Life-Cycle Energy Use and Greenhouse Gas Emissions Inventory for Water Treatment Systems

Alina I. Racoviceanu[1]; Bryan W. Karney, M.ASCE[2]; Christopher A. Kennedy[3]; and Andrew F. Colombo[4]

Abstract: Given the rising concerns over scarce energy resources and global climate change, life-cycle inventories focusing on energy use and greenhouse gas (GHG) emissions were developed for the City of Toronto municipal water treatment system (WTS). Three processes within the facility use phase of the life cycle were considered: Chemical production, transportation of materials, and water treatment plant operation. The impacts of chemical manufacturing were estimated using the economic input-output life-cycle assessment model, while the inventories for transportation and operational environmental effects were based on data from the GHGenius model and regionally averaged data. Operational burdens, 60% of which are attributed to on-site pumping, accounted for 94% of total energy use and 90% of GHG emissions. By contrast, transportation-related energy use and emissions were deemed insignificant. The normalized energy use of the studied WTS was found to be between 2.3 and 2.5 MJ/m^3 of water treated. Water conservation practices are recommended as abatement strategies for the energy use and GHG emissions associated with water treatment. The limitations and uncertainties introduced by selected model parameters and through combining various estimation methodologies are discussed, as is the model's relevance.

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Introduction

An emerging impetus to measure and evaluate the long-term sustainability of water infrastructure has been noted among economists, engineers, environmental and social scientists. Approaches for assessing the environmental performance of projects and services include the use of environmental sustainability indicators (Harger and Meyer 1996; Lundin and Morrison 2002) and ecological footprint analysis (Nicholson et al. 2003). In this context, Life-Cycle Assessment (LCA) has emerged as a tool for promoting sustainable development and increasing environmental awareness. LCA is a well-established and standardized method designed to estimate and reduce the environmental burdens associated with all phases of the life of a product, process, or service, starting with raw material extraction, through manufacture to use and recycling or final disposal. This "cradle-to-grave" approach identifies major environmental impacts related to system inputs and outputs, flags any hazards, and highlights possibilities for improvements (USEPA 1993). Both business and industry have started to recognize the usefulness of LCA as a tool for saving natural resources and energy while minimizing pollution, waste, and financial expenditure.

Traditionally LCA has been applied to consumer goods and services, however, its field of application has expanded to include infrastructure. The sustainability of urban water systems and solutions to improve their performance have been recently questioned and examined by some researchers (Lundin and Morrison 2002; Friedrich 2002; Hermann and Hasse 1997; van der Hoek et al. 1999; Lundie et al. 2004).

A handful of researchers have specifically considered the City of Toronto's water system. The sustainability of this urban water system was assessed through a set of criteria (i.e., environmental, economic, engineering, and social) and indicators (e.g., energy and chemical use, greenhouse gas (GHG) emissions, infrastructure renewal expenditures, and leaks detection) developed by Sahely et al. (2005). A historical cost review of Toronto's urban water infrastructure indicated a decreasing trend in the operational cost of providing drinking water over the past 45 years (Pharasi and Kennedy 2002). The energy potential of municipal wastewater systems was examined by Shizas and Bagley (2004). An analysis of the Greater Toronto Area urban metabolism (Sahely et al. 2003) revealed an increase in water supply higher than wastewater discharges over the study period. However, to date, the City of Toronto's water treatment system (WTS) has not been examined from a life-cycle perspective.

The aim of this study is to quantify the total energy use and one specific environmental burden, namely GHG emissions, for a conventional WTS that is typical of major urban centers in North America, using a life-cycle-based approach. The energy use and GHG emissions inventories are characterized for the facility use phase only, as these effects are more significant relative to those associated with the construction or decommissioning stages of water systems (Friedrich 2002; Lundie et al. 2004; Lundin et al. 2000). The study is one of the few of its kind to contemplate the

[1]Graduate Student, Dept. of Civil Engineering, Univ. of Toronto, Toronto ON, Canada M5S 1A4.

[2]Professor, Dept. of Civil Engineering, Univ. of Toronto, Toronto ON, Canada M5S 1A4 (corresponding author). E-mail: karney@ecf.utoronto.ca

[3]Associate Professor, Dept. of Civil Engineering, Univ. of Toronto, Toronto ON, Canada M5S 1A4.

[4]Research Associate, Dept. of Civil Engineering, Univ. of Toronto, Toronto ON, Canada M5S 1A4.

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life-cycle energy use and GHG emissions of a North American WTS.

The heightened attention paid to energy use and GHG emissions is the main reason for using them as environmental indicators in this study. It is well understood how energy use contributes to resource depletion and environmental degradation, especially as fossil fuels establish the basic source of energy for most activities. Consequently, some researchers have sought to reduce the energy used in fabricating products or providing services (in this case, drinking water treatment). For this analysis, gross energy requirement is selected as a screening indicator to reveal opportunities for energy and GHG reduction throughout the life cycle of the system. The results from the Life-Cycle Inventories (LCIs) are then used to identify processes with the greatest environmental impact so that they can be targeted for improvement. Limiting the study to energy use/GHG inventories reduces data collection and processing effort required for a complete LCI. Further, streamlining the LCA to inventory analysis circumvents several methodological problems, such as the complex and uncertain assessment associated with impact and improvement analysis. The information overload stemming from the inclusion of these two traditional LCA components derives from the interconnectedness between different sectors of the economy and the enfolded character of society, ecosystems, and the biosphere. It can render a comprehensive LCA impractical for use in any timely, accurate, and cost-effective design and decision-making regime (Vanderburg 2000).

LCA Studies for Water Sector

Over the last decade, many LCA studies have been conducted for electronic products, building materials, packaging, chemicals, etc. (EEA 1998) with only a small segment of LCA research directed toward the water sector. Of the published work that focuses on municipal drinking and wastewater systems, the majority are comparative LCAs that assess the environmental performance of new treatment technologies.

Studies of wastewater treatment systems (WWTS) are the most common because of the opportunity for energy recovery as well as their higher operational energy requirements. LCA studies have been used to demonstrate the environmental superiority of new treatment technologies, such as ultraviolet disinfection over traditional chlorination/dechlorination (Das 2002) or for selecting the most cost effective wastewater treatment technology (Tsagarakis et al. 2002). Lundin et al. (2000) overview LCAs for WWTS and note the influence of system boundaries and scale on the analysis outcome.

Several studies assess the environmental impacts of water distribution infrastructure. Dennison et al. (1999) contemplated alternative strategic options for water mains infrastructure management, such as dual-use or reuse of abandoned mains for housing other infrastructure (i.e., fiber optic cables) and the recovery of metal waste from repair works. Herz and Lipkow (2002) employed LCA to quantify the mass, energy flows, and GHG emissions associated with the installation and replacement of water mains and sewer conduits in Germany. Investigating the energy expenditures pertinent to water supply conduits (the New York City water supply tunnels), Filion et al. (2004) examined optimal pipe replacement time according to a gross energy use criterion that served as a proxy measure of environmental burden by considering the energy flows associated with pipe fabrication, replacement, disposal, and pumping.

Only a few studies have focused on the performance of WTSs. A LCA study of Bologna's water and wastewater systems identified pumping as entailing the highest environmental impact, whereas the production and transport of chemicals used in conventional treatment accounted for only 10% and a negligible amount of total energy use, respectively (Tarantini and Ferri 2001). Herrmann and Hasse (1997) recommended a combination of strategies including decentralized rainwater collection, leak repair, water conservation, and wastewater recycling in order to realize water savings in a Bavarian water system. A dual water supply system, using both household water (produced from a local surface water source) and drinking water, was selected by van der Hoek et al. (1999) as the optimum alternative for reducing water use in a new residential development in Amsterdam. A LCA model of Sydney's water and wastewater system was developed to assess the environmental impacts of alternative future scenarios (Lundie et al. 2004). In comparing the environmental impacts from conventional treatment and membrane filtration for potable water treatment, Friedrich (2002) indicated that the generation of electricity consumed during treatment involved the highest environmental burden. A similar LCA, assessing the energy use and associated environmental outputs of two disinfection systems, indicated that conventional chlorination/dechlorination outperformed an UV/filtration system (Beavis and Lundie 2003). Lundin and Morrison (2002) used a LCA-based decision-making tool to select environmental sustainability indicators for urban water systems. Ilomaki et al. (2003) determined that a modified public water supply system in Hanoi (providing increased pressure and water quality) was able to realize energy savings while lessening environmental impacts. The use of rainwater instead of drinking water for laundry and toilet flushing can result in energy savings, economies of scale, and a diminished global warming impact related to water supply operations, but can also lead to some toxic impacts (Crettaz et al. 1999).

These relatively early studies begin to reveal the potential for improving energy efficiency and environmental performance of water supply infrastructure.

Methodology for Energy Use and GHG Emissions Inventory

The conceptual framework of this analysis draws on a study undertaken by MacLean et al. (private communication). Focusing on the environmental accountability of water systems, the objective of the study is to evaluate the environmental impact of potable water production for a large North American city by tracing major energy flows, as in a LCA. GHG emissions, based on a quantitative estimate of energy requirements, serve as the measure of environmental burden. This assessment aims at identifying opportunities to curtail energy use and associated GHG emissions.

System boundaries are drawn so that only the use phase of the water treatment facility (WTF) is analyzed (construction and end-of-life stages are omitted). During the facility operation stage, examined processes include the production and transportation of chemicals and treatment plant operation. The impacts associated with maintenance and repair activities have not been included in this analysis due to the lack of data. Fig. 1 outlines these processes and depicts the analytical boundaries. The reasons for choosing these boundaries are threefold. First, both energy use and GHG emissions during WTF construction and demolition can be considered impulse consumption/emissions (occurring over a short period relative to the total lifetime of the facility), whereas

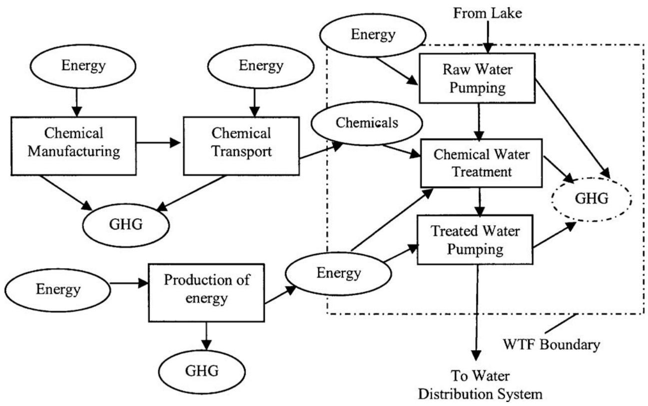

Fig. 1. Life-cycle energy and GHG flow diagram. Dashed boundaries represent processes/products not included in this analysis.

the service provided by a WTF (i.e., filtration of raw water, chemical addition and pumping) occurs throughout the facility's operational life (MacLean et al., private communication). Consequently, there are fewer opportunities to reduce energy use/GHGs from the construction/demolition stages relative to those from the use stage. Second, as construction/demolition energy use/GHGs are weakly related to the quality of water treatment, alternative treatment technologies or abatement strategies can be implemented with little or no construction modifications. When analyzing the use stage of a WTF, the construction-related impacts remain fixed unless significant site alterations are introduced by a dramatic process change (Bagley 2000). Third, the WTF operational phase is the most energy and material intensive as indicated in various studies (Friedrich 2002; Lundie and Beavis 2002; Lundin et al. 2000; Tarantini and Ferri 2001). Methods for calculating total energy use and GHG emissions for each process are developed later.

In order to consistently compare the examined processes, the estimated flow of energy or GHGs is normalized to the functional unit, which is the annual production of drinking water, expressed in cubic meters, and selected in accordance with the main purpose of the WTS.

Total energy use encompasses all the fuel and electricity consumption associated with the examined processes and is expressed in terrajoules/year (TJ/year). Specific GHG emissions include carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). All such emissions are represented by their corresponding Global Warming Potential (GWP), expressed in metric tonnes or kilograms (kg) of CO_2 equivalents (eq.) per year. Converting the GHG flows into equivalent CO_2 emissions is done in accordance with the guidelines issued by the Intergovernmental Panel on Climate Change (IPCC 1997).

The GHG inventory allocates emissions to all selected processes within the system control volume. Only GHG emissions arising from the off-site energy production associated with the above-mentioned processes (i.e., chemical manufacturing, chemical transportation, WTF operation) are considered. Due to the lack of data, the indirect emissions due to the off-site production and the transmission of fuels (i.e., natural gas and diesel) that are used on-site for heat and electricity generation (i.e., for emergency supply) are not included, nor are the direct GHG emissions from the on-site combustion of these fuels. However, these emissions are expected to be negligible. Further, the direct emissions generated from the water treatment processes can be ignored because the chemicals used are inorganic and most of the technologies employed in a conventional WTF are physical, resulting in a trivial contribution to the GHG inventory.

Chemical Manufacturing

The total energy use incurred in treatment chemical manufacturing, $E_{\mathrm{CHEM\_MGF}}$ (in TJ/year) is derived from an economic input-output life-cycle assessment (EIO-LCA) model (CMUGDI 2004). This model facilitates an economy-wide accounting of monetary flows associated with a given transaction [currently utilizing the 1997 input-output (I/O) tables of the U.S. Department of Commerce] and can be coupled with industry sector estimates of unit energy consumption in order to calculate the total energy consumption resulting from a single economic transaction. The EIO-LCA model traces and estimates the monetary flows between all direct and indirect suppliers involved in the manufacturing of a product/service (i.e., chemicals in this case), linking the economic output with environmental data in order to assess the impacts (i.e., energy use, GHG emissions) of this production throughout the entire economy. A comprehensive overview of EIO-LCA methodology along with its advantages/disadvantages is provided by Hendrickson et al. (1998). Notwithstanding EIO-LCA approach limitations (discussed later), its greatest strength is that it reflects upstream environmental impacts throughout all sectors of the economy.

To analyze the impact of chemical manufacturing, the relevant industry sectors are selected for each product in accordance with the 1997 North American Industry Classification System (NAICS), using U.S. Census Bureau (2004) tables. The 2002 purchase prices in local currency used in the study are first discounted by 20%, accounting for a typical average freight and wholesaling cost (Norman et al. 2004). Next, producer prices are deflated to 1997 values by employing Industry Producer Price Indexes (IPPI) (Statistics Canada 2004a). These are then converted from local currency into 1997 U.S. dollars (the parameter of exchange in the EIO-LCA model) using Purchasing Power Parities (OECD 2004). Last, the total energy use/GHG emissions are obtained by running the EIO-LCA model (available on-line at *www.eiolca.net*) for each chemical, with producer prices (in 1997 U.S. dollars) as input. As the EIO-LCA model is not well suited for assessing transportation or WTF operation environmental impacts, alternative estimation methodologies, outlined next, have been chosen.

Chemical Transportation

The most common transportation modes for chemicals include rail and trucks. However, because truck delivery is preferred due to its simplicity, maneuverability, predictability, and wide range of load-size options (Kawamura 2000), it is assumed to be the exclusive transportation mode for this study.

The total energy use, $E_{\mathrm{CHEM\_TR}}$ (TJ/year), associated with chemical transportation is calculated as the sum of combustion (end-use) energy, E_C, and upstream (process) energy, E_U. The combustion energy is given by: $E_C = e_c DM$, where e_c=combustion energy intensity for bulk transportation, in megajoules per ton km (MJ/t km); D=average distance from the chemical manufacturing facility to the treatment plant (km); M=total mass of chemicals delivered (ton). Data for e_c is taken from the literature and government data sets. The upstream energy is expressed as: $E_U = e_u E_C$, where e_u=upstream energy-use intensity, calculated as MJ of upstream energy consumed per MJ of fuel delivered (combustion energy). Data for energy intensity

e_u is taken from a greenhouse-gas emissions model (GHGenius), for the base year 2000 (NRCan 2003). No allowance is made for the possible increase in GHG emissions due to better fuel economies in 2000 relative to 1997.

The GHGenius model was developed to analyze the full-cycle energy use and emissions of transportation systems. The full-cycle concept combines in a single analysis fuel production, vehicle manufacturing, and fuel use. However, for this analysis, the vehicle cycle includes only the vehicle operation phase as it is the dominant contributor. The GHG emissions/energy use associated with vehicle manufacturing, including emissions from indirect energy use (i.e., energy required to manufacture, repair, and maintain the infrastructure) accounts only for 10% of overall full cycle emissions (NRCan 2003). The GHG emissions are traced from production of the energy source, through to fuel processing, distribution, and, finally, to combustion in a motor vehicle. The upstream energy accounts for the exploitation of the raw energy source, transportation of fuel to a refinery/production plant and then to a retail site, and fuel distribution, storage, and dispensing. The combustion energy considers only the fuel used during the vehicle operation stage $[(\mathrm{S\&T})^2$ Consultants 2000$]$. GHGenius has the capability to perform regional analysis for Canada, the United States, and Mexico, as well as the selection of engine-fuel combination, and feedstock type.

Full cycle emissions are calculated by multiplying the total energy use, $E_{\mathrm{CHEM\_TR}}$ (in kW h), with GHG emission factors, expressed in grams of CO_2 equivalents per kW h of electricity delivered (g CO_2 eq./kW h).

WTF Operation

To estimate the electricity consumption during facility operation (operational electricity), technical reports for the four WTFs were reviewed to ascertain relevant design/operating information about the dominant processes in a conventional treatment plant. The estimation method draws on the EPA's technical report, "Estimating Water Treatment Cost" (Gummerman et al. 1979), which assesses fabrication and process energy for the main component systems. The plant is assumed to be operating at 70% of full capacity (Gummerman et al. 1979). The energy use associated with the production and usage of maintenance materials is neglected.

Subsequently, the amount of electricity, expressed in MW h/Year, is multiplied by the energy intensity (in TJ per kW h of electricity generated), in order to estimate the total energy use, E_{OPER} (TJ/year). This conversion permits a comprehensive assessment of the energy used to generate electricity and the associated GHG emissions from electricity production in terms of upstream and combustion components, ensuring consistency with foregoing methods. Upstream components include the resource use/GHG emissions from mining, refining, and transporting fuels to their consumption points, whereas the combustion components represent the quantity of fuel consumed and the emissions arising from such consumption.

The GHG emissions from electricity generation required for WTF operation are calculated by multiplying E_{OPER} with GHG intensity, expressed in tonnes of CO_2 equivalents per terrajoules of electricity generated (tonnes CO_2 eq./TJ).

Case Study: City of Toronto Water Treatment System

The City of Toronto WTS includes four filtration plants: Harris, Clark, Horgan, and Island, supplying all Toronto and York Region residents and businesses. The WTFs draw water, generally of good physical and chemical quality, from Lake Ontario. The Harris and Clark plants employ conventional treatment, consisting of chemical coagulation, flocculation, sedimentation, and dual media filtration. The chemical treatment processes are: Coagulation with alum, disinfection with gaseous chlorine (applied in pre- and postchlorination modes), dechlorination with sulfur dioxide gas (to remove excess chlorine), fluoridation with hydrofluosilicic acid and ammoniation with aqueous-ammonia. The Horgan and Island plants utilize direct filtration for treatment, a process similar to conventional treatment except that sedimentation is not included (MacLaren Engineers Inc. 1990a; b; Gore & Storrie Ltd. 1990). The filtered and treated water is then pumped into the distribution system. Sedimentation tank sludge and filter backwash water are discharged to the lake, except from the Horgan plant where backwash water is treated for solids separation, with sludge pumped to Highland Creek Pollution Control Plant for further treatment.

Chemical Manufacturing

Information for the 2002 chemical use regime (i.e., annual consumption, purchase prices, suppliers) for water treatment was provided by the City of Toronto, Works and Emergency Services Department. The chemicals considered in this study are: Alum and polyaluminum chloride (coagulation); cationic polymer and sodium acetate (coagulation additives/polymers); chlorine gas and sodium hypoclorite (chlorination); hydrofluosilicic acid (fluoridation); aqueous-ammonia; sodium metabisulfite and sulphur dioxide (dechlorination). The Industry Producer Price Indexes necessary to calculate producer prices for chemicals, in 1997 Canadian dollars, were taken from Statistics Canada (2004a). The Manufacturing Code #325188 and the corresponding industry "Other Basic Inorganic Chemical Manufacturing" were selected, as per 1997 NAICS, for all analyzed chemicals, except for ammonia, which corresponds to "Nitrogeneous Fertilizer Manufacturing," Code #325311. The total energy use/GHG emissions for each chemical were determined by running EIO-LCA. The output values normalized to functional unit were calculated using the total capacity of Toronto's WTS for 1997 (City of Toronto 2004).

Chemical Transportation

The combustion and upstream energy use were calculated for the transportation of each chemical. A combustion energy intensity, e_c, of 2.68 MJ/t km was selected for 1997 Ontario from publicly available data in the "Energy Use Data Handbook, 1990 and 1996 to 2002" (NRCan 2004) for heavy-duty diesel vehicles (HDDV). This rate falls in the range of values reported in the literature: 1.23 MJ/t km (in GHGenius), representative for Canada (year 2000), and 0.27 to 3.25 MJ/t km for Canadian trucks reported by Nix (1991). Different energy intensity values reflect temporal and geographical variations. The distances (in kilometers) from manufacturers to WTFs were estimated based on production facilities located in Ontario when applicable (the hydrofluosilicic acid is delivered from Florida). Where suppliers had multiple locations, the distance was averaged. Chemical requirements for 2002 were used in the calculations. The implications arising from temporal variations in chemical usage are discussed later in the paper.

The upstream energy-use intensity, e_u, of 0.22 kW h of energy consumed per kW h of energy delivered, representative for Canada in 2000, was taken from the GHGenius database (NRCan 2003) for HDDV using diesel fuel from crude oil feedstock. The

Table 1. 1997 Electricity Mix for Canada and Ontario (Adapted from NRCan 2004)

Source	Percent of total generation ON	CAN
Coal	17.9	16.7
Gas	6.0	3.6
Oil	0.4	2.0
Nuclear	47.7	14.0
Hydro	27.2	62.5
Other	0.8	1.2

Table 3. 1997 Total and Operational Capacities

Plant	Total annual capacity (ML[a])	Operational capacity (MLD[b])
R. C. Harris	238,196	680
R. L. Clark	146,703	402
F. J. Horgan	129,787	376
Island	6,611	132

[a]ML=million liters.

[b]MLD=million liters per day.

emission factors (Sabrina Spatari, private communication, May 10, 2004), calculated for Ontario's electricity mix profile for 1997 (Table 1), are given in Table 2. Because GHGenius emissions are based on the USEPA AP-42 Standard (1995), a similar emissions structure from electricity sources was assumed for both Canada and the United States. The 1997 electricity generation mix was determined using data from NRCan (2004).

WTF Operation

The main systems in a conventional WTF, used to calculate the operational electricity requirements, are: Raw and treated water pumping; alum, polymer, ammonia, and chlorine feed systems; flocculation; treatment of backwash water; gravity filtration (including surface wash); backwash water pumping; mixing; administrative and laboratory activities, and building maintenance. The raw water pumping refers to the headworks and the treated water pumping includes exclusively the pumping (at the treatment plant) of the final effluent into the distribution system. All water treatment plants pump water in the same distribution system and are used interchangeably. Moreover, the differences in topologies of different pressure areas in the City of Toronto's distribution system are not significant. The analysis is based on data taken from Water Plant Optimization Study reports (MacLaren Engineers Inc. 1990a; b; Gore & Storrie Ltd. 1990) conducted for three of Toronto's WTFs (Harris, Clark, and Horgan). The operational electricity use includes process and building requirements. For the latter, an average building-related demand of $0.11\ \mathrm{kwh/cm^2}$ year was assumed (Gummerman 1979). When using process and building electricity curves, design, and operational flow/parameters, respectively, were employed. The 1997 total and operating capacities of each plant, given in Table 3, were taken from publicly available data (City of Toronto 2004).

Total electricity consumption, expressed in MWh /year with normalized values on a per m^3 of treated water basis (specific electricity use, in $\mathrm{kW\,h/m^3}$), were calculated for each plant. Due to the lack of data for the Island plant, the associated electricity requirements were extrapolated by multiplying the specific electricity use for Horgan (also using direct filtration) with its average production for 1997.

To estimate total energy use (TJ/year), the total amount of electricity is multiplied by the Ontario energy intensity of $8.86\ \mathrm{MJ/kW\,h}$ generated (NRCan 2004). Subsequently, the total energy use is multiplied by the GHG intensity of 48.1 tonnes CO_2 eq./TJ generated to calculate total GHG emissions (in tonnes CO_2 eq./year). Energy/GHG intensities for 1997 Ontario were taken from Energy Use Data Handbook (NRCan 2004).

Results and Discussion

Chemical Manufacturing

Owing to the insufficient sectoral disaggregation of the EIO-LCA model, the chemical manufacturing results do not reflect the relative contribution of different chemicals, but the total upstream impacts associated with the fabrication of chemicals. This limitation is further discussed later.

The curtailment of production-related energy use can be achieved through the efficient use of chemicals. Potential solutions are the alternative disinfection technologies, such as membrane filtration, ozonation, and UV filtration. However, some of these practices are more energy-intensive, trading off the embodied energy in treatment chemicals for greater energy use during operation. Friedrich (2002) noted that the energy consumption for conventional (using chlorination) and membrane water treatment methods are comparable, with the latter being slightly higher. When selecting and promoting a new disinfection technology, such as membrane filtration, thoughtful consideration of all environmental impacts (e.g., generated waste, replacement cost, energy use) is required. One suggested improvement to conventional treatment processes is the full automation of dosage systems (van der Helm and Rietveld 2002). Presently, the chemical feeders for Toronto's WTFs are operated in a semiautomated

Table 2. 1997 GHG Emissions per kW h of Energy Delivered by Source, in Ontario (Adapted from Personal Communication with Sabrina Spatari, May 10, 2004)

GHG emissions (g/kW h)	Coal	Natural gas	Oil	Nuclear	Hydro	Other (biomass)	Ontario's weighted average
CO_2	1,030	391	791	14	16	0	222
CH_4 (g CO_2 eq.)	0.2277	1.8561	0.207	0	7.5	0.0092	0
N_2O (g CO_2 eq.)	0.1184	4.3808	0.3552	0	0	0.0592	2
Total GHG (g CO_2 eq.)	1,031	397	792	14	24	0.0684	224

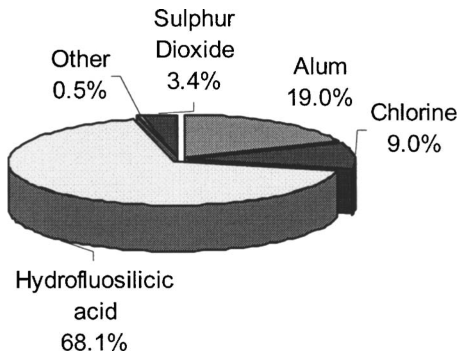

Fig. 2. Chemical transportation total energy use/GHG emissions

mode through a process controller, the feed rates being manually adjusted (MacLaren Engineers Inc. 1990a; b; Gore & Storrie Ltd. 1990).

In light of these findings, water demand management strategies clearly emerge as a strategic way to reduce the amount of chemicals used in treatment and can thus serve to limit the energy use and GHGs associated with water treatment.

Chemical Transportation

The four chemicals accounting for most of the transportation energy use and GHG emissions are shown in Fig. 2. The hydrofluosilicic acid shows the highest energy use/GHGs, its relative contribution being approximately 68%. Following, in a descending order, are alum accounting for 19%, chlorine 9%, and sulfur dioxide 3%. The remaining chemicals pose an insignificant contribution of less than 1%.

The highest transportation impacts of hydrofluosilicic acid are due to manufacturer location (Florida) which is the farthest from the WTFs. In the case of the other chemicals, their contribution to overall energy use/GHG emissions is primarily derived from the quantities consumed in the treatment process. Finding a closer supplier for hydrofluosilicic acid, or using alternative chemicals produced locally, is essential for reducing overall transportation impacts. Additionally, a more efficient use of water leads to chemical and consequently energy savings, minimizing related GHG emissions.

WTF Operation

The relative contribution of dominant WTF systems to total operational energy use/GHG emissions for the Harris plant is illustrated in Fig. 3. The facility has the highest total operational energy use (603 TJ/year) and GHG emissions (29,000 tonnes CO_2 eq.). However, based on this study's results, no direct correlation can be established between operational impacts and plant capacity. An example in this sense is Horgan plant, which despite its smallest water production, has a higher energy use, of 2.4 MJ/m^3, compared with 2.3 MJ/m^3 for Clark. This indicates that energy use is influenced not only by plant capacity but also by other criteria, such as system type (e.g., flash in-line mixers, rapid mixing), plant topology (affecting the pumping head), gradient velocity (for flocculation and mixing systems), and the existence of backwash water treatment (e.g., for Horgan).

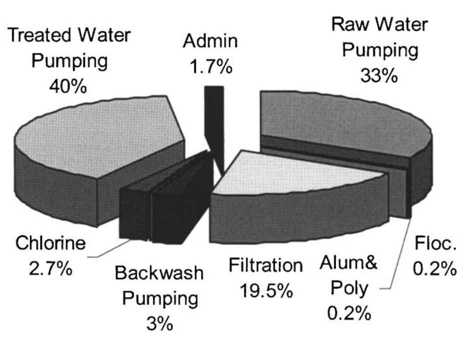

Fig. 3. R.C. Harris—total operational energy use

The analysis of WTF operation reveals that on-site raw and treated water pumping, and filtration systems (including backwash pumping), in all three plants (Harris, Clark, and Horgan) are responsible for 93–95% of the total electricity/energy used in 1997. The treated water pumping energy alone can be as high as 64% (Horgan), the different values obtained for the other facilities (Harris—40%; Clark—59%) being attributed to plant capacity and topology. A similar value (65%) for the relative contribution of on-site treated water pumping energy to total primary energy use was found by Tarantini and Ferri (2001) for Bologna's WTS. The determination that pumping (of raw and treated water) is the most energy-intensive process is also consistent with other findings (Friedrich 2002; Rihon et al. 2002; Ilomaki et al. 2003). Further, the average electricity consumption rate of 0.28 $kW\,h/m^3$ is of similar magnitude to values reported in the literature: 0.217 $kW\,h/m^3$ for a conventional WTS in Taiwan (Cheng 2002); 0.17 $kW\,h/m^3$ for Goteborg's (Sweden) WTS (Lundin et al. 1999). The slight disparity could be attributed to a number of factors: System topology (distance or elevation of serviced area), WTS structure (different disinfection technologies not distinctly specified in all cited studies), raw water quality, and system boundaries (e.g., treated water pumping not included in the Swedish study), among others.

In general, electricity use is less for water treatment than for wastewater processing or water distribution. In Toronto, the specific operational electricity use in the WTF is approximately half the average electricity consumption in a wastewater treatment facility, 0.47 $kW\,h/m^3$, and between 30 and 50% of specific pumping electricity used in the water distribution system, 0.5–0.7 $kW\,h/m^3$ (Sahely et al. 2003).

Because capacity/demand is the major determinant of pumping electricity/energy use, water conservation programs and strategies can contribute significantly to reducing the operational energy consumption of WTFs, thereby minimizing associated GHG emissions.

Facility Use Phase

The total and normalized energy expenditures/GHG emissions for each phase are given in Table 4. The cumulative impacts of these activities (chemical manufacturing and transportation, WTF operation) are shown in Fig. 4. It is apparent that WTF operation is the major contributor to overall energy use and GHG emissions, accounting for 94% of total energy use and 90% of GHGs, dwarfing the impacts of the other two processes. Chemical manufacturing is responsible for 5 and 7% of total energy use and GHG emissions, respectively. The relative contribution of transportation is small when compared with operational impacts. These results are consistent with those reported for Bologna's WTS (Tarantini and Ferri 2001): 10% for chemical manufacturing with a minimal contribution from transportation.

The normalized energy use was found to lie between 2.3 and

Table 4. Total Energy Use and GHG Emissions

Process	Total energy use (TJ/year)	GHG emissions (tonnes CO_2 eq./year)	Specific energy use (MJ/m^3 year)	Specific gHG emissions (g CO_2 eq./m^3 year)
Chemical manufacturing	71	4,622	0.14	8.87
Chemical transportation	16	1,018	0.03	1.95
WTF operation	1,271	61,156	2.44	117.31
Total	**1,359**	**66,796**	**2.61**	**128.13**

2.5 MJ/m^3, which is close to the value of 2.067 MJ/kL reported by Friedrich (2002). The slight difference could be due to differences in estimation methodology (Friedrich uses conventional LCA), different conventional treatment systems, system topology, raw water quality, and the electricity generation mix. On a larger scale, the WTS specific energy consumption on a per capita basis, calculated based on the population served in 1997 (Statistics Canada 2004b) is 488 MJ/ca-year. For comparison, this amount represents only 2% of dwelling operational energy use (27,500 MJ/capita-year), in a high-density development, in Toronto (Norman et al. 2004).

It is interesting that both transportation and manufacturing contribute more to GHG emissions than to energy use. The explanation for this is related to the fuel mix for the individual activities. The use of high carbon content fossil fuels like diesel in HDDV for bulk transportation is more GHG intensive, a trend noted in the literature (NRCan 2004; Norman et al. 2004). In contrast, the fuel mix for the electricity generation that powers WTF operation includes less GHG-intensive energy sources such as nuclear and hydropower (even though these involve other ecological burdens). Further, the manufacturing impacts can be attributed to direct emissions from production processes (i.e., the embodied energy in materials) as opposed to much smaller emissions realized during WTF operation.

Model Relevance, Limitations, and Uncertainties

The basic EIO-LCA model is suitable for estimating the energy use/GHG emissions arising from the fabrication of a product (i.e., chemicals), but does not explicitly include the use stage of a product or service (i.e., WTF operation, chemical transportation) (Hendrickson et al. 1998). Consequently, the impacts of these two processes were estimated using different methods, based on data compiled from public databases, government reports and publications, and the engineering literature. Some of the strengths, limitations, and uncertainties inherent with the application of these methodologies are briefly outlined next.

EIO-LCA approximates the connections among the chemicals for water treatment with their corresponding industrial sectors, not reflecting correctly the specifics of each product. One manufacturing industry representing almost all chemicals attests to this insufficient level of sectoral disaggregation. Nonetheless, the economy-wide scope of the EIO-LCA model advantageously applied for chemical manufacturing process (Hendrickson et al. 1998) provides a comprehensive assessment, the results of which are suitably combined with those from chemical transportation and WTF operation models, given their comparable boundaries.

The EIO approach is not geographically sensitive, which is another limitation of the study. Due to the high aggregation level of the Canadian EIO-LCA model, the U.S. economic input-output model is used for a Canadian metropolis (Toronto), based on the hypothesis that both countries have identical economic and industrial structures. When comparing the results of the two models, Bjorn and MacLean (2003) observed that Canadian sectors exhibited lower GHG emissions and higher energy/electricity use, owing to different electricity generation profiles, economic and export structures, and climate conditions. However, the study indicated that most sectors yielded similar output. These results were expected given the emerging North American economic integration and the growing interest in improving Canadian economic growth and productivity (Harris 2001). Along the same vein, efficient fuel consumption and GHG emissions from turbines and industrial boilers, used in the electricity generation sector, were assumed to be similar for Canada and the United States. This assumption was made when estimating transportation and operational energy use/GHG emissions based on US EPA standards (EPA AP-42) and technical reports (Gummerman 1979).

Due to the lack of data for 1997 chemical usage, the consumption for 2002 was used in modeling the manufacturing and transportation impacts. Judging by the 10% increase in the WTFs average production between 1997 and 2002 (City of Toronto 2002b), chemical use, in general, is expected to grow. Nonetheless, the chemical consumption, also influenced by raw water quality, could as well exhibit a decreasing trend. An increase between 7 and 11% in chlorine use over the period 1999–2001 was calculated based on data from Sahely et al. (2003). The impact of chemical usage uncertainty on the study results is assessed through a sensitivity analysis. The amount of chemicals is varied by ±10%, in graduated 5% increments. The analysis indicates that manufacturing and transportation energy use are slightly sensitive to moderate changes in the amount of chemicals used for treating water. However, variation in this parameter produces no change in the overall WTS results (represented as "Total" in Table 5). This is owing to the small contribution of chemicals manufacturing and transportation to overall energy use/GHG emissions.

Data from the GHGenius model were employed in modeling transportation-related energy use/GHG. The GHGenius database has the advantage that the model boundaries are consistent with the EIO-LCA model and thus allow for a realistic evaluation of transportation energy use/GHG emissions relative to chemical manufacturing or WTF operation. Moreover, the model is regionally sensitive, having the capability to estimate energy and emis-

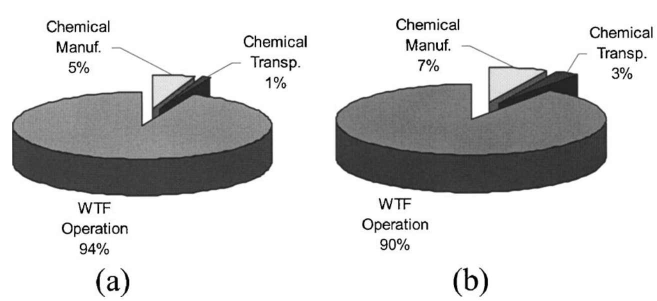

Fig. 4. Total annual impacts: (a) energy use; (b) GHG emissions

Table 5. Sensitivity Analysis

Qty. chemicals	−10%	−5%	0%	5%	10%
Energy use (TJ/year)					
Manufacturing	67.60	71.36	71.03	78.87	82.62
Transportation	14.71	15.53	16.35	17.17	17.98
Operation	1,271	1,271	1,271	1,271	1,271
Total	**1,354**	**1,358**	**1,359**	**1,367**	**1,372**

sions for the specific location of a WTF. However, this estimation approach is somewhat limited in scope and introduces some uncertainties. In quantifying the transportation-related GHGs, emission factors for 2000 were employed. It could be argued that transportation GHGs were lower in 2000 relative to 1997 (the time frame used for manufacturing and operational estimates) due to improved feedstock properties, and consequently lower combustion and upstream energy intensity values. Additionally, these emission factors also depend on the regional (Ontario's) fuel mix that shifted toward "dirtier" primary energy sources with higher carbon content (the share of coal in the fuel mix rose to offset the reduction of nuclear/hydro generation) (NRCan 2004), a change that increased the 2000 GHG emissions relative to 1997 values. Another uncertainty is the hypothesis that diesel trucks are the transportation mode for all chemicals. However, the transportation contribution to overall energy use/GHG emissions is only 1 and 3%, respectively, and, therefore, the temporal variations/inconsistencies of the above-mentioned parameters are not expected to alter the final results significantly.

Indirect emissions from on-site heating and electricity generation for any WTF operation were not included in this study. The inclusion of these emissions would have increased the operational estimates, which in light of the study's goal, leaves the overall results for the facility use phase unchanged. Nevertheless, the estimation of indirect GHGs could be useful when analyzing their relative contribution.

Conclusions

LCI has been used to estimate the total energy use and GHG emissions for the facility use phase of a WTS in a major North American city. The choice of a life-cycle perspective was motivated by an attempt to achieve a holistic estimate of energy and GHG flows. Additionally, this approach assisted with identifying abatement strategies with respect to these two environmental indicators.

Of the three examined processes involved in the production of drinking water for the City of Toronto case study, WTF operation was found to be the dominant contributor to overall energy use and GHG emissions, whereas the environmental impacts from chemicals transportation were deemed small. Further water consumption was identified in all processes as an important factor. Pumping of raw and treated water, directly correlated with water demand, is the most energy- and GHG-intensive process. Increased water use is also reflected in growing chemical requirements. By reducing water consumption, the overall energy burden of a WTS can be improved. Toronto, for instance, has already begun to implement a more sustainable management/operation scheme for its water system by advocating the implementation of its Water Efficiency Plan (City of Toronto 2002a).

Other steps toward better energy use in water supply infrastructure could include the full automation and optimization of WTF systems, increased pumping efficiency, reduced energy waste from leaky distribution systems (Colombo and Karney 2002), and electricity consumption monitoring. Additionally, minimizing chemicals use, while meeting human health requirements, would lessen the total impact (total energy use/GHGs) and result in an enhanced environmental performance of the system.

While this LCI helped to identify the most energy and GHG intensive processes during the operation phase, it was somewhat limited by a hybrid methodology. The assessment model could be further improved as more geographically and temporally sensitive data/models, covering the information gaps encountered in this study, become available. Notwithstanding these disadvantages, the analysis possesses certain merits, such as a reduction in time and resource requirements. By incorporating such an analysis in system planning and the assessment of new technologies, the study can serve to develop a more preventive-oriented design and decision-making framework (Vanderburg 2000). Expanding the boundaries of this life-cycle-s based assessment to the complex urban water infrastructure (including drinking water distribution and waste water systems) could cast a helpful light on future planning questions. By highlighting the relative importance of upstream and downstream energy use to total energy expenditures, the study provides useful insights into the consideration of alternative inputs (cleaner energy/fuel sources, chemicals) and the selection of supply chains. The results of this analysis can be subsumed in more detailed studies to identify energy use and GHG emissions abatement strategies and can play a role in comparative studies assessing the environmental sustainability of innovative technologies for water conservation, such as rainwater harvesting and/or graywater reuse, or in an overall municipal energy and mass flow analysis.

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