Environmental impact analysis of chemicals and energy consumption in wastewater treatment plants: case study of Oslo, Norway
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ABSTRACT
Wastewater treatment plants, while performing the important function of treating wastewater to meet the prescribed discharge standards, consume energy and a variety of chemicals. This paper analyses the consumption of energy and chemicals by wastewater treatment plants in Oslo over eight years, and their potential environmental impacts. Global warming and acidification were the dominant impacts for chemicals and energy, respectively. Avoided impacts due to usable by-products – sludge, ammonium nitrate and biogas – play a key role in shrinking the environmental footprint of the wastewater plants. The scope for decreasing this footprint by streamlining energy and chemicals consumption is limited, however, considering that over 70% of the impact is accounted for by the eutrophication potential (thanks to the nitrogen and phosphorus which is discharged to the sink) of the treated effluent wastewater.

Key words | acidification, chemicals, energy, environmental impacts, eutrophication, global warming, life cycle environmental analysis, wastewater treatment plants

INTRODUCTION
Wastewater treatment plants (WWTPs), while performing the important function of treating wastewater to meet the prescribed discharge standards, entail the consumption of a variety of chemicals and energy. It is likely that this consumption accounts for a major part of the life cycle environmental footprint of WWTPs. It is also likely that the WWTPs represent an important part of the overall urban water cycle system, with respect to aggregated environmental impacts from the system as a whole. Water and wastewater utilities show increased attention to the consumption of chemicals and energy, and their influence on environmental efficiency overall. Very few studies, however, take a systems life cycle perspective to this issue. Our research goal is to develop a comprehensive systems analysis metabolism model of the whole urban water system of cities, including inflow chemicals, materials and energy as well as outflow emissions and by-products. We believe that such a model will be a useful tool in developing better design, operation, maintenance, rehabilitation and management procedures for the future, in response to the growing demand for sustainability in management of infrastructure systems. This paper examines one part of the urban water system, the WWTPs and their chemicals and energy consumption, using the city of Oslo as a case.

LITERATURE
The chemicals and energy aspects of wastewater treatment have been examined by many researchers and several published works bear testimony to this. Clauson-Kass et al. (2001) reported the results of an LCA carried out at the Avedore WWTP in Denmark in 1998. Electricity sourced from the grid contributed most to global warming (290 grams of CO₂-eq per m³), while the energy generated by combusting the biogas was the biggest contributor to acidification (1.47 grams of SO₂-eq per m³). Funamizu et al. (2001) discussed the reuse of...
heat energy in treated wastewater by using heat pumps, in Japan. Heated wastewater can be very effectively used to warm flooring and melt ice in doorways, or even to heat the incoming fresh cold water. An emphasis has been placed on the need for harnessing the renewable energy potential of wastewater pollutants by resorting to anaerobic processes to generate biogas, cfr. Keller et al. (2003). They also pointed out that the economics of energy recovery from sludge are governed by several constantly-varying factors. Nowak et al. (2005) recommended a benchmark energy demand (excluding aeration) of nutrient removal WWTPs of 7–12 kWh (electrical) per person equivalent per year.

Lassaux et al. (2007), in a Belgian case study, established that in WWTPs the wastewater discharge into the final sink dominates the environmental impact score calculated for wastewater treatment. Kawashima (2002), while averring that infrastructure LCA is extremely important for policy planning, pointed out that over 85% of the life cycle carbon dioxide emissions in activated sludge wastewater treatment in Japan occur during the operation phase. Strategies to estimate, analyse and reduce greenhouse gas emissions from operations offer consumers solid evidence that their utilities are committed to positive stewardship of the environment, according to Strutt et al. (2008). Eckard (2008) pointed out that in the USA, in the year 2006 alone, over 50 billion kilograms of CO2-equivalent greenhouse gases were emitted from wastewater treatment, sludge handling and methane degassing. An LCA of selected sanitation concepts focusing on urine-separation, faeces-composting and grey wastewater treatment was carried out in Remy et al. (2007). Treatment of the wastes separated from the wastewater is as important as treating the wastewater to meet the standards of final discharge to the sink. Different ways of handling sludge solids were compared using LCA in Gouda et al. (2005). Energy consumption and GHG emissions of different types of sewage sludge treatment systems in Japan were analysed in Soda et al. (2010), wherein the effectiveness of the use of digestion gas (biogas) for the energy needs in sewage sludge treatment processes, in reducing GHG emissions, was emphasised.

Eastern Research Group Inc. et al. (2007) discussed the development of CHP facilities at WWTPs in the USA. The report states that, if all the 544 WWTPs in the USA which operate anaerobic digesters and have influent flow rates greater than 18,500 cubic metres per day were to install CHP units, 340 MW of clean electricity would be generated, offsetting over 2 million tonnes of carbon dioxide emissions annually. The preceding instances form a small subset of published research outputs pertaining to energy consumption, energy efficiency, energy recovery, energy-related environmental impacts and energy expenses in WWTPs. Zhang et al. (2009) revealed, by means of a comprehensive LCA analysis of a wastewater treatment and reuse project in China, that the life cycle environmental impact benefit is greater than the life cycle energy consumption for tertiary treatment, making treatment and reuse feasible from an energy-saving point of view.

In this paper, the consumption and environmental impacts of chemicals and energy in the two aforementioned WWTPs in Oslo are studied for the period 2000–2007. Indicators are derived, and the impacts due to chemicals and energy consumption are compared with each other and also with the impacts attributable to the final effluent discharge.

### METHODOLOGY

#### Case description

This case study includes two wastewater treatment plants that serve the population of Oslo: Vestfjorden Avlopselskap (VEAS) and Bekkelaget Vann og Avlopselskap (BEVAS). The VEAS treatment plant has pre-treatment with screening and aerated grit chambers, before coagulant (iron chloride / polyaluminium chloride) and polymer is added prior to primary sedimentation, followed by biological nitrification–denitrification and secondary sedimentation. The nitrification bio-filters are aerated to provide the aerobic nitrifying bacteria with oxygen, while methanol is supplied as a carbon source for the denitrifying bacteria in the denitrifying step. A small fraction of the treated effluent is used periodically for backwashing the filters, and the backwash water (with some of the washed-out biomass and other filtered particles) is re-circulated back upstream of the screening stage. The sludge treatment includes centrifuges followed by anaerobic sludge digestion that supplies biogas for electricity generation in-plant. Heat recovered from the exhaust gas is utilised for heating requirements within the plant by using sea water as a heat exchanger fluid. The sludge is conditioned and vacuum-dried, while the filtrate water from the drying unit is sent to an ammonia stripping unit, which is fed with nitric acid to remove the ammonia as ammonium nitrate. This is a useful by-product which finds use as a nitrogenous fertiliser. The dried, dewatered and digested sludge is a useful output from the facility, while the sand and the screened impurities from the upstream comprise another waste stream.

At BEVAS, after screening and grit chambers, the flow is split into two. A portion of the flow (stormwater essentially) is
only subject to physical and chemical treatment, while the sewage and combined flow undergo biological treatment as well. Iron chloride and polyaluminium chloride (PAX) are added to the storm-water flow which is then directed to the biofiltration units. The sewage passes through primary sedimentation, before entering biofiltration (aerobic and anaerobic treatment) and secondary sedimentation. Ethanol is used as a carbon source prior to the denitrification process. A part of the sludge from the secondary clarifier is recycled back to the immediate upstream of the biological treatment, while the remaining is thickened, blended with the primary sludge from the primary sedimentation tanks, and sent to the anaerobic digester. The treated effluent from the secondary clarifier and the storm-water flow with the coagulants are then sent to sand filtration units. As the secondary clarifier outflow is not coagulated, some more iron chloride and PAX are added at the inlet of the filters. The biogas generated is used to supply the heat requirements of the treatment plant. The digested sludge is dewatered and conditioned before storage in silos, wherefrom it is trucked away to its end-use destinations.

Inflows – annual consumption of chemicals and energy

Historical data about the consumption of chemicals and energy were obtained from the annual reports of VEAS and BEVAS. Considering that VEAS treats wastewater not just from Oslo but also from neighbouring municipalities, allocations have been made in this paper, on the basis of the proportion of the total flow originating from Oslo. Henceforth in the text, though it is not explicitly stated that only the Oslo portion of the flows into and out of VEAS are considered, the same is implied. Some gaps in the data were plugged by making suitable assumptions.

Outflows – emissions attributed to the influent wastewater constituents

Apart from emissions related to chemicals and energy consumption, there are outflows associated with the constituents of the influent wastewater, which get transformed at the WWTPs and exit to the hydrosphere or the atmosphere.

The carbon and hydrogen in the influent wastewater are lost to the atmosphere. To determine the carbon content of the influent wastewater, the authors make the assumption that the long-chain polymers are totally retained in the sludge. Only subject to physical and chemical treatment, while the sewage and combined flow undergo biological treatment as well. Iron chloride and polyaluminium chloride (PAX) are added to the storm-water flow which is then directed to the biofiltration units. The sewage passes through primary sedimentation, before entering biofiltration (aerobic and anaerobic treatment) and secondary sedimentation. Ethanol is used as a carbon source prior to the denitrification process. A part of the sludge from the secondary clarifier is recycled back to the immediate upstream of the biological treatment, while the remaining is thickened, blended with the primary sludge from the primary sedimentation tanks, and sent to the anaerobic digester. The treated effluent from the secondary clarifier and the storm-water flow with the coagulants are then sent to sand filtration units. As the secondary clarifier outflow is not coagulated, some more iron chloride and PAX are added at the inlet of the filters. The biogas generated is used to supply the heat requirements of the treatment plant. The digested sludge is dewatered and conditioned before storage in silos, wherefrom it is trucked away to its end-use destinations.

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The carbon and hydrogen in the influent wastewater organic solids are split into three streams: i) carbon dioxide, methane and water vapour to the atmosphere, ii) the non-removed COD to the effluent wastewater, and iii) overflow (which varies as a percentage of the treated wastewater) and organics to the sludge. After determining the concentrations of COD in the influent and effluent, the content in the sludge is determined by utilising the data about the volatile solids content of the dry solids, from the reports, and multiplying the same by the factor 1.42 (Droste 1997). By mass balance we can find the COD lost to the atmosphere; however, there are two streams to atmosphere – a direct emission of carbon...
dioxide and water vapour from biological wastewater treatment and an indirect one from the combustion of biogas generated in the anaerobic sludge digesters. The carbon dioxide emissions to air, due to degradation of COD in the wastewater treatment and sludge digestion process, can be considered to be essentially biogenic and therefore ignored. NMVOC emissions are also kept out of the analysis.

Nitrogen in the influent is also split up into three streams. It is emitted by denitrifying bacteria in the biological treatment units, along with a little nitrous oxide (Kampschreur et al. 2008). There is ammonia in the biogas which is oxidised to nitrogen dioxide, assuming complete combustion. Fugitive emissions of ammonia from the plants are difficult to estimate and hence not considered. As far as the stream to land or soil is concerned, at VEAS, the ammonia in the wastewater is stripped with nitric acid and in the process some of the ‘non-denitrified’ nitrates which migrate from the digested sludge to the water during the dewatering process are also removed as ammonium nitrate. The concentration of nitrogen in the digested sludge is assumed to be 2.7 grams per kilogram of dry solids (Droste 1997; page 740). The nitrogen content of the sludge used for agriculture can be assumed to be replacing an equivalent amount of urea (CO(NH2)2).

Phosphorus, unlike carbon and nitrogen, leaves the wastewater treatment plant in two streams: hydrosphere and sludge. A simple mass balance allows one to estimate the mass of phosphorus in the sludge. The phosphorus content of the sludge used for agriculture may be assumed to be replacing an equivalent amount of superphosphate (Ca (H2PO4)2).

As far as heavy metals are concerned, the data are not robust and not available as a time series, and, considering that toxicities are not examined in this analysis (as mentioned in the next section), these are not dealt upon. Suspended solids content of the effluent is less than that of the influent, and has been recorded for both the WWTPs.

The fate of the grit, sand and screenings separated from the influent wastewater, which may find use in road repair or as fuel material, is beyond the scope of the analysis. The transportation of these out of the WWTPs is also not considered.

As far as the sludge is concerned, it has to be mentioned that the masses include the sludge which is transported to the WWTPs from the water treatment plants. Masses of dry solids, nitrogen and phosphorus in the sludge can be calculated. Nitrogen content is calculated by assuming that the concentration of nitrogen in digested sludge is 2.7 g/kg, while phosphorus content is determined by a simple mass balance (difference between influent and effluent concentrations). The end-uses of the sludge, which is basically for agriculture, landscaping and deposits to be used later for landscaping, are known for both the WWTPs. These data are useful in estimating the fertiliser (urea and superphosphate) replacement potential of sludge.

Life cycle assessment method

During its lifetime – construction, use, renovation and expansion, and demolition – a wastewater treatment plant has a range of negative impacts on the environment. As pointed out in Sahely & Kennedy (2007), the operational environmental impacts from urban water systems are much more important than impact from the capital infrastructure over the life cycle. The goal of this analysis is to combine Material Flow Analysis (MFA) and Energy Analysis (EA) with environmental Life Cycle Assessment (LCA) to analyse the life cycle environmental impacts caused by chemicals and energy consumption in the operation phase of the WWTPs in Oslo, as well as the environmental impacts of pollutants in the treated effluents from the WWTPs. The scope and the inventory are explained in brief in the subsequent paragraphs.

Figure 1 illustrates the processes and the flows upstream, within and downstream to the WWTPs. The processes and flows are named/numbered and referred to, in the paragraphs that follow in this sub-section. The flows are identified as X_{ij} in accordance with Material Flow Analysis practice, where ‘i’ is the origin and ‘j’ is the destination of the flow. The flows and the processes have been described in the subsequent paragraphs.

The flow of wastewater is from left to right – from Process 1 to Process 3, as shown. Process 2 depicts the wastewater treatment plants and Processes 2a, 2b and 2c represent the processes within the wastewater treatment plants. The treated effluent flows out (X_{2a,3}) to the fjord. The wastewater collection process is beyond the scope of the LCA in this paper and hence outside the LCA system boundary. Processes 5, 6 and 7 represent the production of chemicals and energy elements for supply to Process 2, while Processes 8, 9, 10 and 11 depict the end-uses of the by-products of wastewater and sludge treatment. Process 12 represents production avoided in other sectors of the economy owing to the reuse of the said by-products. Process 5 represents the recipient of treated wastewater and Process 4 the atmospheric compartment for emissions to air.

The environmental impacts caused during the use phase of the WWTPs can be split into three distinct components – attributable to chemicals, energy and the effluent wastewater. As far as chemicals are concerned, the production and transport of iron chloride, iron sulphate, PAX, nitric acid,
calcium hydroxide, polymer, methanol and ethanol are considered (Process 5 and flow X5,2 in Figure 1). Ethanol and methanol result in the emission of carbon dioxide and methane from biogas in the anaerobic digester when the biomass in sludge gradually decays. This ultimately manifests as emission of carbon dioxide when the biogas is utilised (a part of X2c,4). Impacts due to energy consumption are attributed to the production and transport/transmission of fuel oil and electricity (Processes 6 and 7, and flows X6,2 and X7,2); as well as emissions from the combustion of fuel oil on the one hand (a part of X2,4), and biogas produced within the plants on the other. The biogas is assumed to be combusted completely, resulting in the emissions of only carbon dioxide (partly biogenic and partly originating from the ethanol and methanol referred to earlier), water vapour, sulphur dioxide and nitrogen dioxide (a part of X2,4). The use of biogas (Process 10) for heating avoids the use of an equivalent amount of natural gas (X12,4). Likewise, in-plant, biogas-derived electricity (Process 2c) reduces the demand exerted by the plant/s on the grid (X7,11). There are emissions associated with the production and maintenance of the energy converters deployed at the plants, but these are assumed negligible in this analysis. Suitable changes are thereby made in the datasets. The authors have decided to retain the transportation/power transmission details as-are; there is of course scope for investigating and determining with greater accuracy how, and from where, heating oil, diesel and electricity are transported/transmitted for use at the plants. Nitrogen dioxide and sulphur dioxide emissions are taken into account. All ammonia is assumed to be completely oxidised to nitrogen dioxide. Also bracketed under energy are the avoided impacts, due to the substitution effect of biogas, for natural gas and electricity. Attributed to the effluent wastewater are the environmental impacts related to the nitrogen and phosphorus in the treated effluent and the untreated overflow (X2c,3), and the small amounts of nitrous oxide from the aerobic/anaerobic wastewater treatment processes (part of X2,4). The potential for the WWTP's by-products to replace other chemicals (ammonium nitrate, superphosphate and urea) is taken into consideration.
In this paper, the authors have decided to interpret these as offsets to the impacts caused by the chemicals consumption. Alternately, these could be looked upon as offsets to impacts attributable to the effluent wastewater.

As referred to earlier, the non-inclusion of the carbon dioxide emissions from wastewater treatment is due to its biogenic origin. Volatile organic compounds, such as ammonia and methane emissions from wastewater treatment processes, are difficult to estimate with any reasonable degree of accuracy (all parts of X₂₄). The presence of nitrogen, phosphorus and organic carbon in the sludge is advantageous, especially as the sludge is used for agriculture and silviculture. Heavy metals in the sludge and the effluent impart toxicity, but, as this paper restricts itself to environmental impact potentials other than toxicity, the various toxicities associated with heavy metals in sludge and effluent water are not accounted for due to larger uncertainties for these issues. Besides, most of the sludge finds use in agriculture and silviculture as the heavy metal content is well below the maximum allowable limits (Venkatesh et al. 2009). The emissions related to the transportation of the sludge, and the ammonium nitrate produced as a by-product, to end-use destinations have not been considered in this analysis. Sea water, oxygen consumption, and the avoidance potential of screenings (for fuel) and recycled wastewater (for treated water supply) have not been taken into account.

In the absence of comprehensive data about the distances over which chemicals are transported, the authors assume that calcium hydroxide (550 km) and PAX (80 km) are purchased from the same source that Oslo’s WTPs get these supplies from (Aasebo 2009). As far as polymers are concerned, while the WTPs get their supply from two sources – one in Sweden and another in Norway – this paper assumes that the WWTPs rely only on the Norwegian supplier (80 km). For each of the others, Norway-based supply sources and transport by road over a fixed average distance are considered. A sensitivity analysis is done by changing this assumed fixed average distance (50 km, 100 km and 150 km are considered). Data for poly-aluminium chloride are not available from the Ecoinvent database. Instead of contacting suppliers or looking for other sources, the authors recalled that the aluminium chloride is produced by reacting aluminium hydroxide with hydrochloric acid, and that the reaction is exothermic. Emission data for these reactants were used and the exothermicity of the reaction was used as a premise for neglecting energy inputs to the process. This would be a close approximation. Polyacrylamide is used as a polymer, but datasets for polyacrylamide are not available in the Ecoinvent database. Hence, the authors proceed with EDTA (ethylenediaminetetraacetic acid), basing the approximation on the knowledge that the nitrogen in the polymer plays a key role in the coagulation/flocculation. The SIMAPRO 7.1.5 software (Pré Consultants, 2008) was used in concert with the Ecoinvent database (Swiss Centre for Life-Cycle Inventories, 2008) and the CML 2001 impact assessment method (CML, 2002). The normalisation and weighting values for the CML method are also listed in Appendix I. The authors have focused only on the potentials for six impact categories–abiotic depletion (ADP), acidification (AP), global warming (100 years; GWP), ozone depletion (ODP), photochemical ozone creation (PCOP) and eutrophication (EP)–as traditionally these are the key impacts considered in the analysis of water and sanitation networks. The abbreviations for the impacts are used in the illustrations.

The Norwegian electricity mix has been considered for this analysis. A sensitivity analysis is performed at the end, by replacing the Norwegian electricity mix with the Nordic electricity mix (Appendix III), since the latter may be more relevant given the use of marginal electricity for prospective LCA calculations.

The specific aggregated environmental impacts per m³ wastewater, attributable to energy and chemicals, are also compared with each other. While doing so, it is borne in mind, that there are impacts attributable to the effluent wastewater as well. The greenhouse gas emissions are isolated, expressed in terms of CO₂-eq per cubic metre of wastewater handled, for chemicals and energy separately.

RESULTS AND DISCUSSION

The population served by the two WWTPs has increased from 509,000 in year 2000 to 560,000 in year 2007. The per-capita value dropped from a high of 235.4 cubic metres per annum in 2000 to 199.1 cubic metres per annum in 2007 (a drop of 15.4%), cfr. Table 1. The corresponding trends in the energy and chemicals consumption, per cubic metre of treated wastewater, are discussed below.

Quantities and costs

Table 2 lists values associated with energy consumption at the WWTPs. It is seen that from 2001 to 2007 the electricity purchased from the grid remained fairly constant at around 28–30 GWh, while that produced in-house from the biogas rose steadily. In year 2000, the in-house production accounted for 22.9% of the total electricity consumed, and in year 2007 this increased slightly to 24.3%. On a
per-cubic-metre-of-wastewater-treated basis, the total energy consumption rose from 0.61 kWh to 0.78 kWh (by 27%), the average for the eight year period being 0.75 kWh.

There is a 34% rise in the in-house generation of electricity from biogas, from 7.1 GWh to 9.52 GWh, while the volume of biogas generated (volume measured at NTP conditions) rose by over 70%, from 8.1 to 14 million cubic metres. At present, it is only at VEAS that electricity is generated using the biogas produced in-house. With a rise in efficiency of conversion – heat to electricity – it is quite possible that in the future the electricity yield could be improved further.

Figure 2 lists all the values associated with chemicals consumption. In year 2000, 22.5 million kilograms of chemicals were consumed in the WWTPs in Oslo, or 189 grams per cubic metre of wastewater treated. Iron chloride as coagulant accounts for the bulk of the consumption. According to Droste (1997), iron sulphate has much greater coagulating ability for both positive and negative species than iron chloride. When iron sulphate made an entry in the system, in 2001, the overall demand for iron salts dropped, and iron chloride consumption decreased by 60%. Thereafter, the chemical masses remained well below the level of 15 million kilograms. Figure 2 shows that the masses consumed have dropped from 22.5 to less than 15 million kilograms during the period.

**Table 1** | Population served and wastewater treated at Oslo’s WWTPs

<table>
<thead>
<tr>
<th>Year</th>
<th>Population served (million)</th>
<th>Wastewater treated (million m³)</th>
<th>Per capita wastewater treated (m³/cap/year)</th>
</tr>
</thead>
<tbody>
<tr>
<td>2000</td>
<td>0.509</td>
<td>119.8</td>
<td>235.4</td>
</tr>
<tr>
<td>2001</td>
<td>0.513</td>
<td>110.3</td>
<td>215.0</td>
</tr>
<tr>
<td>2002</td>
<td>0.517</td>
<td>102.5</td>
<td>198.3</td>
</tr>
<tr>
<td>2003</td>
<td>0.522</td>
<td>105.6</td>
<td>202.3</td>
</tr>
<tr>
<td>2004</td>
<td>0.530</td>
<td>109.5</td>
<td>206.6</td>
</tr>
<tr>
<td>2005</td>
<td>0.538</td>
<td>111.4</td>
<td>207.1</td>
</tr>
<tr>
<td>2006</td>
<td>0.549</td>
<td>119.4</td>
<td>217.5</td>
</tr>
<tr>
<td>2007</td>
<td>0.560</td>
<td>111.5</td>
<td>199.1</td>
</tr>
</tbody>
</table>

**Table 2** | Energy consumption at BEVAS and VEAS

<table>
<thead>
<tr>
<th>Year</th>
<th>Hydropower (GWh)</th>
<th>Total biogas combusted (million m³)</th>
<th>Biogas-electricity (GWh)</th>
<th>Heating oil</th>
<th>Biogas-derived heat</th>
</tr>
</thead>
<tbody>
<tr>
<td>2000</td>
<td>23.87</td>
<td>8.12</td>
<td>7.10</td>
<td>1.70</td>
<td>40.23</td>
</tr>
<tr>
<td>2001</td>
<td>29.05</td>
<td>8.58</td>
<td>7.37</td>
<td>1.43</td>
<td>42.22</td>
</tr>
<tr>
<td>2002</td>
<td>30.12</td>
<td>10.56</td>
<td>7.37</td>
<td>1.16</td>
<td>41.23</td>
</tr>
<tr>
<td>2003</td>
<td>28.73</td>
<td>11.54</td>
<td>8.49</td>
<td>0.89</td>
<td>43.08</td>
</tr>
<tr>
<td>2004</td>
<td>29.73</td>
<td>11.87</td>
<td>8.09</td>
<td>1.39</td>
<td>45.99</td>
</tr>
<tr>
<td>2005</td>
<td>28.74</td>
<td>12.55</td>
<td>9.96</td>
<td>0.99</td>
<td>51.25</td>
</tr>
<tr>
<td>2006</td>
<td>30.21</td>
<td>12.66</td>
<td>8.78</td>
<td>1.21</td>
<td>49.54</td>
</tr>
<tr>
<td>2007</td>
<td>29.65</td>
<td>14.05</td>
<td>9.52</td>
<td>1.11</td>
<td>47.05</td>
</tr>
</tbody>
</table>
despite a slight increase the last two years. The results correspond to 189 grams per cubic metre in year 2000 and 131 grams in year 2007. The data also confirm a fairly equal consumption (kg per year) of six chemicals, plus a much smaller use of polymers.

**Environmental impacts**

The major environmental impact from *energy consumption* is acidification, courtesy of the emissions of NOx and sulphur dioxide from biogas combustion. Due to 70% increased consumption of biogas, cfr. Table 2, it can be observed that the aggregated environmental impact per cubic metre of wastewater treated nearly doubled between 2000 and 2007; see Figure 3. What is noteworthy, besides the domination of acidification, is the negative abiotic depletion and negative global warming potentials. This can be explained by the role of biogas in obviating the need for production of an equivalent amount of natural gas. The ozone depletion potential results are not included in the figure, considering their relative insignificance.

In the case of *chemicals consumption* the major impact was global warming, followed by abiotic depletion; see Figure 4. Interestingly, it is these two impacts which had a subtractive effect on the score in the case of energy. The avoided production of urea, superphosphate and ammonium nitrate has been allocated to chemicals, as mentioned earlier in the methodology section. This tends to bring down the aggregated impact score quite significantly (by over 50% in most cases). From the results in Figure 4, which depicts the environmental impacts for chemicals excluding the transportation-related impacts, it is seen that the score halved over the period. Ferric chloride contributed to the bulk of the impacts in year 2000, and when a substantial proportion of it was substituted by ferric sulphate the impacts decreased conspicuously. The small spurt in year 2003 can be explained by the increase in the use of ferric chloride during that year, relative to 2002. The carbon dioxide originating from ethanol and methanol used as carbon sources in the denitrification process was a significant contributor to the global warming potential of chemicals. In fact, it accounted for between 10 and 25% of the aggregated score. The greenhouse gas emissions per gram of chemicals amounted to 10.07 grams of CO2-eq in year 2000, peaked at 11.8 grams in 2003 and thereafter dropped down to 7.76 grams in 2007.

As far as transportation-related emissions are concerned, with the assumptions made, it is found out that these accounted for between 1.9% and 5.5% of the total emissions related to chemicals. When an average distance of 50 kilometres is assumed for transportation of the iron salts, nitric acid, ethanol and methanol, the percentage ranges from 1.9% to 3.7%. When a distance of 100 kilometres is considered, the range is 2.6% to 4.6%. With 150 kilometres, it is 3.3% to 5.4%. On the premise that the distances may not be greater than 150 kilometres, it can be concluded that the contribution of transportation to the life-cycle environmental impacts attributed to chemicals was minimal. By the same token, one can also now justify the omission of the sludge transportation process.

**Chemicals vis-à-vis energy**

Figure 5 sums up the results of the previous two sub-sections with a comparison between environmental impacts.
associated with energy and chemicals consumption (excluding transportation-related emissions for chemicals). The environmental impacts attributable to energy consumption were always greater than those due to chemicals consumption. The ratio of the specific aggregated environmental impact score for energy to that for chemicals increased over time, from close to 1 in year 2000 to over 4.5 in 2007. The emissions due to energy use were primarily a consequence of the sulphur dioxide and NOx emission from combustion of biogas. These emissions kept increasing, as more and more biogas was consumed in-plant. In the case of chemicals, on the other hand, the fact that the avoided production of urea, superphosphate and ammonium nitrate offset a good deal of the impacts caused by the chemicals kept the overall score lower.

If one would look into the greenhouse gas emissions per unit volume of wastewater treated, it decreased from 1.9 kilograms of CO2-eq to 1.0 kg of CO2-eq in the case of chemicals. In absolute terms, the corresponding values are 230 million kg and 113 million kg (a decrease of more than 50%). For energy, as illustrated in Figure 3, the avoided greenhouse gas emissions increased from 2.7 grams to 6.0 grams of CO2-eq per cubic metre of wastewater treated.
In absolute terms, the corresponding values are 330 and 663 metric tons, i.e. a doubling in 8 years. The greenhouse gas emission-related values are tabulated in Table 3. One could discuss in depth issues about other emissions and LCA impacts as well, but there is not room for that in this paper.

If the overall environmental impacts are taken into consideration, it is the effluent wastewater (X_{2a,3}) which dominated, accounting for between 73% and 80% of the total impacts throughout the eight year period. While acidification dominated in the case of energy and global warming accounted for the largest chunk of the aggregated score in the case of chemicals, for the effluent wastewater eutrophication was the key impact, not surprisingly. It follows that, when the operation phase of the WWTPs is studied for environmental impacts, eutrophication generally accounts for the lion's share of the total environmental impact score. This is also confirmed by Lassaux et al. (2007), for a Belgian case study in their paper. Figure 6 shows that there were no appreciable changes in the total environmental impacts (measured as the aggregated score) over time. In the latter years, the avoidance potential of by-products introduces a net negative abiotic depletion potential.

Eutrophication was always the major concern, and it will most likely continue to be so. Source reduction of nitrogen and phosphorus in the wastewater – the importance of which is emphasised by Tollan (2010) – along with enhanced nitrogen and phosphorus removal treatment processes both have their own limitations. The latter would call for higher energy and chemicals consumption. The impacts owing to the consumption of additional chemicals and energy may even more than offset the reduction in eutrophication which they would bring about. What can be accomplished by reducing the impacts attributable to chemicals and energy consumption is little.

Table 3 | Greenhouse gas (GHG) emissions from energy and chemicals consumption in WWTPs

<table>
<thead>
<tr>
<th>Indicator/Year</th>
<th>2000</th>
<th>2001</th>
<th>2002</th>
<th>2003</th>
<th>2004</th>
<th>2005</th>
<th>2006</th>
<th>2007</th>
</tr>
</thead>
<tbody>
<tr>
<td>Specific GHG emissions – energy use (g CO₂-eq per m³ wastewater treated)</td>
<td>-2.8</td>
<td>-3.5</td>
<td>-4.5</td>
<td>-5.9</td>
<td>-4.8</td>
<td>-7.5</td>
<td>-5.7</td>
<td>-5.9</td>
</tr>
<tr>
<td>Specific GHG emissions – chemicals consumption (g CO₂-eq per m³ wastewater treated)</td>
<td>1,909</td>
<td>1,374</td>
<td>1,164</td>
<td>1,521</td>
<td>1,205</td>
<td>997</td>
<td>1,120</td>
<td>1,014</td>
</tr>
<tr>
<td>Energy-related emissions (g CO₂-eq per kWh energy consumed)</td>
<td>-4.5</td>
<td>-5.0</td>
<td>-5.8</td>
<td>-7.7</td>
<td>-6.2</td>
<td>-9.1</td>
<td>-7.6</td>
<td>-7.6</td>
</tr>
<tr>
<td>Chemicals-related emissions (g CO₂-eq per g chemicals consumed)</td>
<td>10.1</td>
<td>9.7</td>
<td>8.3</td>
<td>11.8</td>
<td>10.3</td>
<td>8.9</td>
<td>9.9</td>
<td>7.8</td>
</tr>
</tbody>
</table>

Figure 6 | Environmental impact from effluent wastewater, energy and chemicals. Oslo’s WWTPs, 2000–2007.
Sensitivity analysis

The Norwegian electricity mix which has been considered in the foregoing analysis is 99% hydropower and has a negligible fossil-fuel component. The Nordic mix – which is the average of the Danish, Swedish, Norwegian and Finnish electricity mixes – has a non-renewable fossil-fuel component (hard coal, oil, natural gas, industrial gas and peat) of 21.8% (calculated from the Ecoinvent database – Refer Appendix III). As recorded in Norwegian Ministry of Petroleum and Energy (2008), during the period 2000–2007, Norway imported over 60 TWh of electricity through the Nordic grid. It follows that if using the Norwegian electricity mix would underestimate the environmental impacts, and the only way to take into account this import component, although not very accurately, would by performing a sensitivity analysis using the Nordic electricity mix. Using the Nordic electricity mix in the analysis would, however, lead to overestimates of all or some of the environmental impacts.

When the Nordic electricity mix is considered, there are no appreciable changes in the environmental impacts due to chemicals consumption. The increase in impacts owing to electricity consumption in the production of chemicals consumed is almost offset by the avoided electricity consumption courtesy of the substitution of urea and superphosphate with sludge fertiliser and ammonium nitrate from the plant. The impacts attributable to energy also increase slightly with the Nordic mix – over a range of 2% to 7%. While there are savings in GHG emissions (negative GWP-100 potential in other words) with Norwegian electricity throughout the study period, net positive emissions result when the Nordic mix is used, for all years except year 2005. The increase in specific GHG emissions with the Nordic mix vis-à-vis the Norwegian mix is between 92% (in 2005) and 320% (in 2000).

While the impacts caused by the effluent wastewater remain the same irrespective of which electricity mix is considered, it follows that with the Nordic mix the percentage shares of energy and chemicals will increase.

CONCLUSIONS, LIMITATIONS AND RECOMMENDATIONS

Harremoes (1998) prophesied over a decade ago in a paper on urban water systems: “Mass balances and life cycle analyses will have a much greater effect on our daily life than we can imagine today”. While the Material Flows Analysis (MFA) and Energy Analysis (EA) provide an understanding of the progress attained by the WWTPs, the Life Cycle Assessment (LCA) gives an insight into how the environmental footprint of the plants has evolved over time. Knowledge about the key contributors to the impacts is obtained, and the realistic possibilities of shrinking the said footprint in the future can be investigated, with a more thorough process analysis. The “avoidance potential” of by-products (sludge, screenings, biogas etc.) is encouraging, and calls for the entrenchment of sustainable end-uses for these by-products. It would be apt to mention the Gothenburg Protocol signed by Norway along with US, Canada and other European countries. The protocol came into effect on the 17th May 2005 (Norwegian Ministry of Petroleum and Energy 2008) and aims to solve the environmental problems of acidification, eutrophication and ozone level depletion.

The analysis in this study has a number of limitations due to unavoidable assumptions and lack of data. A more robust and complete database would enable a more comprehensive, expansive and inclusive MFA/EA-LCA. Numerous assumptions have been made in the analysis, and have been indicated at the relevant junctures in the methodology section. It could be argued that what have been calculated are the least estimates of the environmental impacts. A more elaborate analysis might include issues such as toxicity, land use, transportation, and the construction of the WWTPs themselves (including production of concrete, metals, machinery, etc.). For instance, the effect of micro-pollutants like pharmaceuticals and endocrine disrupters in the wastewater, on human and animal life, can be modelled if accurate data would be made available. One could also include the use of other life cycle impact assessment methods. On the other hand, we do believe that we have covered the most relevant and important issues, and come to fairly robust conclusions.

A reduction in the use of water and source control of the water-borne residuals will possibly free some funds for capital investments and enable the utility to improve the energy and materials efficiency. BEVAS is exploring the possibility of refining and selling biogas to the transportation sector. Methanol may be replaced by glycerine (a by-product from a biodiesel plant) and this would have an impact on the expenses on chemicals, and also render the associated carbon dioxide emissions during microbial decay and subsequent combustion of the biogas, biogenic (BEVAS, Oslo 2007). Overall there are several improvement options for the future.

However, a main conclusion from this study is that, in the case of Oslo’s WWTPs, the eutrophication potential of treated wastewater has the highest contribution to environmental life cycle impact, of all impact categories, and much larger than the sum of the other ones. This demonstrates that the first priority of WWTPs should be to minimise the concentration
of nutrients in treated wastewater. Then, since the environmental impacts from energy consumption are significantly larger than that of chemicals, cfr. Figure 5, the second priority should be to minimise the consumption of energy from outside, and try use renewable energy whenever possible. We also found significant avoided benefits of downstream use of by-products from the sludge treatment processes, such as digested sludge, ammonium nitrate, excess biogas and electricity. WWTPs should therefore aim at optimising their by-product usages. The environmental impact from chemicals production is the least important issue; however, an optimum selection and use of chemicals is of course crucial to achieve low concentrations of pollutants in the wastewater, and therefore a major concern of any WWTP.

The authors are working more closely with the Oslo VAV to do more in-depth studies of metabolism and environmental impacts for the whole urban water system, as well as the economics of wastewater treatment in the city. More results are likely to be published in the future. A study similar to this one has been carried out for the water treatment plants in Oslo in Venkatesh and Brattebo (2010).

REFERENCES


PRÉ Consultants 2008 Simaprio 7.1.5. Amersfoort, the Netherlands: Pré Consultants.


Toftdahl, O. Managing Director, Bekkelaget WWTP, Oslo. Personal communication in June 2009.


APPENDICES

Appendix I

Datasets used from Ecoinvent Database (7.18)

1. Ammonium nitrate, as N, at regional storage / RER U (no changes to the dataset).
2. EDTA, ethylenediaminetetraacetic acid, at plant / RER U is used – Norwegian and Nordic electricity mixes are used in turn for the sake of comparison. Usually, polyacrylamide is used as a polymer, but datasets for polyacrylamide are not available in the Ecoinvent database. Hence, the authors proceed with EDTA, basing the approximation on the knowledge that the nitrogen in the polymer plays a key role in the coagulation/flocculation.
3. Electricity, medium voltage, at grid – Norwegian and Nordic electricity mixes are tried in turn. It is assumed that the voltage is stepped down to medium for use in the treatment plants.
4. Ethanol, from ethylene, at plant / RER U, using Norwegian and Nordic electricity mixes in turn.
   a. Hydrochloric acid, 30%, at plant / RER U – no changes.
   b. Aluminium hydroxide, at plant / RER U, with the field pertaining to light fuel oil use changed to the average European value and both Norwegian and Nordic electricity mixes used in turn, for the sake of comparison. (PAX has the chemical formula (according to Wikipedia) of $\text{Al}_2\text{Cl}_{12}(\text{OH})_{24}$. So, 924 grams of aluminium hydroxide would require 1,460 grams of hydrochloric acid dissolved in water (30% in water) to form 1,140 grams of PAX in solution (water mass excluded) ... that means, 0.81 mass units of hydroxide would require 1.28 mass units of hydrochloric acid solution to give 1 mass unit of PAX. However, the molecular formula of PAX which is used at the Oslo WTPs may not be what has been assumed in this analysis.)
5. Iron (III) chloride, 40% in water, at plant / CH U, with both the Norwegian and Nordic electricity mixes used in turn.
6. Iron sulphate, at plant / RER U, with both Norwegian and Nordic electricity mixes used in turn. (This is supplied generally as a 50% solution.)
7. Light fuel oil burned at boiler, non-modulating 100 kW / CH U, with Norwegian and Nordic electricity mixes considered in turn, in lieu of the corresponding Swiss dataset. This includes light fuel oil production, transport to end-user and final combustion.
8. Lime, hydrated, packed, at plant / CH U – no changes made. Here, we do not differentiate between burnt (calcined) lime and hydrated lime.
9. Methanol, at regional storage / CH U (no changes made to the dataset).
10. Natural gas, at consumer / RNA U – with the unit production process changed to Natural gas, at production offshore / NO U. This takes into account production, transport and distribution of natural gas to consumer.
11. Nitric acid, 50% in water, at plant / RER U, with both Norwegian and Nordic electricity mixes used in turn.
12. Single superphosphate, as P$_2$O$_5$, at regional storage / RER U (both Norwegian and Nordic electricity mixes used in turn).
13. Transport, lorry, 16–32t, EURO5/RER U – changes made to include only the production, transport and consumption of low-sulphur diesel fuel.
14. Urea, as N, at regional storage / RER U (both Norwegian and Nordic electricity mixes used, in turn); assume that urea would be imported from within Norway.

Appendix II

Datasets used from Ecoinvent Database (7.18)

Normalisation and weighting factors for the CML 2001 methodology

<table>
<thead>
<tr>
<th>Normalisation factor*</th>
<th>Weighting factor**</th>
</tr>
</thead>
<tbody>
<tr>
<td>ADP 1.48E+10</td>
<td>5</td>
</tr>
<tr>
<td>AP 2.43E+10</td>
<td>5</td>
</tr>
<tr>
<td>EP 1.25E+10</td>
<td>5</td>
</tr>
<tr>
<td>GWP(100) 4.81E+12</td>
<td>16</td>
</tr>
<tr>
<td>ODP 8.33E+07</td>
<td>5</td>
</tr>
<tr>
<td>PCOP 8.22E+09</td>
<td>8</td>
</tr>
</tbody>
</table>

*Western European values for 1995
**USEPA
Appendix III

Nordic electricity mix and the non-renewable fossil component of the same

<table>
<thead>
<tr>
<th>Energy carrier</th>
<th>Shares of energy carriers in electricity mix</th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Norwegian</td>
<td>Swedish</td>
<td>Danish</td>
<td>Finnish</td>
<td>Nordic mix</td>
</tr>
<tr>
<td>Hard coal</td>
<td>0.03%</td>
<td>1.29%</td>
<td>51.20%</td>
<td>20.35%</td>
<td>10.68%</td>
</tr>
<tr>
<td>Oil</td>
<td>0.01%</td>
<td>1.21%</td>
<td>11.20%</td>
<td>0.80%</td>
<td>1.87%</td>
</tr>
<tr>
<td>Natural gas</td>
<td>0.32%</td>
<td>0.28%</td>
<td>23.50%</td>
<td>14.80%</td>
<td>6.13%</td>
</tr>
<tr>
<td>Industrial gas</td>
<td>0.05%</td>
<td>0.73%</td>
<td>0.00%</td>
<td>1.27%</td>
<td>0.61%</td>
</tr>
<tr>
<td>Hydropower</td>
<td>99.06%</td>
<td>55.03%</td>
<td>0.09%</td>
<td>16.70%</td>
<td>56.90%</td>
</tr>
<tr>
<td>Wind</td>
<td>0.27%</td>
<td>0.31%</td>
<td>12.40%</td>
<td>0.13%</td>
<td>1.55%</td>
</tr>
<tr>
<td>Wood and cogeneration</td>
<td>0.26%</td>
<td>2.50%</td>
<td>1.61%</td>
<td>10.57%</td>
<td>3.73%</td>
</tr>
<tr>
<td>Nuclear</td>
<td>0.00%</td>
<td>38.60%</td>
<td>0.00%</td>
<td>25.86%</td>
<td>21.91%</td>
</tr>
<tr>
<td>Lignite and peat</td>
<td>0.00%</td>
<td>0.05%</td>
<td>0.00%</td>
<td>10.99%</td>
<td>2.57%</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td><strong>21.86%</strong></td>
</tr>
</tbody>
</table>