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The environmental comparison of landfilling vs. incineration of MSW accounting for waste diversion

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ABSTRACT

This study evaluates the environmental performance and discounted costs of the incineration and landfilling of municipal solid waste that is ready for the final disposal while accounting for existing waste diversion initiatives, using the life cycle assessment (LCA) methodology. Parameters such as changing waste generation quantities, diversion rates and waste composition were also considered. Two scenarios were assessed in this study on how to treat the waste that remains after diversion. The first scenario is the status quo, where the entire residual waste was landfilled whereas in the second scenario approximately 50% of the residual waste was incinerated while the remainder is landfilled. Electricity was produced in each scenario. Data from the City of Toronto was used to undertake this study. Results showed that the waste diversion initiatives were more effective in reducing the organic portion of the waste, in turn, reducing the net electricity production of the landfill while increasing the net electricity production of the incinerator. Therefore, the scenario that incorporated incineration performed better environmentally and contributed overall to a significant reduction in greenhouse gas emissions because of the displacement of power plant emissions; however, at a noticeably higher cost. Although landfilling proves to be the better financial option, it is for the shorter term. The landfill option would require the need of a replacement landfill much sooner. The financial and environmental effects of this expenditure have yet to be considered.

1. Introduction

The disposal of municipal solid waste (MSW) is one of the more serious and controversial urban issues facing local governments globally. Increasing waste generation due to population growth, societal lifestyle changes, development and consumption of products that are less biodegradable, have led to the diverse challenges for MSW management in various cities around the world (Asase et al., 2009). Over the past few decades, governments and citizens have become especially aware and concerned about how wastes are managed (Statistics Canada, 2005).

In Canada, the availability of non-developed land has made land disposal or landfilling, the most popular and cheapest method of waste disposal (Ministry of Environment, 2004). However, with 30% of landfills in Canada expected to be full by 2010, recycling is viewed as a preferred method of reducing the amount of waste going to landfills while biological treatment of waste such as composting is becoming more widespread (Statistics Canada, 2005). In the past, the presence of appropriate landfill sites close to major urban centres has limited the development of incineration facilities in Canada. Furthermore, thermal treatment of waste has received strong local opposition due to beliefs that incinerating: threatens human health and the environment; and is incompatible with the concept of reducing, reusing, and recycling (Sawell et al., 1996). Although incineration is not very popular in Canada, there are currently seven municipal solid waste (MSW) thermal treatment facilities operating that have a capacity greater than 25 tonnes per day (tpd); in 2006, these thermal treatment facilities handled approximately 3% of Canada's MSW. There have been no thermal treatment facilities constructed in Canada since 1995, with the exception of demonstration facilities in Ontario and Quebec (Environment Canada, 2007).

Various municipalities view the basic management options: (1) waste prevention (2) recycling (3) biological treatment (4) thermal treatment (5) landfilling, as a hierarchical and not an integrated waste management system (Tchobanoglous et al., 2002). However, the idea behind integrated solid waste management (ISWM) is that, rather than accepting a simple hierarchy, alternatives should be examined systematically so that waste is managed in the most resourceful and environmentally friendly manner (Clift et al., 2000).

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Thermal treatment is currently a management option that is being dismissed as a possible method for treating waste that has been already reduced through waste prevention, recycling and biological treatment. In making use of the ISWM concept, this study assesses the environmental implications of implementing waste incineration to reduce the amount of waste being landfilled in an existing Canadian waste management system that currently has waste reduction and diversion measures in place. A life cycle assessment (LCA) was used to carry out this study. In addition to an environmental study, a generic discounted cost analysis was done in order to compare the cost of the waste management technologies.

Few studies such as Rigamonti et al. (2009), Emery et al. (2007) and Cherubini et al. (2009) have incorporated a method of accounting for different waste compositions. Rigamonti et al. (2009) evaluate possible optimum levels of source-separated collection that lead to the most favourable energetic and environmental results. Emery et al. (2007) examined the environmental and economic impacts of a number of waste disposal systems used in a typical South Wales valley location. Four options were analyzed using one constant MSW composition; however waste arisings assuming a 3% per year increase was included. Cherubini et al. (2009) who evaluates emissions, total material demands, total energy requirements and ecological footprints of four waste management scenarios, included a scenario that splits the inorganic waste fraction (used to produce electricity via Refuse Derived Fuels, RDF) from the organic waste fraction (used to produce biogas via anaerobic digestion);

Several other studies, such as, Zhao et al. (2009), Liamsanguan and Gheewala (2008), Moberg et al. (2005) use one constant waste composition to undertake the LCA. Similarly to Rigamonti et al. (2009), the source-separated collection level is parameter in the analysis, along with the increase in waste generation included in Emery et al. (2007). This study focuses on how the current waste diversion initiatives and the goal of increasing the diversion rate affect waste management methods that treat residual waste.

2. Methodology

2.1. Life cycle assessment

An LCA is a useful tool to evaluate the performance of MSW management systems (Ekvall et al., 2007; Liamsanguan and Gheewala, 2008). The international standard ISO 14040-43 defines LCA as a compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle (Arena et al., 2003). Life-cycle assessments were initially developed for the purpose of analysing products, although recently, it has also been applied to the treatment of waste. The use of LCA for resources and waste management issues implies a slightly different focus than traditional product-oriented LCAs (Obersteiner et al., 2007). The popularity of LCAs in analyzing MSW management systems is illustrated by the numerous published studies of the life cycle emissions of these systems, as well as by the substantial number of LCA computer models addressing MSW management (Cleary, 2009).

The structure of a LCA consists of four distinct phases, which contribute to an integrated approach (Arena et al., 2003):

(1) *Goal and scope definition*, which serves to define the purpose and extent of the study, to indicate the intended audience and to describe the system studied as well as the options that will be compared.
(2) *Inventory analysis or life cycle inventory (LCI)* focuses on the quantification of mass and energy fluxes.
(3) *Impact assessment or LCIA*, which aims at understanding and evaluating the magnitude and significance of potential environmental impacts of a system (Clift et al., 2000).
The LCIA organises the LCI inputs and outputs into specific, selected impact categories and models the inputs and outputs for each category into an aggregate indicator; such final aggregation is controversial, whereby many authors terminate the assessment without attempting any synthesis of different impact indicators (Consonni et al., 2005).
(4) *Interpretation*, evaluates the results from the previous phases in relation to the goal and scope in order to reach conclusions and recommendations (Finnveden et al., 2009).

Although LCAs allow for a holistic view of the environmental consequences of a process, product or service, it is important to be aware of the limitations of the methodology and to understand that the environmental information it generates is neither complete, nor absolutely objective or accurate. LCA results are dependent on methodological decisions, such as assumptions made in the study and sources of input data that may be influenced by the values and perspectives of the LCA practitioner (Ekvall et al., 2007).

2.2. Goal and scope definition

The objective of this study is to evaluate the environmental performance of the incineration and landfilling of MSW that is ready for the final disposal while accounting for existing waste diversion initiatives, using the LCA methodology. Parameters such as changing waste generation quantities, diversion rates and waste composition are also considered. A generic discounted cost analysis was done in order to compare the cost of the incineration.

2.3. Selected study site

The City of Toronto was used as a selected study site for this life cycle assessment due to its increasing number of waste diversion initiatives; resistance to considering MSW thermal treatment as a potential waste management technology; as well as accessible detailed documentation of its waste diversion initiatives and landfill operations. The City of Toronto diverts waste from the landfill through various programmes, such as, programmes that:

- offer curbside collection of organic materials (i.e. fruit and vegetables scraps, paper towels, coffee grinds, etc.) turns it into compost;
- encourage residents to leave grass clippings on the lawn in order reduce the need for fertilizer and water;
- promote the reuse of glass bottles; and
- enable residents to combine both paper and container recyclables into a single stream.

Simplifying assumptions that were made in this study do not reflect the current or future activities of the City of Toronto nor of the Green Lane Landfill.

2.4. Scope

In this analysis, two different waste management scenarios, with both recovering electricity only, were investigated:

The "status quo" Scenario: The landfilling option. All the residual waste is sent to the landfill without any further treatment.

Scenario 2: The incineration option. 1000 tonnes/day of residual waste will be incinerated while the remainder will be sent to a landfill.

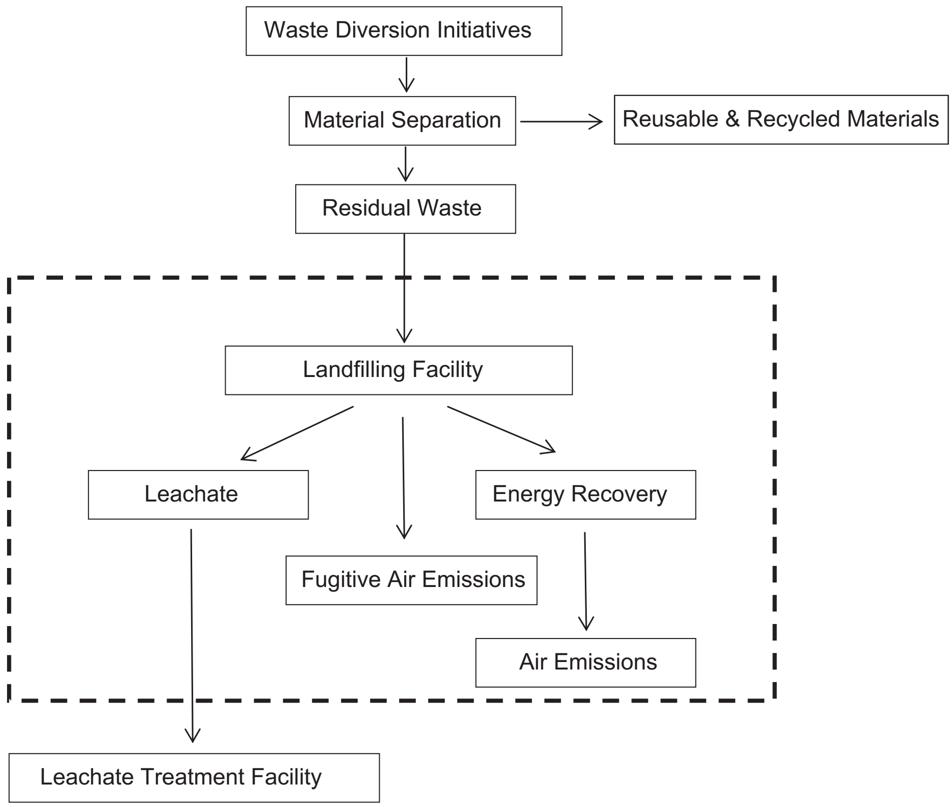

Fig. 1. Scenario 1, the landfilling option with electricity recovery only.

The life cycle of MSW in this study begins after the material recovery processes. Therefore, it is assumed that the waste collection, separation processes, and transfer station operations will be the same for both waste management scenarios and can be omitted from the LCA. The scope of this LCA is on the transportation and the treatment of the waste. The system boundaries for where the LCA applies in each scenario are illustrated in Figs. 1 and 2.

The term "residual waste" was used to define the waste that remains after the diversion of waste through recycling, composting or prevention has occurred. The residual waste is a combination of residential and ICI (Industrial, Commercial and Institutional) waste. The residential waste refers to the waste that is collected by the City of Toronto and that are required to meet certain specifications. The ICI waste refers to the waste that is not collected by the City of Toronto, but that is dropped off at transfer stations by companies for disposal. Construction and demolition debris, and wastewater residuals were not included in this analysis. The diversion initiatives apply only to the residential waste and the ICI waste remains unsorted. Furthermore, any increase or change in the diversion of waste will affect the waste composition for both scenarios in the same manner.

For each scenario, a detailed LCI has been used to determine the environmental emissions. The emissions produced from the construction of facilities are not included in this study. Other studies such as Liamsanguan and Gheewala (2008), Eriksson et al. (2005), and Wanichpongpan and Gheewala (2007) have made similar assumptions by considering these emissions smaller compared to those released during the use of the facility. The environmental effects of auxiliary materials such as supplemental fuels, daily covers[1] and pollution control chemicals were not examined. All of the methods and emissions factors used to develop the LCI are described in the following sections.

The environmental performance and cost of the incineration and landfilling options were analyzed over a period of 30 years, from 2011 to 2040. This study focused on the active life phase of the landfill and did not include the environmental implications of landfill closure and post-closure emissions.

The functional unit of this study is "tonnes of MSW from the City of Toronto between 2011 and 2040". Using an average of previous data, it was estimated that in 2011, approximately 875,000 tonnes of residential waste would be generated while the diversion rate would be 46%. The quantity of industrial residual waste during that same year was estimated to be 202,500 tonnes.

In this study the following emissions were considered:

- emissions from the stack of incineration plants;
- emissions from the transport of solid residues to the waste management facilities;
- emissions from landfill operations;
- avoided emissions from power stations and thermal plants displaced by the WTE plant and landfill;

The following elements were not considered:

- auxiliary fuel requirements;
- emissions related to ash disposal;
- emissions relating to leachate treatment from the landfill;
- emissions relating to the use and transport of daily and final cover for the landfill facility.

Leachate treatment was not included in the scope. The Green Lane landfill operates an on-site wastewater treatment plant for the leachate. In order to reduce the complexity of this initial

[1] Daily cover is the material such as native soil that is applied to the working faces of the landfill at the end of each operating period (O'Leary and Tchobanoglous, 2002).

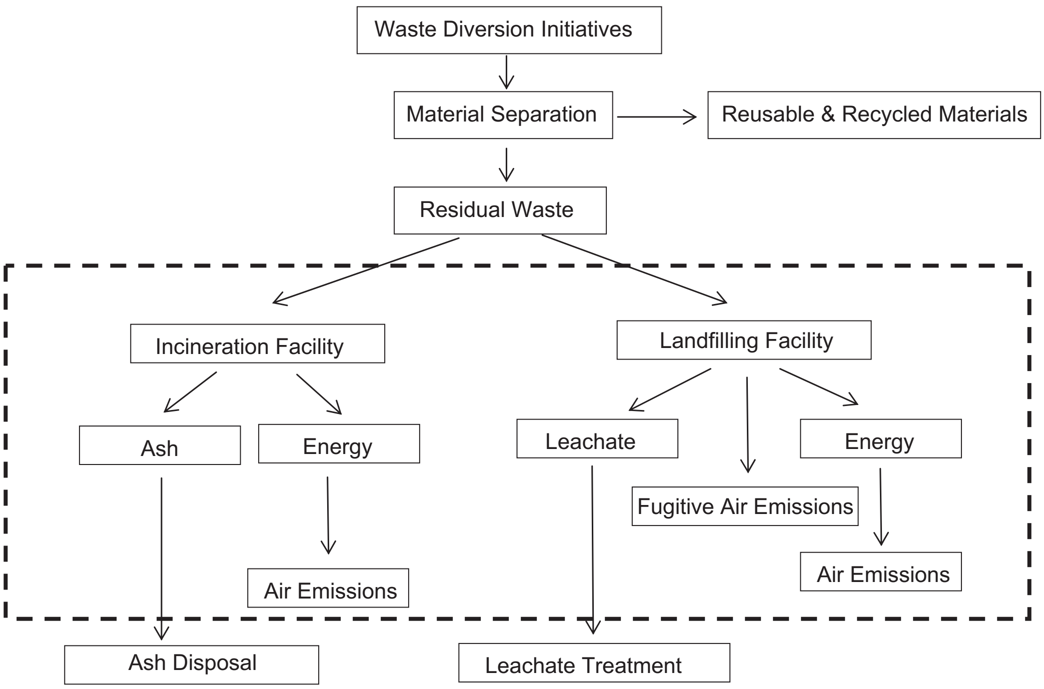

Fig. 2. Scenario 2, the incineration option with electricity recovery only.

Residential

Waste Generated = WG (Tonnes)

Waste Distribution	%	Waste Generated (Tonnes)
Paper	26	WG_{Paper}
Food	27	WG_{Food}
Yard	19	WG_{Yard}
Wood	2	WG_{Wood}
Plastics	8	$WG_{Plastics}$
Textile	1	$WG_{Textile}$
Leather	<1	$WG_{Leather}$
Rubber	<1	WG_{Rubber}
Ferrous	1	$WG_{Ferrous}$
Non-Ferrous	<1	$WG_{Non\text{-}Ferrous}$
Glass	5	WG_{Glass}
Others	10	WG_{Other}

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Waste Diverted = WD(Tonnes)

Waste Distribution	%	Waste Diverted (Tonnes)
Paper	31	WD_{Paper}
Food	23	WD_{Food}
Yard	33	WD_{Yard}
Wood	-	WD_{Wood}
Plastics	2	$WD_{Plastics}$
Textile	-	$WD_{Textile}$
Leather	-	$WD_{Leather}$
Rubber	-	WD_{Rubber}
Ferrous	1	$WD_{Ferrous}$
Non-Ferrous	<1	$WD_{Non\text{-}Ferrous}$
Glass	8	WD_{Glass}
Others	2	WD_{Other}

\+

Industrial Waste = ID(Tonnes)

Waste Distribution	%	Residual Waste (Tonnes)
Paper	27	ID_{Paper}
Food	26	ID_{Food}
Yard	3	ID_{Yard}
Wood	9	ID_{Wood}
Plastics	15	$ID_{Plastics}$
Textile	1	$ID_{Textile}$
Leather	<1	$ID_{Leather}$
Rubber	1	ID_{Rubber}
Ferrous	4	$ID_{Ferrous}$
Non-Ferrous	1	$ID_{Non\text{-}Ferrous}$
Glass	2	ID_{Glass}
Others	13	ID_{Other}

=

Residual Waste = Σ((WG - WD) + ID)

Waste Distribution	Residual Waste (Tonnes)	Waste Composition %
Paper	$((WG-WD)+ID)_{Paper}$	$((WG-WD)+ID)_{Paper}/\Sigma((WG-WD)+ID)$
Food	$((WG-WD)+ID)_{Food}$	$((WG-WD)+ID)_{Food}/\Sigma((WG-WD)+ID)$
Yard	$((WG-WD)+ID)_{Yard}$	$((WG-WD)+ID)_{Yard}/\Sigma((WG-WD)+ID)$
Wood	$((WG-WD)+ID)_{Wood}$	$((WG-WD)+ID)_{Wood}/\Sigma((WG-WD)+ID)$
Plastics	$((WG-WD)+ID)_{Plastics}$	$((WG-WD)+ID)_{Plastics}/\Sigma((WG-WD)+ID)$
Textile	$((WG-WD)+ID)_{Textile}$	$((WG-WD)+ID)_{Textile}/\Sigma((WG-WD)+ID)$
Leather	$((WG-WD)+ID)_{Leather}$	$((WG-WD)+ID)_{Leather}/\Sigma((WG-WD)+ID)$
Rubber	$((WG-WD)+ID)_{Rubber}$	$((WG-WD)+ID)_{Rubber}/\Sigma((WG-WD)+ID)$
Ferrous	$((WG-WD)+ID)_{Ferrous}$	$((WG-WD)+ID)_{Ferrous}/\Sigma((WG-WD)+ID)$
Non-Ferrous	$((WG-WD)+ID)_{Non\text{-}Ferrous}$	$((WG-WD)+ID)_{Non\text{-}Ferrous}/\Sigma((WG-WD)+ID)$
Glass	$((WG-WD)+ID)_{Glass}$	$((WG-WD)+ID)_{Glass}/\Sigma((WG-WD)+ID)$
Others	$((WG-WD)+ID)_{Other}$	$((WG-WD)+ID)_{Other}/\Sigma((WG-WD)+ID)$

Fig. 3. Methodology used to determine annual waste compositions. *Note*: The waste generated increases by 0.2% annually, the Residential Diversion rate progresses from 46% to 70% and the Industrial residual waste decreases by 0.05% annually.

analysis, the treatment of leachate from the landfill was not included. Furthermore, the more substantial aspect of managing ash landfills is the management of leachate. Therefore, the disposal of the ash was also not included to keep the scenarios comparable.

2.5. Waste quantity and compositions

An important aspect of this work is its ability to account for changes in waste quantity and composition as well as diversion rates. The method used to account for the various changes is summarised in the process chart (Fig. 3) below.

Fig. 3 describes the process undertaken for every year from 2011 to 2040 in order to determine the composition of the waste annually.

In an attempt to better simulate realistic waste management scenarios, the amount of residential waste generated annually increases by 0.2%, which is a projected population increase for the City of Toronto between 2011 and 2030 (City of Toronto, 2011); although, it is evident that factors, such as societal lifestyles and trends, in addition to population growth, affect the amount of waste being generated. The diversion rate which is initially 46% will increase to 70%, using an annual growth of 5% annually, in order to account for a continuous improvement in waste diversion effectiveness. The maximum residential diversion rate used in this model was 70%, which corresponds with the City's "Getting to 70% waste diversion from landfill" plan that would stretch the landfill lifetime expectancy of the landfill to 28 years (City of Toronto, 2011). It was estimated that the City of Toronto would go from 46% diversion to 70% residential waste diversion in the year 2020, at the rate which the amount of waste diverted is currently increasing.

It is important to note that due to a lack of data regarding industrial waste diversion for the City of Toronto, only the residential waste was diverted in this study. There is currently no reliable ICI waste generation or diversion baseline data for the province of Ontario. Information regarding other industrial waste trends

Table 1
Waste compositions used in this study.

MSW Components	Heat values (Gj/t)	Composition (% by weight) Residential waste Generated	Residential waste Diverted	Industrial waste Residual
Paper	16	26	31	27
Food	4	27	23	26
Yard	11	19	33	3
Wood[a]	17	2	–	9
Plastics	35	8	2	15
Textiles[a]	18	1	–	1
Leather[a]	17	<1	–	<1
Rubber[a]	25	<1	–	1
Ferrous	1	1	1	4
Non-ferrous	1	<1	<1	1
Glass	<1	5	8	2
Others	0	10	2	13

[a] MSW components are currently not included in the residential waste diversion programme.

for the purpose of this study was unavailable. ICI generators use the waste management industry for recycling services when the quantity, quality, frequency and value of waste generated make it unattractive for them to investigate, establish and execute diversion options outside the waste management system. Furthermore, ICI generators choose to divert recyclable material from waste destined for disposal when the quantity, quality and frequency make it economically attractive to do so (OWMA, 2006). Due to the fact that no figure supported by a reference could be found, and that the amount of waste is of a dynamic nature, a conservative percentage to represent a waste trend was chosen. The industrial residual waste was assumed to decrease by an arbitrary value of 0.05%. This decrease represents a trend of companies attempting to improve industrial processes in order to reduce the amount of waste being disposed for financial and environmental reasons.

All compositions, presented in Table 1, were determined based on the tonnage of waste, and are assumed to remain constant throughout the life of the study. The composition of the waste diverted was determined by analysing 5 years-worth of diversion data from the City, which showed that the composition of diverted waste remained fairly constant without any introduction of new waste diversion initiatives. The residual residential and industrial waste compositions were based on a detailed waste audit done for another Canadian city, Metropolitan Vancouver, as they were not available for the City of Toronto.

2.6. Life cycle inventory

The life cycle inventory was developed using a combination of publicly available LCA model technical reports, greenhouse gas inventory guidelines and LCA literature. Unfortunately, there was no publicly available software with the ability of providing the flexibility needed to incorporate various changing parameters.

The technical documents reviewed in the development of this life cycle inventory (LCI) are from the following models: the Canadian Integrated Waste Management Model for Municipalities (IWM), developed jointly by the commission of the Environmental Plastics Industry Council and Corporations Supporting Recycling (Haight, 2004); the Waste Reduction Model (WARM), developed by the US Environmental Protection Agency (US EPA, 2006); and the Municipal Solid Waste Decision Support Tool (MSW-DST) developed by Research Triangle Institute (RTI) for the US EPA Office of Research and Development. Other key literature used to develop this model include, the 2006 IPCC Guidelines for National Greenhouse Gas Inventories, the Canada's National Green House inventory Report (NIR) and the US EPA "Compilation of Air Pollutant Emission Factors" (AP-42). The emission factors used in this model are outlined in Tables 2–5.

Table 2
Incineration facility emission factors.

Pollutants	Parameters	Units	References
Arsenic	2.12×10^{-3}	kg/Mg	US EPA (1996)
Cadmium	1.36×10^{-5}	kg/Mg	US EPA (1996)
Chromium	1.50×10^{-5}	kg/Mg	US EPA (1996)
Nickel	2.58×10^{-5}	kg/Mg	US EPA (1996)
Lead	1.31×10^{-4}	kg/Mg	US EPA (1996)
CDD/CDF	3.31×10^{-8}	kg/Mg	US EPA (1996)
Mercury	2.80×10^{-4}	kg/Mg	US EPA (1996)
NO_x	2.75×10^{-1}	kg/Mg	US EPA (1996)
Sulphur dioxide	2.77×10^{-1}	kg/Mg	US EPA (1996)
Hydrogen chloride	1.06×10^{-1}	kg/Mg	US EPA (1996)
Particulate matter	3.11×10^{-2}	kg/Mg	US EPA (1996)
CO	2.31×10^{-2}	kg/Mg	US EPA (1996)

Note: Emission factors were calculated from concentrations using an *F*-factor of 9570 dscf/MBtu (0.26 dscm/Joule (J)) and a heating value of 4500 Btu/lb (10,466 J/g). Other heating values can be substituted by multiplying the emission factor by the new heating value and dividing by 4500 Btu/lb.

Table 3
Landfill parameters.

Landfill gas	Parameters	Units	References
Methane content	55	%	Pichtel (2005)
CO_2 content	45	%	Pichtel (2005)
Energy content	19,730	kJ/m^3	Pichtel (2005)
Gas collection efficiency	75	%	US EPA (2008)
Leachate characteristics			
BOD	279	mg/L	Green Lane Landfill (2006–2010)
COD	967	mg/L	Green Lane Landfill (2006–2010)
TSS	191	mg/L	Green Lane Landfill (2006–2010)
NH_3	247	mg/L	Green Lane Landfill (2006–2010)
Total nitrogen	352	µg/L	Green Lane Landfill (2006–2010)
Phosphorus	3	mg/L	Green Lane Landfill (2006–2010)

2.6.1. Air emissions

This study estimated the following air emissions of compounds for both the landfilling and incineration systems: Criteria Air contaminants (CAC); Greenhouse gases (GHGs); and acid gases. CACs are ozone precursors that consist of nitrogen oxides (NO_x), sulphur dioxide (SO_2), carbon monoxide (CO), volatile organic compounds

Table 4
Emissions from power generation for coal.

Pollutants	Parameters	Units	References
CO_2	1082	Mg/net-GWh	OPG (2009)
CH_4	n.a.	Mg/net-GWh	OPG (2009)
CO	0.19	Mg/net-GWh	OPG (2009)
NO_X	1.40	Mg/net-GWh	OPG (2009)
SO_X	3.09	Mg/net-GWh	OPG (2009)
TPM	0.28	Mg/net-GWh	OPG (2009)
HCl	0.11	Mg/net-GWh	OPG (2009)
N_2O	n.a.	Mg/net-GWh	OPG (2009)
VOCs	n.a.	Mg/net-GWh	OPG (2009)

Table 5
Waste haulage emissions.

Pollutants	Parameters	Units	References
CO_2	2263	g/L	Environment Canada (2009)
CH_4	0.14	g/L	Environment Canada (2009)
N_2O	0.082	g/L	Environment Canada (2009)
NO_x	10.2	g/vehicle-km	ICF (2007)
CO	1.64	g/vehicle-km	ICF (2007)
SO_X	0.20	g/vehicle-km	ICF (2007)
TPM	0.22	g/vehicle-km	ICF (2007)
HCl	0.11	g/vehicle-km	ICF (2007)
VOCs	0.3	g/vehicle-km	ICF (2007)

(VOCs), and particulate matter, including total particulate matter (TPM), particulate matter with a diameter less than or equal to 10 microns (PM_{10}), and particulate matter with a diameter less than or equal to 2.5 microns ($PM_{2.5}$) (Environment Canada, 2009). This study only considered the total particulate matter emissions. GHGs are comprised of carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O), sulphur hexafluoride (SF_6), perfluorocarbons (PFCs) and hydrofluorocarbons (HFCs) (Environment Canada, 2009). However, only CO_2, CH_4 and N_2O emissions were included in this study as emission factors for the rest of the GHGs were not common. The emissions of hydrogen chloride (HCl) were the only acid gas emissions reported for both technologies.

Only CO_2 emissions of fossil origin (e.g., plastics) were included in the CO_2 emissions estimate. It is important to note that, according to the IPCC 2006, textiles and rubber are comprised of approximately 0–50% and 20% of fossil fuel carbon respectively. However the default % of fossil fuel carbon suggested by IPCC (2006) is 20% for both textile and rubber, and that is taken into account in the calculations of anthropogenic carbon. The CO_2 emissions from the combustion of biomass materials (e.g., paper, food, and wood waste) contained in the waste are biogenic emissions and were not included in the CO_2 emission estimates (IPCC, 2006).

2.6.2. Incineration plant emissions

The incineration facility was modelled using a mass burn/waterwall design with a capacity of 1000 tonnes/day. The air pollution equipment in the WTE facility includes: a spray dryer for acid gas control; injection of activated carbon for mercury control; ammonia or urea injection by means of selective catalytic for reduction of NO_x; and a fabric filter for PM control. The WTE facility is assumed to be zero discharge with respect to waterborne pollutants. The greenhouse gases (CO_2, CH_4, and N_2O) for the incineration facility were calculated according to the methodology provided in IPCC (2006) while the heavy metals and acid gases emissions factors listed in Table 2 were from US EPA "Compilation of Air Pollutant Emission Factors" (AP-42).

The anthropogenic CO_2 was calculated by determining the amount of fossil fuel carbon in each MSW component while the other emissions were determined based on the heating value of the waste. Both the amount of fossil fuel carbon in the MSW components and the heating value of the MSW components are dependent on the MSW compositions and would be adjusted as the MSW composition changes.

The energy produced is recovered only as electricity, of which 20% will be used for in-house purpose with the remainder sold to the grid. The mass burn incinerator is assumed to have a conservative energy recovery efficiency of 20%. This efficiency corresponds to an incinerator that was built to minimise investment costs and is not optimised for power generation (AECOM, 2009). All auxiliary fuels required to run the facility are not included in this study.

The resulting bottom ash and fly ash are handled separately. The bottom ash and fly ash would account for 20% and 5% of the original weight of the waste, respectively. This assumption is consistent with what has been reported in literature (i.e. Sabbas et al., 2003; Hickman, 1999; Quina et al., 2010). In this study, no bottom ash was reused. Instead, the bottom ash was mixed with the fly ash for hazardous waste disposal. The environmental benefits and burdens of the ash reuse and ash disposal are not investigated in this LCA.

2.6.3. Landfill facility emissions

The landfill facility was designed as a sanitary landfill. Landfill gas is composed of mainly CO_2 and CH_4, but can contain trace concentrations of compounds such as VOCs and HCl. The quantity of CO_2 and CH_4 were determined using the Scholl Canyon model (see Eqs (1) and (2)), which is the most commonly used model for determining methane gas generation (US EPA, 2005). This model assumes that the lag phase is negligible and that CH_4 production is highest in the early phase, followed by a slow steady decline in annual production rates and that first-order kinetic rates apply. Although, the Scholl Canyon has been widely used, this study follows the landfill modelling method specifically used in Environment Canada (2009).

$$Q_{T,x} = kM_xL_oe^{-k(T-x)} \tag{1}$$

where $Q_{T,x}$ = the amount of CH_4 generated in the current year, (T) by the waste, M_x, tonnes CH_4/year, X = the year of waste input, M_x = the amount of waste disposed of in year x, tonnes, K = CH_4 generation rate constant/yr, L_0 = CH_4 generation potential, kg CH_4/t waste, T = current year.

$$Q_T = \sum Q_{T,x} \tag{2}$$

where Q_T = the amount of CH_4 generated in the current year (T), tonnes CH_4/year.

The CH_4 generation potential (L_0) represents the amount of CH_4 that could be theoretically produced per tonne of waste landfilled. It is determined using the amount of organic carbon that is accessible to biochemical decomposition, which is based on the composition of the waste (Environment Canada, 2009); therefore, as the waste composition is altered, the annual landfill gas emissions are also modified through L_0.

Landfill gas (LFG) is composed of many constituents such as nitrogen, oxygen and hydrogen, in addition to methane and carbon dioxide. However, only compounds contributing to the formation of HCl, SO_2 and volatile organic compounds (VOCs) were included in this analysis for consistency purposes. The concentration of VOCs was expressed in terms of hexane.

In estimating HCl emissions, it was assumed that all of the chloride from the combustion of chlorinated LFG constituents is converted to HCl. The chlorinated constituents used in this analysis were: dichloromethane, 1,1,1-Trichloroethane (methyl chloroform), and perchloroethylene; these compounds represent the LFG constituents that are that are most prevalent in LFG. Concentrations of reduced sulphur compounds within the LFG were used to estimate of SO_2 emissions. The sulphur compounds consisted of hydrogen sulphide and dimethyl sulphide as these gases appear in the greatest concentrations (US EPA, 2008).

The quantity of HCl, SO_2 and VOCs compounds emitted by the landfill was estimated using methods and emission factors provided by US EPA (2008).

Landfill leachate is produced from precipitation that falls directly on the site and percolates through the landfill cover (daily, intermediate, or final) into the waste. For the purpose of this study, a method that related the quantity of leachate directly to the average precipitation was used for simplification. The following values of leachate production as a percentage of precipitation are based on field data (Environmental Research and Education Foundation, 1999). These constants were developed by EREF (1999) using empirical data and the US EPA HELP (Hydrologic Evaluation of Landfill Performance) model as resources.

- Leachate Production Period 1: waste 0–1.5 years old, 20% of precipitation.
- Leachate Production Period 2: waste 1.5–5 years old, 6.6% of precipitation.
- Leachate Production Period 3: waste 5–10 years old, 6.5% of precipitation.
- Leachate Production Period 4: waste 10 years old and older, 0.04% of precipitation.

This leachate estimation method and the default parameters are valid for the gradual covering of a landfill. In reality, some parts of the site may never be covered with intermediate cover and be directly covered by final cover (EREF, 1999). A volume of precipitation can be calculated given the precipitation in depth/year and an area of landfill surface. A certain percentage of that volume ends up as leachate depending on the time after the placement of the waste. Together, these values provide the amount of leachate generated per area of landfill surface. Furthermore, if the tonnes of waste placed per area of landfill surface are known, then the quantity of leachate per tonne of waste can also be determined. For a numerical example of how to use these constants and further details on the methodology used to obtain these, see EREF (1999). The leachate quality information shown in Table 3 is an average of the concentrations reported by the Green Lane Landfill progress annual reports between 2005 and 2009.

As stated in Table 3, this study assumes that 75% of the landfill gas is collected, as suggested by US EPA (2008). The landfill gas collected is used for energy recovery in the form of electricity. Other energy recovery technologies such as combined heat and power were not analyzed. The remainder of the gas escapes to the environment and is considered a source of greenhouse gases. As the remainder of the gas passes through the landfill cover, a portion of the methane is oxidized. It is assumed that 10% of the methane that is not captured will be oxidized (IPCC, 2006), although, according to Spokas et al. (2006), total methane oxidation rates can from 4% to 50% of the methane flux through the cover at sites with positive emissions.

Finally, the default energy recovery efficiency from LFG was reported to be 30% (in gas turbines). The energy recovery efficiency is consistent with that stated in Diaz and Warith (2006) and Bove and Lunghi (2006). Auxiliary fuels needed to operate the technology are out of the scope of this study. It is assumed that 20% of the electricity generated was used for in-house purposes while the remainder is sold to the grid.

2.6.4. Avoided emissions from power plants

The electricity generated from the waste management facilities offsets only emissions from thermal power plants of which four are fuelled by coal and the fifth by oil and natural gas. The thermal stations' (coal and natural gas) role is to generate electricity, complementing generation produced by lower cost nuclear and hydroelectric facilities. Thermal stations provide a flexible source of energy and can operate as base load, intermediate and peaking facilities depending on the needs of the electricity system (Ontario Power Generation [OPG], 2009). These power plants are a significant source of anthropogenic CO_2, NO_x and SO_x amongst other pollutants. In Ontario, nuclear and Hydro are used primarily for meeting base load demand (i.e. the minimum amount of electricity demand, regardless of time of day or season). The thermal stations' role is to generate electricity, complementing generation produced by lower cost nuclear and hydroelectric facilities. The pollutants emitted by the thermal power plants are presented in Table 4.

2.6.5. Waste haulage emissions

This study examines the environmental burdens for only the transportation of the waste from the City to the waste facility. The vehicles are classified as Class 8 vehicles and run on diesel fuel. These trucks have an average fuel efficiency of 41.5 L/100 km (ICF, 2007). It was assumed that the trucks would have a load of 37 tonnes. The truck load is based on the figures used in the contract for Waste Transportation/Haulage Services from the City of Toronto's Transfer Stations to the Green Lane Landfill (City of Toronto, 2007). The Criteria Air Contaminants (CACs) emission factors and Greenhouse Gas (GHG) emissions were provided by Environment Canada and are listed in Table 5.

3. Results

3.1. Waste composition

The changes in composition caused by residential waste diversion are important because the waste composition determines the energy content for incineration and the methane generation potential for landfilling. These parameters determine the amount of energy that can be recovered from both waste management methods. The changes in waste composition are presented in Table 5.

The waste groups that were subjected to diversion were: paper; food; yard; plastics; ferrous and non-ferrous material; and other waste. The increase in residential waste diversion from 46% to 70% caused the presence of selected waste group to also increase in the waste stream (see Fig. 4). Plastics and other waste account for approximately 20% of the waste generated whereas only less

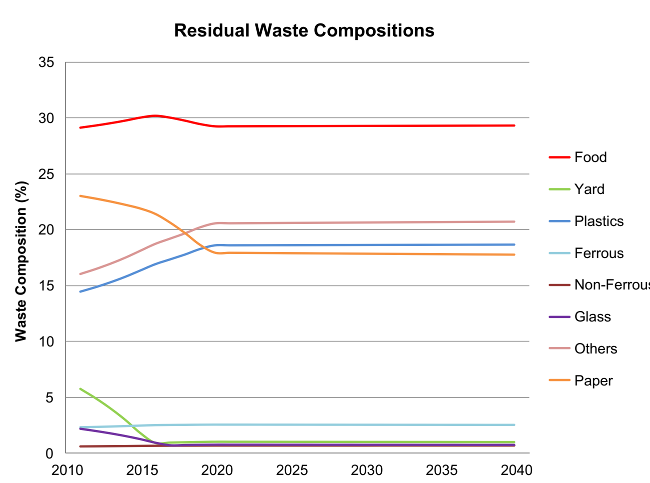

Fig. 4. Compositions of total residual waste.

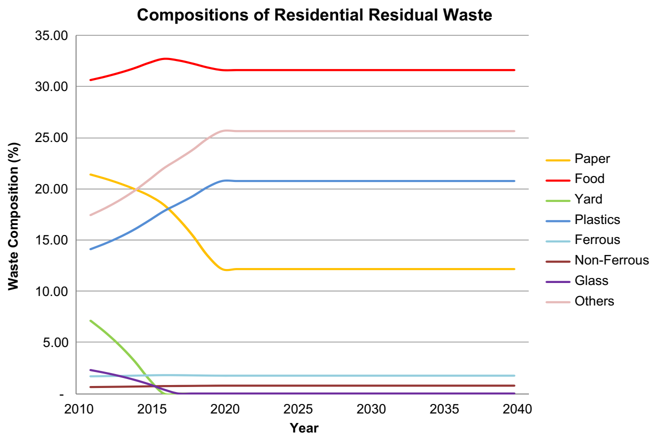

Fig. 5. Composition of residential residual waste.

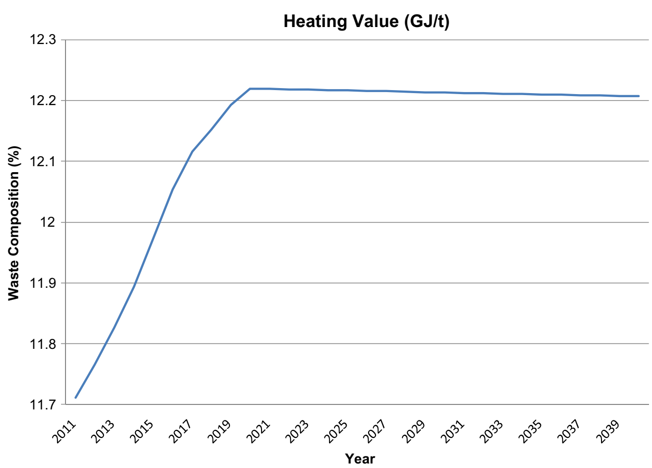

Fig. 6. The residual heating value (Gj/t).

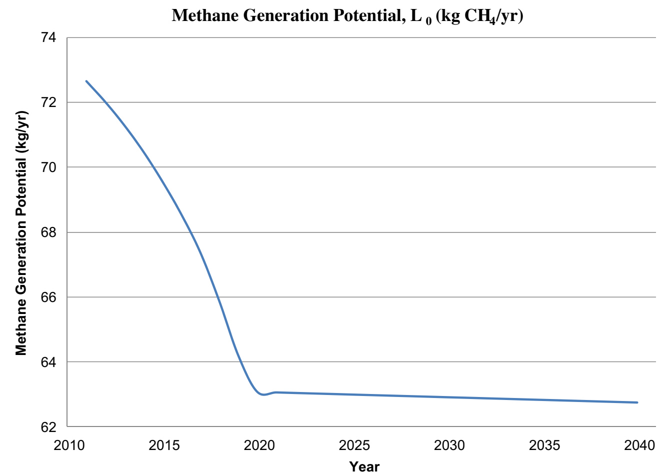

Fig. 7. Methane generation potential for the landfill option.

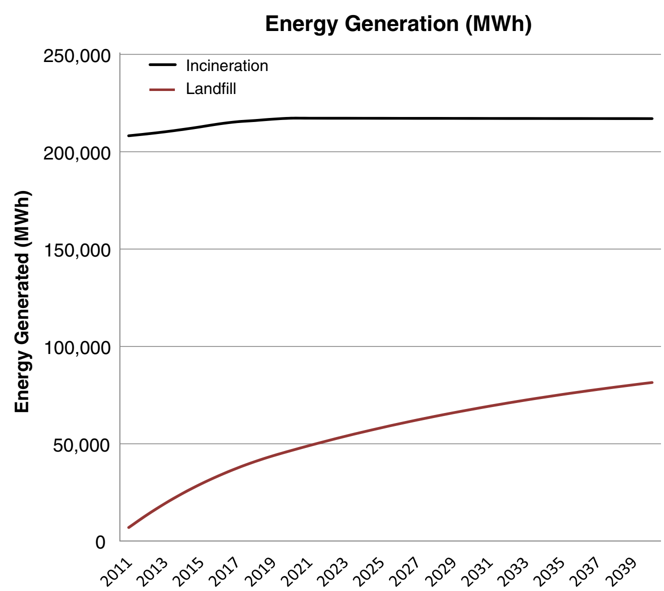

Fig. 8. Comparison of energy generated from incineration and landfilling.

than 4% of that waste makes up the waste being diverted. Therefore, the significant decrease in the tonnage of paper, yard and food waste through diversion causes the distribution of the plastic and "other" waste streams to increase. Ferrous and non-ferrous metals constitute such a small percentage of the diverted waste that their composition is not noticeably affected.

A plateau is created once 70% diversion is reach because once the maximum diversion is reach, the diversion rate becomes constant and the waste composition becomes influenced only by the waste generation rate, which is currently 0.2%.

Since the industrial residual waste has a slightly different waste distribution and was unsorted, the influence of the residential waste diversions was slightly diminished; however, the residual waste follows the waste composition trend of the residential waste (see Fig. 5).

3.1.1. Energy content and methane generation potential

The reduction of organic wastes (i.e. paper, food and yard) and the increase in other wastes such as plastics, leather and rubber waste, as a result of an increase in diversion rate, enhances the energy content of the residual waste. This in turn increases the amount of electricity that could be potentially produced through incineration (see Fig. 6). On the other hand, the reduction in the organic content of the waste, mainly paper and yard waste, reduces the methane that can be potentially generated and recovered for energy from landfilling (see Fig. 7).

With regards to the amount of energy generated, the incineration option generation significantly more energy than the landfilling option (Fig. 8). The energy generation from the incineration option is a more constant as opposed to the landfill energy generation that will peak and then decrease to a point where energy recovery is no longer possible.

3.2. Life cycle impact assessment

Environmental impact categories were used to facilitate the environmental comparison between the two waste management technologies and to allow for a clear presentation of the results. This analysis only included the following categories: global warming potential (GWP), acidification potential (AP) and nutrient enrichment potential (NEP), which are the most common impact categories included in the LCIA phase. The impact categories, their respective emissions, and equivalency impact factors applied in this study are presented in Table 6.

Global warming potential (GWP) accounts for the emission of greenhouse gases (CO_2, CH_4, N_2O), whose characterisation factors are based on the model developed by the Intergovernmental Panel

Table 6
Impact categories, emissions, and equivalency factors.

Global warming[c] potential 100 years (kg CO_2)		Acidification (g SO_2)[d]		Nutrient enrichment[d] potential (g NO_3^-)	
Emissions	Equivalency factors	Emissions	Equivalency factors	Emissions	Equivalency factors
CO_2[a]	1.00	SO_2[a]	1.0	Total[b] nitrogen	4.43
CH_4[a]	21	NO_2[a]	0.70	NO_x[a]	1.35
N_2O[a]	320	HCl[a]	0.88	N_2O[a]	2.82
				Total[b] phosphorus	32.03

[a] Emissions to air.
[b] Emissions to water.
[c] Source: Environment Canada (2009).
[d] Source: Mendes et al. (2004).

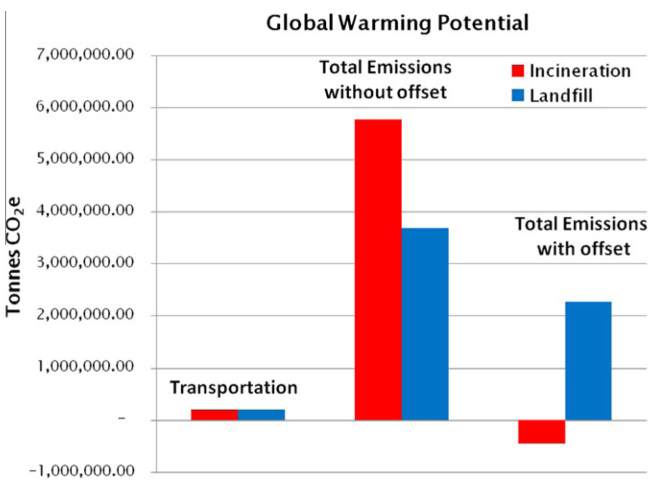

Fig. 9. Global warming potential results for incineration and landfilling option.

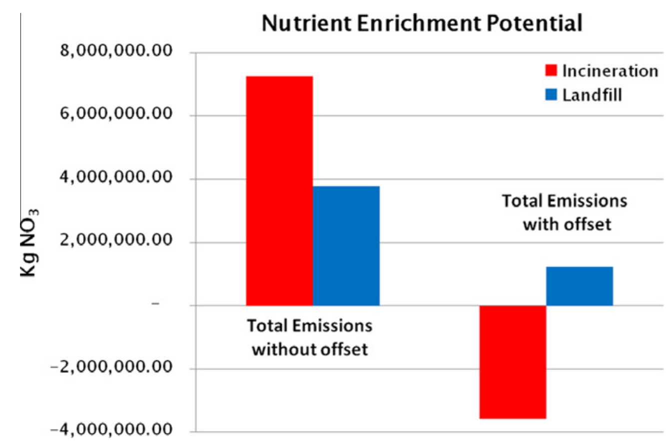

Fig. 11. Nutrient enrichment potential results for incineration and landfilling option.

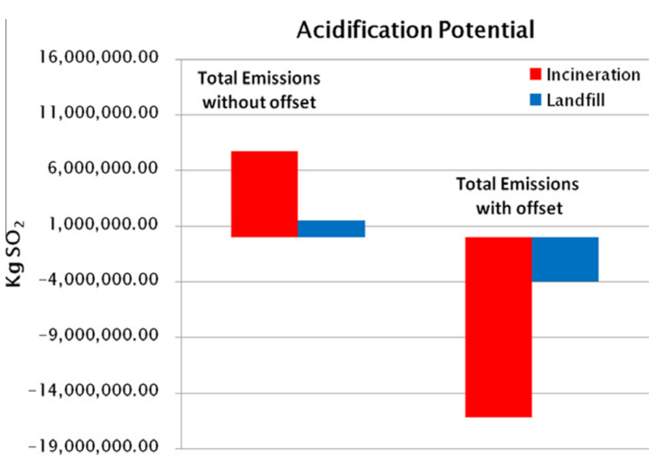

Fig. 10. Acidification potential results for incineration and landfilling option.

Table 7
Summary of all the costs and revenues included in the model.

Scenario	Incineration facility	Landfill facility
Revenues	–Electricity generation –ICI disposal fees –Material recovery	–Electricity generation –ICI disposal fees –Landfilling disposal fees
Costs	–Capital costs –Operation costs –Waste haulage to incineration facility cost –Ash disposal costs –Landfill disposal costs –Waste haulage to landfilling facility costs	–Capital costs –Operation costs –Waste haulage to landfilling facility costs

on Climate Change (IPCC, 2006) and referred to a time horizon of 100 years (GWP100). "Greenhouse gases" (GHGs) refers to the gases (primarily water vapour, carbon dioxide, methane and nitrous oxide) present in the earth's atmosphere which contribute to global temperatures through the greenhouse effect (Feo and Malvano, 2009). Fig. 9 shows the GWP expressed in tonnes CO2e. The majority of the emissions come from the operation of the waste management facilities as the emissions from transporting the waste to the facility can be considered insignificant. The CO_2 emissions result from the landfilling option mainly due to the combustion of methane, whereas the CO_2 emissions from the incineration facility result from the combustion of plastics. In addition, the gas recovery system significantly decreased the uncontrolled methane and VOCs emissions. As stated previously, only anthropogenic CO_2 was considered in this analysis, consequently, a large quantity of CO_2 released by the landfill were disregarded. Furthermore, plastics are stable elements and therefore contribute little to the methane generation.

Acidification potential (AP) is the process whereby air pollution, mainly ammonia, sulphur dioxide and nitrogen oxides, are converted into acidic substances. Some of the principal effects of air acidification include lake acidification and forest decline (Feo and Malvano, 2009). Acidification Potential (AP) accounts for the emissions of NO_x, SO_x and ammonia. Fig. 10 shows the AP, expressed as kg of SO_2 equivalent per kg of emission. The incineration option performed more poorly from an environmental perspective than the landfill option in terms of AP. Compounds such as sulphur

Table 8
Incineration cost parameters.

Parameters	Value	Units
Capital investment	300,000,000	$
Landfill disposal costs	20	$/tonne
Operating and maintenance costs	47	$/tonne
Residue disposal costs	100	$/per tonne
Transportation costs		
Waste haulage costs	18	$/per tonne
Fuel surcharge costs	4	$/per tonne
Revenue		
Electricity price	0.04	□/kwh
Tipping fees	40	$/tonne
Customer price index	2	%
Discount rate	5	%
Days of operation	320	Days

Table 9
Landfill cost parameters.

Parameters	Value	Units
Capital investment[a]	260,000,000	$
Operating and maintenance costs	18	$/tonne
Transportation costs		
Waste haulage costs	18	$/per tonne
Fuel surcharge costs	4	$/per tonne
Revenue		
Electricity price	0.04	□/kwh
Tipping fees	40	$/tonne
Customer price index	2	%
Discount rate	5	%
Days of operation	320	Days

[a] The capital investment includes the cost of the land as the landfill is already built and operating.

dioxide, nitrogen dioxide and hydrogen chloride are emitted at much higher concentrations with incineration compared to landfilling. The amount of sulphur dioxide and hydrogen chloride emitted from incineration is dependent on the sulphur and chlorine content in the waste. Furthermore, landfill gases such as sulphur dioxide, nitrogen dioxide and hydrogen chloride; typically occur in concentrations less than 1% (v/v).

Nutrient enrichment potential (NEP) or Eutrophication is the enrichment of mineral salts and nutrients in marine or lake waters from natural processes and manmade activities such as farming (Emery et al., 2007). Fig. 11 illustrates the NEP is expressed as g NO_3^- which includes both emissions to air and to water. It accounts for the total phosphorus and nitrogen in the water and the NO_x and N_2O emissions in the air. The landfilling option has a noticeably smaller eutrophication impact on the environment. The majority of the emissions that contribute to the landfilling option's NEP result from the leachate produced. However, the incineration emissions include greater NO_x and N_2O air emissions, in addition to the total dissolved nitrogen and phosphorus water emissions, which originate from the leachate produced by the remaining waste landfilled.

In this study, it was assumed that the energy produced by both options would displace the emissions generated by thermal power plants. When the environmental offsets are applied, incineration outperforms landfilling in all environmental categories (Figs. 9–11). Nitrogen oxides (NO_x), sulphur dioxide (SO_2) are among the most prominent air emissions from thermal energy facilities. As a result, the acidification potential is significantly impacted by the electricity offset. The incineration option receives environmental offsets from the electricity produced both from the combustion as well as from the remainder waste that is landfilled. With the inclusion of the electricity offset, the incineration option performs better environmentally because incineration generates significantly more electricity than landfilling.

3.3. Financial analysis

An economic analysis was done in addition to the environmental LCA, in order to carry out a more complete comparison of the technologies. The costs include operational and maintenance costs and costs associated with the haulage of the waste, while the revenues are comprised of waste-drop off fees, electricity sales and material recovery. A summary of all of the costs and revenues

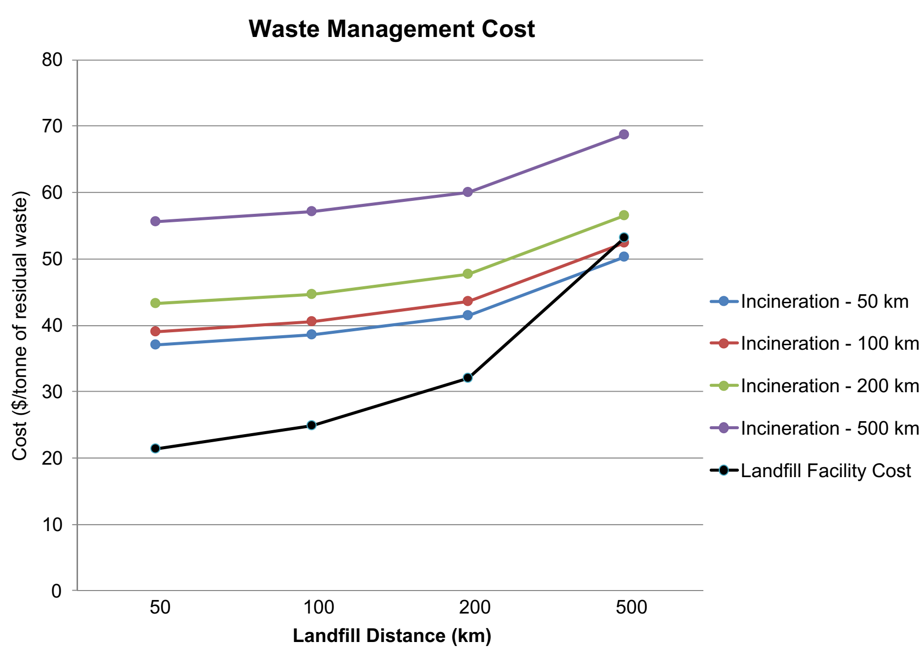

Fig. 12. Waste management cost with respect to distance from city core.

can be found in Tables 7–9. In this study, the costs are reported on a per tonne basis and in actual Canadian dollars to facilitate comparisons.

The results for the baseline scenario indicate that the incineration option costs net $48.54/tonne whereas the landfill option costs net $32.85/tonne[2]. In the time span analyzed (2011–2040), the landfilling option generates approximately $400 million in revenue compared to incineration that generates approximately $610 million. The elevated cost of incineration option is due to the greater number of costs incurred compared to landfilling, which are listed in Table 7.

In attempt to identify scenarios in which the incineration option may become more financially feasible, the effects of distance on the financial results were evaluated because transportation accounts for a significant part of the costs (see Fig. 12). The proximity of the incineration facility and the landfill facility to the City core were varied. It is important to the note that due the limited incinerator capacity, the remainder of the waste is sent to a landfill. The distance of this landfill is also considered in this sensitivity analysis. All the facilities are located 200 km one-way from the City core in the baseline scenario. Distances of 50, 100, and 500 km for the incinerator and landfill from the City core were used in the sensitivity analysis. No distances above 500 km were examined because these types of distances would most likely require waste transport by train.

In the financial model, the cost for the haulage services of the waste to the facilities are a function of distance. Consequently, the cost of both waste management options increases as the facilities are located further away from the City. The incineration facility becomes competitive financially when the landfill facility is located 500 km away from the City and the incineration facility is located 50 to 100 km away with its corresponding landfill facility located 50 to 200 km away from the City. Landfilling all the waste remains the preferred financial option with all the other scenarios examined in this model.

4. Conclusion

The goal of this study was to compare the use of an incineration and landfilling facility in the management of residual waste while accounting for residential waste diversion initiatives, from both an environment as well as a financial perspective. Waste diversion initiatives have become an integral part of the waste management process and it is important to be able to understand how these initiatives affect the waste composition. The waste composition will ultimately dictate the type of waste management method that is the most suitable.

The results indicated that the use of an incineration facility to manage a portion of the waste was better environmentally while landfilling all of the waste would be preferred financially. The waste management option that included the incineration facility performed better environmentally because the incineration facility produced significantly more electricity compared to the landfilling facility, and therefore a noticeably greater environmental offset.

The residual waste composition was significantly impacted by the residential diversion initiatives and increasing diversion rate. The residential waste diversion initiatives proved to be more successful for the organic waste streams (i.e. paper, food and yard waste). Consequently, as the diversion rate increased from 46% to 70%, the significant removal in the organic waste streams caused the composition of other waste groups such as plastics and "other" waste to increase considerably. The removal of organic content also reduced the amount of energy that could be recovered from the landfill by decreasing the amount of methane generated by the landfill. Conversely, the rise in inert wastes like plastics improved the energy recovery capability of the incineration option by increasing in the energy content of waste.

The diversion of these inert waste groups is important because they reduce the landfill capacity without contributing to the generation of methane and energy recovery process. Consequently, the most effective manner of handling these waste groups would be through the increase waste reduction and diversion initiatives as well as incineration. The incineration technology would actually benefit greatly from the presence of plastics in the residential waste stream due to its high energy content, while reducing the quantity to waste being landfilled. The only benefit to incinerating "other" waste since this waste component has no calorific value would be the volume reduction, which would also extend the life of an existing landfill.

This study is an improvement in the undertaking of municipal solid waste (MSW) life cycle assessments where many studies have assumed a constant MSW composition. More updated emission factors and more advanced waste quantity predictive methods would yield more accurate and realistic results. The inclusion of current waste diversion initiatives and a changing waste composition is one step closer towards carrying out an analysis that better reflects the realities in MSW management.

However, an LCA typically does not yield objective answers and the methodology also suffers from large uncertainties. Furthermore, an LCA entails a drastic simplification of the complex reality (Ekvall et al., 2007). Simplifications and assumptions made to reduce the complexity of this analysis diminished the completeness of the LCA. A more complete analysis is required if the results are to be used for decision-making purposes. A major assumption that was made in this study was that the composition of the waste diverted would remain constant over the life of the study. This assumption implies that no new diversion initiatives would be introduced or that technological advancements would not affect the waste diverted. Therefore, in order for the results to remain relevant, future LCAs should be done as new waste diversion initiatives are launched or as new waste management technologies become mainstream.

Other assumptions included the omissions such as the effects of ancillary processes and of leachate treatment for both the hazardous ash and landfill. The processes should be included in future studies to improve the completeness of the analysis. Furthermore, this study considered electricity as the only form of energy recovery for simplification purposes, however, the effects other forms of energy recovery systems, such as combined heat and power, should be explored further.

The capacity of the landfill is an important parameter that was not included in this study, but should be considered in future studies. It was assumed that the landfill would be able to handle all the residual waste generated regardless of the residential and industrial waste generation rate. In reality, an increase in waste generation would reduce the life of the landfill dramatically and could cause additional financial spending. In this type of scenario, incineration could become a more economically feasible option.

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[2] Costs represent net discounted costs per tonne.

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