr>
<td>Gas yield</td>
<td>[L/kg oDM] 416</td>
</tr>
<tr>
<td>Methane content</td>
<td>[%] 61</td>
</tr>
<tr>
<td>Degradation of organic matter</td>
<td>[%] 47</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Table 3</th>
<th>Emission factors for GHG emissions in the inventory of this LCA</th>
</tr>
</thead>
<tbody>
<tr>
<td>Processes</td>
<td>Direct emissions</td>
</tr>
<tr>
<td>MAP reactor</td>
<td>CH₄ [g/kg TS] 0.55</td>
</tr>
<tr>
<td>CH₄ [g/kg CH₄] 7.5</td>
<td>Methane slip (Ronchetti et al. 2002)</td>
</tr>
<tr>
<td>CHP plant or natural gas burner</td>
<td>N₂O [g/kg CH₄] 0.08</td>
</tr>
<tr>
<td>Liquor treatment in mainstream WWTP</td>
<td>N₂O [g N₂O-N/kg Nₐₐₜ] 6</td>
</tr>
<tr>
<td>Mono-incineration of sludge</td>
<td>N₂O [g N₂O/t TS] 990</td>
</tr>
<tr>
<td>Co-incineration of sludge</td>
<td>N₂O [g N₂O/t TS] 100</td>
</tr>
<tr>
<td>Background processes</td>
<td>Indirect emissions</td>
</tr>
<tr>
<td>Electricity generation</td>
<td>CO₂ [g/kWh] 547</td>
</tr>
<tr>
<td>N₂O [g/kWh] 0.02</td>
<td></td>
</tr>
<tr>
<td>CH₄ [g/kWh] 1.38</td>
<td></td>
</tr>
<tr>
<td>Truck transport</td>
<td>CO₂ [g/tkm] 129</td>
</tr>
<tr>
<td>N₂O [g/tkm] 0.005</td>
<td></td>
</tr>
<tr>
<td>CH₄ [g/tkm] 0.13</td>
<td></td>
</tr>
<tr>
<td>Polymer production</td>
<td>CO₂ [g/kg] 1957</td>
</tr>
<tr>
<td>N₂O [g/kg] 0.012</td>
<td></td>
</tr>
<tr>
<td>CH₄ [g/kg] 9.3</td>
<td></td>
</tr>
<tr>
<td>Production of MgCl₂ (30%)</td>
<td>CO₂ [g/kg] 73</td>
</tr>
<tr>
<td>N₂O [g/kg] 0.0002</td>
<td></td>
</tr>
<tr>
<td>CH₄ [g/kg] 0.18</td>
<td></td>
</tr>
<tr>
<td>Natural gas production</td>
<td>CO₂ [g/MJ] 129</td>
</tr>
<tr>
<td>N₂O [g/MJ] 0.005</td>
<td></td>
</tr>
<tr>
<td>CH₄ [g/MJ] 0.13</td>
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2009. Remaining data gaps are filled by qualified assumptions in cooperation with plant operators or from literature data. The entire sludge treatment line is described in a static substance flow model using the LCA software UMBERTO<sup>®</sup> 5.5 (IFU and IFEU 2009), where all relevant input and output streams for each sub-process are characterized in terms of volume, TS, COD, total nitrogen, and total phosphorus content. Due to the setup of a specified process, it is assumed that the use of other commercial LCA software (GaBi, SimaPro) would lead to comparable results if the same background database (Ecoinvent software) is used.

Within the substance flow model, all necessary process-specific inputs (e.g. electricity, heat, chemicals) are compiled from operational data of the WWTP, while direct process emissions of GHG (fossil CO<sub>2</sub>, CH<sub>4</sub>, N<sub>2</sub>O) are estimated according to literature (Table 3). CO<sub>2</sub> from combustion of biogas is short-cycle carbon and not accounted for in GHG emissions (IPCC 2007). N<sub>2</sub>O emissions during sludge incineration can be significant in the case of low incineration freeboard temperatures (<900 °C) such as in many mono-incineration facilities (ATV 1996; Sänger et al. 2001). Hence, a default emission factor of 990 g N<sub>2</sub>O/t TS is used for mono-incineration (IPCC 2006) because no on-site measurement data are available. For co-incineration in power plants and cement kilns, higher incineration temperatures (>950 °C) are assumed to lead to lower N<sub>2</sub>O emissions (Svoboda et al. 2006), using an emission factor of 100 g N<sub>2</sub>O/t TS as best estimate (Table 3). Detailed data of the Life Cycle Inventory is described elsewhere (KWB 2012).

**Credits for secondary products**

Besides the primary function of sludge stabilization and disposal, the sludge treatment line delivers a bundle of secondary products. On-site products include electricity (28,590 MWh/a) and heat from CHP plants, the latter being used for digester heating on-site. Excess heat is not accounted for as credit in this LCA. Additionally, MAP fertilizer (770 t/a) can substitute industrial nitrogen and phosphorus fertilizer. Disposal of dewatered sludge in mono-incineration further generates electricity (620 kWh/t TS), while disposal of dewatered/dried sludge in power plants or cement kilns substitutes other fossil fuels such as lignite or hard coal. Substitution potentials are calculated based on the lower heating value of dewatered/dried sludge (8.2/14.8 MJ/kg TS) in relation to lignite (8.7 MJ/kg TS) or hard coal (27.6 MJ/kg TS). All secondary products are subtracted from the environmental impact of the process (‘avoided burden’ approach), using Ecoinvent datasets for grid electricity production, nitrogen and phosphorus fertilizer, and the supply and incineration of lignite or hard coal (Ecoinvent 2007).

**Impact assessment**

All scenarios for optimization are evaluated in terms of CED (VDI 1997) which is a measure for the total demand of fossil and nuclear non-renewable fuels in megajoules [MJ]. This energy indicator can also serve as a screening indicator for the identification of those processes with the highest total environmental impact, because energy demand and the associated impact via electricity production are often responsible for a major share of the total impacts of industrial processes (Huijbregts et al. 2006). As a second indicator, global warming potential (GWP) is calculated to address the issue of climate change which has been identified as one of the major global threats to humans and ecosystems. This indicator evaluates the emission of GHG such as CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O by relating them to CO<sub>2</sub> as a reference substance [kg CO2-eq] (IPCC 2007). Finally, indicator results can be normalized relating them to the total environmental impact in the respective category (e.g. the total emission of GHG per person and year in Germany). Normalization gives additional information about the contribution of the process under investigation towards the total environmental impacts of society. In this study, normalized scores for CED and GWP relate to a total CED of 156,500 MJ/(PE * a) and total GWP of 11,840 kg CO<sub>2</sub>-eq/(PE * a) in Germany in 2010.

**RESULTS**

Results of the impact assessment are presented in detail for the baseline scenario and for the different disposal routes in terms of CED and GWP. Indicator values are related to the FU of organic load in influent of WWTP (PE<sub>COD</sub> * a)<sup>−1</sup> and describe the environmental impacts and credits which are directly associated with the specific sub-processes.

**Cumulative energy demand**

The total CED of the sludge treatment line amounts to 219 MJ/(PE<sub>COD</sub> * a). Thereof, 46% is due to the electricity demand, 22% due to fuel supply for drying and mono-incineration, and 20% for heating the digestors (Figure 2). The
transport of stabilized sludge and the addition of chemicals account for 5 and 6%, respectively. Sub-processes with highest CED are digestion (62 MJ/(PECOD ° a)) and drying (46 MJ/(PECOD ° a)), both due to the demand of thermal energy in form of heat. However, heat demand of the digestors is met to 100% by off-gas heat from the CHP plant. The treatment of the recycled sludge liquor in the WWTP process requires 20 MJ/(PECOD ° a). In the existing sludge treatment line, credits for secondary products amount to –380 MJ/(PECOD ° a) (Figure 3). The production of electricity from biogas combustion in CHP plant is responsible for the major part of this energy, accounting for –187 MJ/(PECOD ° a) in electricity and –44 MJ/(PECOD ° a) in waste heat which is used on-site for digester heating. Another –68 MJ/(PECOD ° a) of waste heat cannot be utilized on-site in the current process, and this ‘lost’ heat energy is consequently not accounted for in the overall balance of the sludge treatment line. The substitution of fertilizer by MAP and fossil fuels during disposal account for –4 and –133 MJ/(PECOD ° a). Dewatered or dried sludge is a well-suited fuel substitute in power plants or cement kilns, enabling the recovery of remaining organic-bound energy potential which could not be converted into biogas in the digestion process. Overall, the sludge treatment line is positive in its energy balance, with a total ‘impact’ of –161 MJ/(PECOD ° a).

**Carbon footprint**

The greenhouse gas emission profile is correlating with the CED. Most processes requiring energy involve the emission of CO₂ from fossil resources. Thus, the distribution of GWP...
between the different sub-processes is similar to CED (Figure 4). Major contributors to the total GWP of 16.6 kg CO₂-eq/(PECOD * a) are the supply of electricity (38%) and heat (16%), whereas the production of chemicals (5%) and sludge transport to disposal (3%) have only minor contributions. However, direct process emissions also contribute to GWP, namely for the CHP plant (CH₄ slip) and MAP process (dissolved CH₄ from anaerobic digestion, stripped during aeration), liquor treatment in WWTP (N₂O from incomplete denitrification), and mono-incineration of dewatered sludge (N₂O from incineration). Fossil CO₂ from combustion of fossil fuels in drying and mono-incineration also contributes to the carbon footprint. Secondary products account for savings of −28.2 kg CO₂-eq/(PECOD * a), mainly from substitution of electricity and fossil fuels (Figure 5).

Overall, the existing sludge treatment line has a carbon footprint of −11.6 kg CO₂-eq/(PECOD * a), thus generating a net environmental benefit for the system.

**Normalization**

Normalized impacts of the sludge treatment line amount to a contribution of −0.1% for CED and −0.1% for GWP compared with the total energy demand and GHG emissions in Germany. This underlines that in general the process of wastewater treatment has only a marginal share of the total energy demand and GHG emissions in Germany. These figures are useful to recall the primary function of WWTPs and its associated environmental impact, which is the protection of surface waters from excessive pollution. Although the energy demand and GHG emissions of a WWTP are relatively small compared with the total impacts.

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**Figure 4** | GWP of existing sludge treatment line (impacts only).

**Figure 5** | GWP of existing sludge treatment line (impacts and credits).
of society, an optimization of the WWTP process in terms of resource demand and emissions is still an imperative goal for the future.

Analysis of disposal routes

For optimization of the environmental profile of sludge treatment, the different disposal routes for stabilized and dewatered sludge are directly compared in their energy demand and carbon footprint. For a better comparability between the various routes, the FU is changed to the disposal of 1 ton total solids (tTS) after dewatering. Thus, the results for the optimization show only the environmental impacts and credits of optional drying, transport, and incineration, because all previous processes (digestion + dewatering) are comparable between all routes.

In total, the CED of the baseline scenario (existing mix of disposal routes) amounts to 4.9 GJ/tTS while generating secondary products equivalent to a benefit of –10.3 GJ/tTS. Overall, the current disposal situation (starting with dewatered sludge) yields a reduced energy demand of –5.4 GJ/tTS. A comparison between the four different disposal routes reveals the energy profiles of each incineration process (Figure 6). Mono-incineration requiring extra fuel oil (due to high water content of dewatered sludge, preventing a self-maintained incineration process) yield the lowest energy surplus of –0.5 GJ/tTS. Co-incineration of dewatered sludge in power plants requires the highest energy for transport because of the high water content of the sludge. Drying of dewatered sludge significantly increases energy demand (dryer needs substantial amounts of natural gas and electricity), but also increases heating value of the sludge, enabling the substitution of more fossil fuels. The net energy balance reveals the following ranking in terms of energy surplus: mono-incineration < dewatered sludge in power plant < dried sludge in cement kiln < dried sludge in power plant. The latter option is the most beneficial way of disposal (~7.9 GJ/tTS): no drying on-site, but transport to the nearest power plant and substitution of primary fossil fuels such as lignite or hard coal.

Figure 6 | CED of different disposal options for stabilized and dewatered sludge.

For GWP, the ranking of the disposal routes is different: compared with the baseline mix (net benefit of –593 kg CO₂-eq/tTS), mono-incineration has a significantly higher GWP, with a net balance of +292 kg CO₂-eq/tTS (Figure 7). This effect is due to the emission of fossil CO₂ (from extra fuel oil) and N₂O (GWP₉₂₈ = 298 kg CO₂-eq), the latter being a problem that has been recognized for mono-incinerators for sewage sludge and that is mainly caused by low incineration temperatures (<900 °C). The other disposal options all yield a reduction in GWP, with a net credit of –740 kg CO₂-eq/tTS for incineration of dewatered sludge in power plant and –803/–778 kg CO₂-eq/tTS for incineration of dried sludge in power plant and cement kiln.

In summary, the comparison of the different disposal options in energy demand and carbon footprint reveals that the most beneficial way of sludge disposal in the present case study is the co-incineration of dewatered sludge in power plants. On-site drying has an inferior energy balance and comparable GHG emissions, even though a long distance to the incineration facility will tend to favour this option due to lower transport volumes. In this study,
a transport distance of more than 735 km will lead to a superior energy balance of drying. In general, energy and GHG balance for drying could be improved if excess heat from CHP plants can be utilized, e.g. for the drying process or sludge dewatering. Mono-incineration of sewage sludge yields the smallest credits in CED and GWP (=lowest recovery potential), at the same time requiring additional fuel for incineration and causing substantial emissions of N₂O. An energy optimization of the mono-incineration facility is recommended to increase the output of electricity and reduce the need for extra fuel oil, e.g. by pre-treatment such as drying or heating the sludge to increase its heating value. The high carbon footprint of mono-incineration caused by direct emissions of N₂O (Figure 7) could be reduced by increasing incineration freeboard temperatures (Sänger et al. 2003), but the excessive generation of other regulated gases of environmental concern (CO, NOₓ) could be limiting this mitigation measure if no additional steps for flue gas cleaning are introduced. Further investigations into suitable measures for N₂O mitigation are required to overcome this intrinsic drawback of mono-incineration.

**DISCUSSION AND CONCLUSIONS**

This study exemplifies the use of LCA as a tool to quantify the overall environmental impacts of processes in wastewater treatment and identify optimization potentials. The present case study analyses the sludge treatment line of a large WWTP via LCA, focussing on energy demand and the emission of GHG. Results indicate that the operation of the digestors, the separation of sludge water in centrifuges, and sludge drying in drum dryers are responsible for the major part of energy demand and GHG emissions. However, secondary products such as electricity from biogas combustion or substitution of fossil fuels during incineration of stabilized sludge offset the total impacts of the sludge treatment line, resulting in an overall credit of 161 MJ or 11.6 kg CO₂-eq per person and year.

A comparison of the different disposal routes for incineration of stabilized sludge reveals that mono-incineration of sewage sludge is not favourable in terms of energy demand and GHG emissions due to high demand of external fuels and significant emissions of N₂O. This is in accordance with previous studies ranking mono-incineration of dewatered sewage sludge inferior in energy balance and carbon footprint compared to co-incineration (IFEU 2002). Another study compared mono-incineration of dewatered sludge and co-incineration of dried sludge in cement kiln, indicating that mono-incineration can be energetically favourable due to the high gas demand in sludge drying (Houillon & Jolliet 2005). However, co-incineration in cement kilns had a considerably lower carbon footprint in this study due to substitution of hard coal, even if N₂O emissions from the incineration process are neglected.

The favourite option in the present case study is the incineration of dewatered sludge in lignite power plants.
On-site drying of sludge does not improve the energy balance, but can slightly decrease the emission of GHG due to the substitution of hard coal with high carbon footprint in power plants or cement kilns. Alternative low-energy drying processes such as solar drying (Bennamoun 2012) are only suitable for smaller WWTPs (up to 50,000 PE) due to the large required surface area.

For the sludge treatment line investigated in this study, several recommendations can be given to optimize the energy balance and GHG emission profile.

- Increase the dosing of co-substrates to make optimum use of digester volume, provided that negative impacts on dewatering of sludge can be excluded.
- Investigate the possible implementation of a side-stream low-energy treatment process (e.g. based on Anammox process (Joss et al. 2009)) for sludge liquor returning to the WWTP.
- Improve the energy balance of the existing mono-incineration facility (e.g. by pre-treatment such as drying or heating of the sludge with excess heat from CHP plants, thus increasing the heating value and lowering the amount of extra fuel needed for incineration).
- Validate the N₂O emission factor from mono-incineration by off-gas measurements. If the high carbon footprint of mono-incineration is confirmed, suitable counter-measures should be investigated to lower the N₂O emissions of mono-incineration in general.

Additionally, the environmental profile of sludge disposal should now be extended to include other categories of environmental concern, e.g. impacts on human or ecotoxicity or the recovery of limited resources such as phosphorus. For example, the transfer of pollutants into the atmosphere during incineration depends on the flue gas cleaning of each incineration facility, presumably favouring mono-incineration with dedicated separation of toxic heavy metals such as mercury (IFEU 2002). Nutrient recovery from sewage sludge (especially phosphorus) could be another valuable feature of mono-incineration, enabling the recovery of phosphorus from incinerator ashes (Cornel & Schaum 2009).

Overall, the method of LCA has proven to be a valuable tool for estimating the environmental impacts of existing processes of wastewater treatment and identifying potentials for optimization. Thus, LCA can support future investment decisions for site development with information of environmental relevance on the impact of WWTPs, but also urban water systems in general.

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