Seasonal k and Independent Carbon Dioxide Approaches
For First Order Decay Landfill Gas Modelling

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By
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Regina, Saskatchewan
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Nathan Edward Paul Bruce, candidate for the degree of Master of Applied Science in Environmental Systems Engineering, has presented a thesis titled, *Seasonal k and Independent Carbon Dioxide Approaches For First Order Decay Landfill Gas Modelling*, in an oral examination held on December 2, 2016. The following committee members have found the thesis acceptable in form and content, and that the candidate demonstrated satisfactory knowledge of the subject material.

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ABSTRACT

Canada's per capita solid waste disposal rates are among the highest in the world. Landfill gas generation requires more accurate modelling in order to properly compare future emission mitigation or energy production projects. The EPA software LandGEM was selected for its common use in the literature. Two alternative methods to increase accuracy in methane and carbon dioxide estimates were studied.

Real-time methane collection data from a municipal landfill in Regina's cold, semi-arid climate were consolidated to fit a linear-interpolated form of LandGEM. LandGEM defaults were found invalid for this landfill due to significant overestimation (76.5% error). Seasonal variations in gas collection were explored, and found that optimal seasonal $k$ and $L_o$ collection parameters had 7.3% error compared to field data, compared to 15.5% error using optimal annual parameters. The optimal $k_{\text{winter}}$ was 0.0082 year$^{-1}$ and the $k_{\text{summer}}$ was 0.0095 year$^{-1}$ (14.7% difference). Three pseudo-second order iterative methods were used to fit the model estimates to the daily data, and they were evaluated using RSS and literature values. Optimized parameters were applied to a simple study using LFGcost-Web. The default parameters overestimated the net present worth by 57-107% for three of the four projects.

LandGEM assumes that carbon dioxide estimates are a function of methane, and that the two gases make up nearly 100% of gas content. This can lead to oversights in collection system design. A total of 25 cases (five formulas, five approaches) were compared for carbon dioxide collection at four western Canadian landfills. The existing Default with Traces approach overestimated production in 3 of the 4 sites, resulting in the highest
RSS. LandGEM's governing formula yielded the most accurate results under this approach (mean RSS increased by 7.0 to 49.9% using other equations). Optimization resulted in the most accurate results for all formulas and approaches, and had the greatest reduction in RSS over the default approach (73.0 to 98.0%). The 1.4 ratio approach yielded the second most accurate results (mean RSS reduction of 66.5%). The annual $k$ formula calculated $k$’s via two empirical formulas (based on precipitation), and yielded the lowest accuracy in 12 of 20 approaches.
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List of Abbreviations and Acronyms

B.C.: Canadian province of British Columbia
CC: Cache Creek landfill
CH₄: Methane
CNG: Compressed Natural Gas
CO₂: Carbon dioxide
COR: City of Regina
EC: Environment Canada
EPA: Environmental Protection Agency (from the United States, also called U.S. EPA)
FOD: First Order Decay
HL: Hartland landfill
LFG: Landfill Gas
MCF: Methane Correction Factor (dimensionless)
MSW: Municipal Solid Waste
NMOC: Non-Methane Organic Compounds
pH: power of Hydrogen, a measure of acidic conditions
Prec.: Precipitation
RE: Regina landfill
RSS: Residual Sum of Squares
RTI: Research Triangle Institute
SK: Saskatoon landfill
U.S.: United States of America
w/: with
w/o: without
yr: year
List of Symbols

DOC: Degradable Organic Carbon (dimensionless)

DOC\(_i\): Degradable Organic Carbon fraction (dimensionless)

i: Scholl Canyon formula 1 year time increment (dimensionless)

j: Scholl Canyon formula 0.1 to 1 year time increment (dimensionless)

k = Scholl Canyon formula methane generation rate (year\(^{-1}\))

k\text{winter}: winter methane collection rate in proposed model (year\(^{-1}\))

k\text{summer}: summer methane collection rate in proposed model (year\(^{-1}\))

L\text{\_o}: Scholl Canyon formula potential methane generation (m\(^3\)/Mg disposed waste)

L\text{\_o-CO\_2}: potential carbon dioxide generation in proposed approaches (m\(^3\)/Mg disposed waste)

M\(_i\): Scholl Canyon formula Mass of waste disposed in year i (Mg or tonnes) (Scholl Canyon formula)

n: Scholl Canyon formula year of calculation - initial year of waste acceptance (year)

P\text{CH\_4}: methane content (\%) 

Q\text{CH\_4}: methane generation or collection in the year of the calculation (m\(^3\)/year)

Q\text{CO\_2}: carbon dioxide generation or collection in the year of the calculation (m\(^3\)/year)

Q\text{i}: see Q\text{CH\_4}

Q\text{mCH\_4} = methane flow rate in proposed model (m\(^3\)/min)

R\text{\_\_2}: Coefficient of determination (dimensionless)

t: days since the landfill began accepting waste in proposed model

t\(_i\): days since the landfill opened at December 31 of the year of calculation in proposed model

\(t\text{\_ij}\): Scholl Canyon formula age of the j\(^{th}\) section of waste mass M\(_i\) disposed in the i\(^{th}\) year (decimal years)

\(\bar{x}\): mean value in the data set
CHAPTER 1: INTRODUCTION & LITERATURE REVIEW

1.1 Solid Waste in Canada and Landfill Gas Modelling

Canada’s solid waste generation rate is among the highest in the world (Bruce et al., 2016). Per capita nonhazardous waste generation ranged between 876 and 961 kg per year between 1996 and 2010, with a peak of 1033 kg in 2006 (Bruce et al., 2015; Wang et al., 2016). Per capita generation remained was 32% higher than in the U.S. in 2010 (965 kg/year). In 2010, a majority of all nonhazardous solid waste collected in Canada (75.6%) was disposed in landfills. This proportion was 86.8% in Saskatchewan, a province with a poor waste diversion record (Statistics Canada, 2013; Wang et al., 2016).

While source reduction and diversion practices are the most effective means of reducing environmental footprints, these practices are not able to mitigate the impacts of waste already disposed in landfills in Canada and throughout the world. Proper landfill design and management before, during, and after disposal operations will therefore remain a necessary field for the foreseeable future due to environmental threats posed by waste in place. In Canada, landfill gas (LFG) collection remains an under-utilized technology due to project costs and a low population density (4 vs. 35 cap/km² in U.S.) (World Bank Group, 2016); only fifty-two landfills across the vast country (9.985M km² of land) operated LFG collection systems by the mid 2000s (Thompson et al., 2009). Additionally, accurate LFG measurement can be expensive, and accurate modelling remains difficult.

Due to anaerobic decomposition, LFG is generated within landfills during and after operation, and continues decades after final closure depending on site conditions and
waste organic content. The major components of LFG are the greenhouse gases methane (CH₄) and carbon dioxide (CO₂), trace gases, air, non-methane organic compounds (NMOCs), and evaporate. Their average dry volumetric proportions are provided in Figure 1-1. LFG management systems are used primarily to prevent methane migration to neighbouring sites, and mitigate emissions that pose aesthetic, health, and safety (i.e. explosive nature of CH₄) threats. These systems can be used for heating or electricity generation projects using methane combustion as an energy source, although the most common project in North America is LFG flaring (Lindberg et al., 2005; Mohareb et al., 2008; Rajaram et al., 2011; Sanchez, 2016; Tolaymat et al., 2010) mainly due to its status as the cheapest alternative in terms of development costs (U.S. EPA, 2015). Operating costs depend on geographic location, available markets, gas quality and quantity (Ahmed et al., 2015; Albanna et al., 2007; Sanchez, 2016). Flare systems convert collected methane into carbon dioxide emissions, a less harmful product due to methane's higher global warming potential (25 for the 100-year time horizon) (IPCC, 2007).
Figure 1-1. Typical LFG composition, dry volume basis (after Tchobanoglous & Kreith, 2002)
Emission mitigation techniques require a means to estimate generation, collection, and emissions to determine the optimal solutions. Due to the temporal and spatial variability of LFG emissions (Bogner et al., 1999; Borjesson & Svensson, 1997; Christophersen et al., 2001; Klusman & Dick, 2000; Scharff et al., 2000), numerical models have been a vital tool in LFG management, as direct measurements at the surface are more expensive. Numerous models have been developed, and many are publicly accessible such as LandGEM, Scholl Canyon, GasSim, Afvalzorg, CALMIM, IPCC, and TNO. Among these, first order decay (FOD) or kinetic generation models are the most common type (Kamalan et al., 2011; Thompson et al., 2009). While IPCC and Afvalzorg are common in Europe, LandGEM is more common in North and South American studies (De la Cruz & Barlaz, 2010; Maciel & Juca, 2011). As such, LandGEM was selected for further study with available data from Canadian landfills.

LandGEM originally appeared under the name Landfill Air Emissions Estimation Model, or LAEEM, developed by the U.S. EPA. The model's governing formula was derived from the empirical, FOD Scholl Canyon model developed by Emcon Associates (1980). LandGEM Version 3.02 (Alexander et al., 2005), the newest version of the software, assumes that nearly 100% of LFG generation consists of CH$_4$ and CO$_2$. CH$_4$ estimates are largely sensitive to site-specific or selected variables “k” (CH$_4$ generation rate constant, or decay constant) and “L$_0$” (CH$_4$ generation potential) (Aguilar-Virgen et al., 2014; Amini et al., 2013; Atabi et al., 2014; Machado et al., 2009; Peer et al., 1993); neither variable’s recommended default values have changed since LAEEM. LAEEM's recommended "k" values were empirically derived from 44 U.S. landfills across various climates (Melcer et al., 1999).
The decay rate \( k \) (units = \( \text{year}^{-1} \)) is considered dependent on waste moisture content and precipitation rates, nutrients, \( \text{pH} \), bacterial culture, and waste temperature. \( L_o \) (units = \( \text{m}^3 \) of \( \text{CH}_4 \)/Mg of waste) largely depends on the amount and type of organic waste, which changes slowly over time due to decay (Ishii & Furuichi, 2013) and shifts in disposal trends (Thompson et al., 2009).

A common approach in LFG studies is to utilize curve fitting techniques to approximate \( k \) and \( L_o \). Amini et al. (2012) used composition data for five landfills in Florida in order to test four different approaches to \( k \) and \( L_o \) calculations. The approaches used were: (1) the Environmental Protection Agency's (EPA) suggested defaults; (2a) calculated \( L_o \) using lab measurements and site specific composition data, then linear regression fitting to determine \( k \); (2b) field methane collection for a closed landfill divided by total disposed waste to calculate \( L_o \), then linear regression fitting again; and (3) the expanded formula was solved for \( k \) and \( L_o \) simultaneously using non-linear regression to minimize the residual between actual and modelled annual LFG for each landfill. Approach “2a” was the most optimal by a slim margin, resulting in the lowest standard error (range from \( 6.8 \times 10^5 \) to \( 8.6 \times 10^5 \)). Approach “3” had equivalent standard error to approach “2a”, and even a slightly higher \( R^2 \) value (0.93 vs. 0.91), though it resulted in invalid \( k \) and \( L_o \) values in one case (\( k = 0.001\text{year}^{-1}, L_o = 3844\text{m}^3/\text{Mg} \)). The LandGEM defaults produced reliable results (\( R^2 = 0.89 \), compared to the highest 0.93), although this was likely due to Florida's temperate climate fitting the default value well.

Alternative models and formulas have been proposed and compared to existing models in the literature. Previous editions of LandGEM used an annual time increment, and found that LandGEM v.3.02’s change to 0.1 led to improvements in estimates, and
estimate reductions of 1-2% depending on the magnitude of k (Alexander et al., 2005). Thompson et al. (2009) compared five different LFG models, with an additional deeper study into the Scholl Canyon model, for 35 Canadian landfills. The models were run with DOC values of 0.50 and 0.77, and it was observed that 0.50 resulted in better fits for the data. For the unmodified models, LandGEM had the lowest mean absolute error for 0.77 DOC and the second lowest for 0.50 DOC. The Scholl Canyon model was also modified with divisors (reciprocal value of the time increment j) between 1 and 10. They found that a divisor of 1.5 had the lowest mean absolute error using L₀ (0.50), and 2.3 had the lowest mean absolute error using L₀ (0.77) via graphs. The use of decimal divisors rather than integers, however, contradicts the formula’s intention for splitting annual waste disposal values evenly. Standard error tended to decrease as divisor values increased in both L₀ (DOC) cases, with the largest range of error falling between divisors 1 and 3.

1.2 Climate influences on LFG

LFG generation is dependent on waste organic content, moisture and bacteria content, temperature, nutrients, pH, type and quantity of daily and final cover, waste density, and waste age (Ishii & Furuichi, 2013; Karanjekar et al., 2015; Machado et al., 2009; Opseth, 1998; Peer et al., 1993). LFG emissions, on the other hand, are dependent on generation, gas well design, cover type and thickness, moisture, ambient temperature and pressure, and operating factors such as effective extraction rates and compaction density (Borjesson & Svensson, 1997; Christophersen et al., 2001; Klusman & Dick, 2000; Sanchez, 2016).
Table 1-1 summarizes the results of other recent studies reporting \( k \) and \( L_o \) values for warm and cold climate landfills. Warm climates have consistently higher \( k \) values, partly due to the studies being located near coastlines (Amini et al., 2012; Karanjekar et al., 2015; Machado et al., 2009) and cities with high precipitation rates (Tolaymat et al., 2010). Aguilar-Virgen et al. (2014) reported a relatively low \( k \) (= 0.0482 yr\(^{-1}\)) given their warm climate, although this may be due to Ensenada's low precipitation rate (250 mm/year). Because Thompson's formula was based only on precipitation (Thompson et al., 2009), Quebec and Ontario’s reported \( k \) values are comparable to those in warm climates. For instance, Quebec's mean annual precipitation (1,070 mm) was high relative to Alberta (445 mm) and Regina (390 mm). Derivations for \( k \) vary widely between the 8 studies, indicating either the lack of a recognized optimal method, or varying degrees of details in available data. Ishii and Furuichi's (2013) study was included to highlight the range in \( k \) and \( L_o \) for different organic MSW components used in the more complex IPCC model.
### Table 1-1. k and L₀ values determined for warm and cold climates

<table>
<thead>
<tr>
<th>Location</th>
<th>k (year⁻¹)</th>
<th>L₀ (m³/Mg)</th>
<th>Data span</th>
<th>k derived from</th>
<th>L₀ derived from</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Warm climate</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Salvador, Brazil</td>
<td>0.2 - 0.21</td>
<td>65.9 - 67</td>
<td>2004-2006</td>
<td>FOD model, curve fitting w/ field data</td>
<td>Biochemical CH₄ potential related equations, IPCC formula using BF values for DOE, curve fitting w/ field data</td>
<td>Machado et. al., 2009</td>
</tr>
<tr>
<td>Louisville, U.S.</td>
<td>0.11 (bio.)²</td>
<td>48.4 - 54.8</td>
<td>2002-2007</td>
<td>Modified LandGEM &amp; samples</td>
<td>Biochemical CH₄ potential measured in lab using 3-11 week old samples</td>
<td>Tolaymat et. al., 2010</td>
</tr>
<tr>
<td>Florida, U.S.</td>
<td>0.04 - 0.13</td>
<td>56 - 77</td>
<td>3 - 16 years</td>
<td>Linear regression fitting</td>
<td>Waste composition records and component-specific L₀ from lab in literature</td>
<td>Amini et. al., 2012</td>
</tr>
<tr>
<td>Ensenada, Mexico</td>
<td>0.0482</td>
<td>94.457</td>
<td>3 months</td>
<td>Waste audit used to weight component-specific k’s in MBM 2.0 model</td>
<td>Waste audit and IPCC formula using DOCᵣ formula dependent on anaerobic zone temperature</td>
<td>Aguilar-Virgen et al., 2014</td>
</tr>
<tr>
<td>Victoria, Canada</td>
<td>0.08</td>
<td>n/a</td>
<td>n/a</td>
<td>Quote from source (Hartland Landfill)</td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Cold climate</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Alberta</td>
<td>0.023</td>
<td>100 - 178</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ontario</td>
<td>0.037</td>
<td>90 - 160</td>
<td>2006</td>
<td>k formula with prec. as independent variable, based on EPA defaults</td>
<td>Provincial composition records and IPCC formula using different DOCᵣ defaults: 0.77 (1996), 0.50 (2006)</td>
<td>Thompson et. al, 2009</td>
</tr>
<tr>
<td>Quebec</td>
<td>0.042</td>
<td>128 - 220</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Regina</td>
<td>0.023</td>
<td>n/a</td>
<td>2008-2014</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>W City, Japan</td>
<td>0.05 (paper)²</td>
<td>214.4 (paper)²</td>
<td>1988-2010</td>
<td>FOD model (IPCC) fitting using multi-age field samples</td>
<td>Field samples at different ages decomposed in lab batch testing</td>
<td>Ishii &amp; Furuichi, 2013</td>
</tr>
<tr>
<td></td>
<td>0.062 (food)²</td>
<td>126.7 (food)²</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Regina</td>
<td>0.011</td>
<td>123</td>
<td>1990-2007</td>
<td>k formula with prec. as independent variable, based on EPA defaults</td>
<td>Provincial composition records and IPCC formula where DOCᵣ = 0.6</td>
<td>Environment Canada, 2014</td>
</tr>
</tbody>
</table>

a) “bio.” denotes “bioreactor landfill cell”  
b) “conv.” denotes “conventional landfill cell”  
c) “prec.” denotes “precipitation”  
d) IPCC model parameters
Default $L_o$ values in LandGEM range between 96 to 170 m$^3$/Mg of waste in Arid and Conventional locations when the U.S. Clean Air Act estimates are required. However, the United States' (U.S.) EPA has identified the range of reasonable U.S. landfill $L_o$ values lies between 56.6 and 198.2 m$^3$/Mg (U.S. EPA, 1995). Table 1-1 shows that only Thompson et al. (2009) calculated $L_o$ values near LandGEM's upper boundary or high default (170 m$^3$/Mg), and 3 of the 4 warm climate $L_o$ values were actually closer to the reasonable lower boundary (56.6 m$^3$/Mg). Machado et al. (2009) recognized this opposes dominant trends in the literature, and suggested that samples from their site had oversaturated water content which countered the high organic content, as the IPCC formula for $L_o$ uses wet weight basis. This is consistent with the high $L_o$ value reported by Aguilar-Virgen et al. (2014), as their warm climate site experienced low annual precipitation. The IPCC formula uses organic waste composition on a wet weight basis, and half of the studies in Table 1-1 reported composition data as such (Ishii & Furuichi, 2013; Karanjekar et al., 2015; Machado et al., 2009; Tolaymat et al., 2010). There may have been bias in modelling approaches as well. Some of the studies calculated $L_o$, and then used regression or curve fitting techniques to determine $k$ (Amini et al., 2012; Machado et al., 2009; Tolaymat et al., 2010). Tolaymat et al. (2010) observed higher $k$ estimates when low $L_o$ values were used.

Recent cold climate LFG literature has been focusing on cover soils and oxidation mechanisms to mitigate methane emissions (Borjesson & Svensson, 1997; Chanton & Liptay, 2000; Christophersen et al., 2001; Klusman & Dick, 2000; Maurice & Lagerkvist, 2003). Some studies express doubt that LFG generation is affected by seasonal climate trends, although the study sites were located in temperate climates
(Chanton & Liptay, 2000; Scharff et al., 2000). Given the field evidence of seasonal variation in LFG emissions (Klusman & Dick, 2000), it is reasonable to expect that seasonal variations in LFG collection may exist in cold and arid climates where key LFG generation factors are limited.

Waste moisture content is directly related to LFG generation, and thus collection. Some studies have proposed or supported linear relations between moisture content and the decay rate $k$ in FOD models (Environment Canada, 2014; McDougall & Pyrah, 1999; Thompson et al., 2009). Aside from moisture content in the waste at disposal and groundwater infiltration, precipitation is the main source of moisture input for landfills. However, in cold climates, infiltration is often impeded during the winter season. Frozen soil can be subject to ice formation in the pores near the surface, or else develop a thin ice layer at the surface (Arnalds, 2015; Iwata, 2011; Orradottir, 2002). Thus, landfills in cold, dry climates can expect low LFG production rates due to the combined effect of low annual precipitation, and little to no infiltration during the winter. In addition, operating difficulties such as frozen wellheads and condensate issues are more common for landfills located in cold climates.

While some semi-arid and cold climate LFG studies have focused on field sampling for quantity (Klusman & Dick, 2000; Maurice & Lagerkvist, 2003; Opseth, 1998), and composition (Opseth, 1998), part of this study utilizes FOD gas modelling with real-time LFG data from Regina's (RE) semi-arid, cold climate landfill recorded between August 2008 and December 2014. In most decomposition-based models, methane generation estimates are largely sensitive to the selected $k$ and $L_0$ values (Aguilar-Virgen et al., 2014; Amini et al., 2013; Atabi et al., 2014; Machado et al., 2009; Peer et al., 1993).
However, no LFG literature could be found which applies and integrates sub-annual temporal variations into FOD models. An ideal starting point is thus to compare seasonal $k$ values to defaults and annual $k$ values for landfills in cold climates. While it is common to use LFG collection data and curve fitting to approximate “generation” constants, seasonal $k$ values will be defined as “collection” constants for this study.

1.3 CO$_2$ Modelling

The second part of the study concerns flawed assumptions within LandGEM's CO$_2$ estimates. The common focus in LFG studies is CH$_4$ due to its higher Global Warming Potential (25 CO$_2$e, 100-year horizon) (IPCC, 2007), utility in heating and electricity projects, and higher volatility than CO$_2$. This focus on CH$_4$ has resulted in multiple models of varying complexity which estimate only CH$_4$ generation (i.e. IPCC model) or LFG components as a function of CH$_4$ (i.e. Afvalzorg model, LandGEM model). A remaining issue with these generation models is that they are difficult to verify due to operational issues with collection systems, and differences between lab and field conditions. Assumed collection efficiencies are often used in order to approximate LFG generation from collection data (Amini et al., 2013; Maciel & Juca, 2011). Field collection efficiencies are highly dependent on cover thickness (Barlaz et al., 2009). This can be difficult to include in modelling since reporting standards vary over landfill lives, and reliable detailed records are often unavailable for onsite cover placement.

Table 1-2 summarizes the results of other studies on CH$_4$, CO$_2$, and trace gas ratios in LFG. The theoretical and lab based studies observed an average ratio between CH$_4$ and CO$_2$ of 1.2:1, while field studies yielded 1.4:1. Field gas data often showed that oxygen and nitrogen concentrations made up the bulk of trace gas concentrations, suggesting
minor to significant air intrusion (Aguilar-Virgen et al., 2014; Guter & Nuerenberg, 1987; Tolaymat et al., 2010) is a common issue in collection systems. Although $L_o$ is traditionally understood in terms of CH$_4$, it can be adapted for use in CO$_2$ modelling due to their similar substrates.

Early work on LFG production by Barlaz et al. (1989a) used an idealized chemical equation (Table 1-2) for anaerobic decay, and compared the methane potential of cellulose, hemicellulose, protein, and sugar. The study found that cellulose and hemicellulose accounted for 91.1% of methane potential in MSW, and their early work may have contributed to the broad assumption in modelling that the ratio between CH$_4$ and CO$_2$ is nearly 1:1 as per the ideal decay products, which is more appropriate when modelling LFG generation, as opposed to collection, emission, and oxidation. Barlaz et al. (1989a) cautioned that the lab conditions, including shredded waste, leachate recycling, and relatively homogeneous materials, may have overestimated CH$_4$ yields.

There are few LFG studies focused on CO$_2$. Several recent studies utilize LandGEM for various applications, such as cost analyses and model comparisons, and assume the 1:1 ratio in their models (Calabro et al., 2011; Chalvatzaki & Lazaridis, 2010; Goswami et al., 2011; Kumar & Sharma, 2014; Marroni et al., 2010; Rezaee et al., 2014). The assumption is used for simplicity despite expected ranges of 30 to 45% CO$_2$ (Abushammala et al., 2012; Ahmed et al, 2015; Saquing et al., 2014).
<table>
<thead>
<tr>
<th>Basis</th>
<th>Location</th>
<th>Notes</th>
<th>CH$_4$ : CO$_2$ : Trace Gas (% v/v)</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Empirical or Lab (at STP)</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Cellulose</td>
<td>C$<em>6$H$</em>{10}$O$_5$ + H$_2$O $\rightarrow$ 3CO$_2$ + 3CH$_4$</td>
<td>50.1 : 49.9 : 0</td>
<td>Barlaz et al. (1989a), Tchobanoglous &amp; Kreith (2002)</td>
<td></td>
</tr>
<tr>
<td>Protein</td>
<td>C$_2$H$<em>5$ON$</em>{0.86}$ $\rightarrow$ 1.548CO$_2$ + 1.653CH$_4$</td>
<td>51.7 : 48.2 : 0</td>
<td>Worrell &amp; Vesilind (2012)</td>
<td></td>
</tr>
<tr>
<td>Approx. Organic fraction in MSW</td>
<td>C$<em>{99}$H$</em>{149}$O$_{59}$N $\rightarrow$ 53CO$_2$ + 46CH$_4$ + NH$_3$</td>
<td>54.0 : 46.0 : 0 (Assumes 50% biodegradable)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Wisconsin (lab)</td>
<td>2-L reactors</td>
<td>Shredded refuse, leachate recycling (111 days old)</td>
<td>Barlaz et al. (1989b)</td>
<td></td>
</tr>
<tr>
<td>Kuala Langat, Malaysia</td>
<td>Flux chambers 15-30cm sand cover</td>
<td>Cell rec’d MSW from 1998-2008 (Total)</td>
<td>52.2 : 47.8 : 0</td>
<td>Abushammala et al. (2012)</td>
</tr>
<tr>
<td></td>
<td>Approx. Organic fraction in MSW</td>
<td>Assumed 0% trace gas, Reported molar, converted to STP</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Recife City, Brazil</td>
<td>5 vertical wells, passive ventilation</td>
<td>9m thick cell Reported as STP</td>
<td>54.3 : 40.7 : 5.0</td>
<td>Maciel &amp; Juca (2011)</td>
</tr>
<tr>
<td>Ensenada, Mexico</td>
<td>Sampling at well 21 days in fall, 3 readings/day</td>
<td>Well 3 was deepest, low intrusion Wells 1, 2, 4 had severe intrusion Varied temp.</td>
<td>55.1 : 44.4 : 0.5 (Well 3) 42.2 : 33.4 : 24.4 (Wells 1, 2, 4)</td>
<td>Aguilar-Virgen et al. (2014)</td>
</tr>
<tr>
<td>Winnipeg, Canada</td>
<td>Five 10m probes (to cell bottom)</td>
<td>15 samples biweekly for 1 year Varied temp. and pressure</td>
<td>56.1 : 40.1 : 3.8 (average)</td>
<td>Thompson &amp; Tanapat (2005)</td>
</tr>
<tr>
<td>Louisville, U.S.</td>
<td>1m clay-capped control cell Daily cover on bioreactor cell Vertical gas collection wells Unknown temp. and pressure</td>
<td>58 : 41 : 1 (cc, low change) 51 : 37 : 12 (bioreactor)</td>
<td>Tolaymat et al. (2010)</td>
<td></td>
</tr>
</tbody>
</table>
Leachate recirculation is used to increase LFG production and decay rates by increasing moisture and nutrient content in the waste mass, thereby increasing the value of energy production projects while decreasing the post-closure gas monitoring period (Reinhart, 1996). Landfills with these operations are called bioreactors. Recirculation is still a popular topic in the solid waste field, and its effect on modelling parameters is still under study (Karanjekar et al., 2015). The EPA’s estimated k for landfills with active recirculation is 0.7 year\(^{-1}\), much higher than the arid and conventional rates (0.02 and 0.04 year\(^{-1}\), respectively). \(L_o\) for bioreactor is 96 m\(^3\)/Mg, slightly lower than the conventional value (100 m\(^3\)/Mg), likely to account for decay which occurred prior to recirculation activities.

Calabro et al. (2011) studied LFG content data between 2004 and 2009 at a landfill in Italy. Significant error resulted due to overestimated methane generation; the ratio of methane to carbon dioxide shifted after introducing leachate recirculation, resulting in a 40:60 ratio for 2 of the 6 years in the study period, as opposed to the use of LandGEM’s basic assumption of 50:50. The cause was believed to be high sulphate content in the leachate favoring sulphate reducing bacteria growth over methanogen growth. They noted that their modified model could not predict significant shifts in LFG content resulting from leachate quality, and was instead limited to reactionary modelling.

Models using field CH\(_4\) content can also overestimate CO\(_2\) generation in curve fitting and forecast applications due to the governing formula being a function of CH\(_4\) content. Lower field CH\(_4\) content is observed at sites with significant air intrusion due to damaged collection lines (Guter & Nuerenberg, 1987) and shallow or permeable cover, which causes partial aerobic production of CO\(_2\) (Jeong et al., 2015), as well as increased
N\textsubscript{2} content. Also, clay covers in semi arid climates tend to desiccate and crack (Sadek et al., 2007), which may increase air intrusion. Both situations result in lowered CH\textsubscript{4} content and increased N\textsubscript{2}, the latter of which is currently erroneously reported as increased CO\textsubscript{2} under the default LandGEM assumption: CH\textsubscript{4} and CO\textsubscript{2} represent almost 100\% of LFG composition. These overestimations then serve as poor inputs into large-scale climate change models used by experts outside the solid waste industry. They may also affect forecasted estimates for upcoming carbon tax audits.

Poorly managed LFG collection systems may also lead to excess atmospheric intrusion as seen in 75\% of the vertical wells studied by Tolaymat et al. (2010). The freeze thaw cycle in cold climates can wear down final covers as well, increasing air intrusion. Hettiarachchi et al. (2013) reported condensate freezing in collection lines at a biocell in Calgary, a cold, western Canadian city. Condensate freezing led to system blockages, and reduced flow rates. As observed in their study, operators may try to mitigate the situation with reduced flow rates by increasing the applied vacuum from the blower system. This leads to intrusion, as the increased flow would not solve the blockage issue. Whether induced by freezing, overburden stress, or settlement, breaks in pipes and connections can occur and increase air intrusion into the collection system (Guter & Nuerenberg, 1987).

1.4 Objectives

1.4.1 Applying Seasonal k in Methane Modelling

The collected gas data in this study were available in daily values (recorded per minute from 2008-2014), thereby enabling a study for seasonal variance in LFG generation and
collection. By using the shorter time frame in each data point, the effects of temporal variability in LFG production became more significant. LandGEM is a common model used in LFG generation studies (Amini et al., 2012; Amini et al., 2013; Atabi et al., 2014; Scharff & Jacobs, 2006; Thompson et al., 2009), and is also widely used by engineers in North America. LandGEM was modified to output daily terms using linear interpolation, and to use different k values for different seasons. The study objectives for part 1 were as follows:

i. examine the effects of pseudo-second order iterative methods for back calculating k and L_0, and compare them to reported values in the cold climate literature,

ii. propose seasonal collection k values (k_winter and k_summer) for use in LandGEM in cold semi-arid climate,

iii. demonstrate the benefits of using seasonal sets of collection constants in LandGEM.

1.4.2 Alternative CO₂ Modelling Approaches

The three issues with respect to LFG CO₂ modelling are as follows: there is a significant lack of CO₂ LFG modelling studies; an unrealistic assumption of 1:1 CH₄ to CO₂ ratio in anaerobic landfill studies; and trace gas' (production or intrusion) effect on CO₂ estimates in LandGEM. To address these issues, the study objectives in part 2 were as follows:

i. compare the effects of trace gases on CO₂ collection estimates for five input approaches in LandGEM dependent and independent of CH₄ content, and establish a ranking by best fit
ii. compare five variations of LandGEM’s governing formula to determine effects on CO₂ model accuracy

iii. Test the applicability of two empirical k formulas to annual k modified LandGEM CO₂ modelling.

1.5 Organization of the Thesis

The Thesis has been organized under a traditional format wherein the chapters contain discussion of the aspects of both studies. Chapter one introduced the topic of landfill gas and current knowledge and assumptions in the literature, as well as the objectives of this study. Chapter two will present the methods and design of the two modelling studies. Chapter three will discuss the results and applications of the two studies, and suggest future applications and studies. Chapter four will summarize the major findings of the two studies.
CHAPTER 2: METHODOLOGY

2.1 Seasonal k Methods and Design

This study required compiling, sorting, and screening of 6-year LFG and waste disposal records, as well as climate data for the Regina's Fleet Street landfill, located in the northeast part of Regina. Gas data was used to compare and optimize a modified form of LandGEM using daily values and climate data derived seasonal sets. Three different pseudo-second order iterative methods were used for determining the site specific $k$ and $L_0$ values for Regina.

2.1.1 Regina Landfill

The data used in this part of the study was collected and processed from the City of Regina (COR), located in a cold, semi arid climate. The RE landfill opened in the first quarter of 1961, and Phase 1 closed in 2011. For this study, the opening date was assumed to be January 1, 1961. The LFG collection and flaring system was operational as of July 2008. The current LFG well field consists of 27 vertical wells at approximately 15m depth, and covers approximately 16 hectares on the north side of Phase 1, a 43 hectare site (Figure 2-1). The final cover on the north peak and side slopes was constructed in 2007, and composed of 1m of clay beneath 0.15m of vegetated topsoil. The waste mass elevation ranges from 600m at the base to 658m at the summit (Conestoga-Rovers & Associates, 2008). Average daily flow rates for the entire system ranged between 12,380 and 15,130 m$^3$/day. Average methane (48%) and carbon dioxide (40%) during the study period were at the lower end of the ranges reported in literature (45-60% and 40-60%, respectively). This is likely due to the young age of the landfill, and that the study period included both open and closed years.
Figure 2-1. LFG well field at the Regina Landfill (after Conestoga-Rovers & Associates, 2008)
The COR currently uses real-time measurement systems that record CH$_4$, CO$_2$, O$_2$ (all by % volume), and total gas flow. Gas composition is measured by Hitech sensors, which use the dual wavelength infrared technique for measuring methane and carbon dioxide. Gas flow is measured using a mass compensated Rosemount MassProBar flow meter. Per-minute gas data was available between August 2008 and December 2013, and daily CH$_4$ data in 2014. The data set was consolidated into 2,198 daily points. Data screening was carefully performed to prevent false data points from systematically altering the optimal k and $L_0$ values, and the resulting residual sum of squares (RSS). Removed methane concentration data points ranged from negative 100.0% to positive 26.8%. These false data likely originated from scheduled sampling, calibration, and maintenance work performed by the landfill crew. About 8.6% of the raw dataset was removed when more than 5 readings in a day recorded negative methane and/or carbon dioxide concentrations.

Waste mass data was compiled and processed from landfill disposal records from 1996 to 2011, Barlishen's report (1996) from 1981 to 1994, and an assumed waste disposal rate of 2.5 kg per capita together with census population data (Dominion Bureau of Statistics, 2015). The census data was increased by 8,900 for neighboring rural populations (Barlishen, 1996). The data used was consistent with a previous study by Opseth (1998). Opseth’s estimate for total waste in place by 1997 was 6.5-7.0 million Mg. The data used in this study up to 1997 (6.1 million Mg before the 38% scale was applied) was about 5.7% different compared to Opseth's lower estimate. The discrepancy may be explained by (i) Opseth including construction and demolition waste for all years, although stockpiling of such waste began in 1992; and (ii) Opseth's estimates for
1961 to 1980 were based on the difference between the total estimates and known data. All raw annual mass data provided was scaled down to match the representative area covered by the well field (approximately 38%, Figure 2-1). Composition data was based on a 12-month audit conducted by Canart and McMartin (2009) starting in October 2007, and used to calculate $L_o$ for the CO$_2$ modelling. The degradable organic carbon (DOC) in RE was calculated using a weighting ratio of 39:61 for the residential waste and industrial, commercial, and institutional waste samples collected in the audit. This additional measure was included to account for the low availability of waste composition data in RE. This ratio is based on disposal values surveyed in Saskatchewan in 2002 (Statistics Canada, 2004).

### 2.1.2 Seasonal Sets using Climate Trends

Regina’s cold, semi-arid climate tends to limit the potential moisture infiltration into the landfill, and freezing conditions further limit the infiltration to mostly summer months. The cold, dry winters are thus more likely to experience insufficient moisture content to support high activity for methanogenic microbes. In addition, more downtimes of the gas collection system are encountered with the longer period of sub-zero temperatures. In this part of the study, average climate data was taken from Environment Canada’s climate data archives (The Weather Network, 2015). Both temperature and precipitation data trends were used in order to determine appropriate boundaries between winter and summer conditions such that the daily data could be separated into two subsets for modelling. 6-month winter conditions (Maurice & Lagerkvist, 2003) and 5-month winter conditions were used in the present study. Given the well field size, well depths and extraction rates, the storage lag of the LFG was assumed to be minimal. Two seasonal
subsets, as opposed to four or more, were selected due to a limited theoretical basis. Observed sub-annual changes in LFG collection is limited to winter months (Hettiarachchi et al., 2013).

Ambient temperature data (Figure 2-2) was used to generate two seasonal sets. The first set was denoted “5-month Winter”, with the subset “Winter 1” using data from the 5-month span where the average temperature was below 0°C. Its corresponding summer period, denoted “Summer 1”, uses data from the remaining 7 months. The second set uses equal-length periods between the subsets, and is denoted as “6-month Temp. Winter”. The periods and number of data points for the 4 seasonal sets of data are summarized in Table 2-1. In addition, precipitation data was used to generate two more seasonal sets because moisture, and thus precipitation, is the dominant factor in landfill gas generation (Albanna et al., 2007; Environment Canada, 2014; Maurice & Lagerkvist, 1997; Thompson et al., 2009). Fan et al. (2006) noted that in Taiwan, autumn and winter are considered the dry seasons with respect to leachate production. Thus for seasonal set “6-month Prec. Winter”, subset “Winter 3” is set between October and March because less rain and snow fall in October than in April in Regina.
Figure 2-2. Average monthly precipitation and temperature in Regina, 1984 – 2014
Table 2-1. Seasonal sets used in modified LandGEM modelling

<table>
<thead>
<tr>
<th>Seasonal set</th>
<th>Subsets</th>
<th>Period starts</th>
<th>Period ends</th>
<th>Data points</th>
</tr>
</thead>
<tbody>
<tr>
<td>5-month Winter</td>
<td>Winter 1</td>
<td>November 1</td>
<td>March 31</td>
<td>745</td>
</tr>
<tr>
<td></td>
<td>Winter 2</td>
<td>October 16</td>
<td>April 15</td>
<td>931</td>
</tr>
<tr>
<td></td>
<td>Winter 3</td>
<td>October 1</td>
<td>March 31</td>
<td>942</td>
</tr>
<tr>
<td></td>
<td>Fall 4</td>
<td>July 1</td>
<td>December 31</td>
<td>1,017</td>
</tr>
<tr>
<td></td>
<td>Spring 4</td>
<td>January 1</td>
<td>June 30</td>
<td>992</td>
</tr>
<tr>
<td></td>
<td>N/A</td>
<td>January 1</td>
<td>December 31</td>
<td>2,009</td>
</tr>
<tr>
<td>6-month Temp.</td>
<td>Summer 1</td>
<td>April 1</td>
<td>October 31</td>
<td>1,264</td>
</tr>
<tr>
<td>Winter</td>
<td>Summer 2</td>
<td>April 16</td>
<td>October 15</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Summer 3</td>
<td>April 1</td>
<td>September 30</td>
<td></td>
</tr>
<tr>
<td>6-Month Prec.</td>
<td>Summer 4</td>
<td>January 1</td>
<td>June 30</td>
<td></td>
</tr>
<tr>
<td>Winter</td>
<td>Prec. Control</td>
<td>December 31</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>


Snow data was not converted into rain equivalents because it was assumed that most snowfall on the landfill slope and summit would be subjected to wind drift and snow melt runoff due to frozen unvegetated soil cover (Orradottir, 2002); however, Environment Canada's National Inventory Report (2014) on GHG sources and sinks used a snow-to-rainwater ratio of 10:1 in order to include snow in its k estimates. In order to test whether winter and summer based seasonal sets led to more accurate results, an opposing set was developed. The “Prec. Control” set, with subsets denoted “Fall 4” and “Spring 4”, were chosen on opposing sides of the average peak rainfall month (June, Figure 2-2).

2.1.3 LandGEM Modification

This part of the study covers 6 years of LFG collection operations, similar to the coverage in studies by Machado et al. (2009) and Tolaymat et al. (2010). LandGEM is designed to output annual methane generation rates (see eq. 1, Table 2-2). LandGEM's annual methane generation rates were converted into daily data points using linear interpolation (eq. 2) to compare to the consolidated daily data records. This model assumes LandGEM's quoted annual flow occurs on December 31, similar to the triangle method (Kumar et al., 2004), wherein LFG generation continually increases to the lifetime peak, then continually decreases. One benefit of this method is that flow rates are continuous at the start and end of the year, as opposed to the discontinuity which would result from applying average values each year.
Table 2-2. LandGEM and the linear-interpolated model

<table>
<thead>
<tr>
<th>Equation No.</th>
<th>Formula</th>
<th>Terms</th>
<th>Source</th>
</tr>
</thead>
</table>
| 1 | $Q_{CH4} = \sum_{i=1}^{n} \sum_{j=0.1}^{1} k L_o \frac{M_i}{10} e^{-kt_{ij}}$ | $Q_{CH4}$ = CH$_4$ generation rate (m$^3$/year)  
$i = 1$ year time increment  
$n = \text{(year of calculation)} - \text{(initial year of waste acceptance)}$  
$j = 0.1$ year time increment  
$k = \text{methane generation rate (year$^{-1}$)}$  
$L_o = \text{potential methane generation (m$^3$/Mg)}$  
$M_i = \text{mass of waste accepted in the $i^{th}$ year (Mg)}$  
$t_{ij} = \text{age of the $j^{th}$ section of waste mass $M_i$ accepted in the $i^{th}$ year (decimal years)}$ | LandGEM |
| 2 | $Q_{mCH4} = \frac{[Q_i + (t - t_i) \cdot \frac{Q_{i+1} - Q_i}{t_{i+1} - t_i}] \cdot \frac{1 \text{ year}}{525600 \text{ min}}}{1 \text{ year}}$ | $Q_{mCH4}$ = CH$_4$ flow rate (m$^3$/min)  
$Q_i = \text{CH}_4$ flow rate (m$^3$/year) output by LandGEM  
i = \text{Year of the data point}$  
t = \text{days since the landfill opened}$  
t_i = \text{days since landfill opened at December 31 of the annual}$ | Modified, this study |
2.1.4 Iterative Method and Initial Values

Pseudo-second order iterative method was used in the optimization of the LandGEM parameters. The variable optimization processes were repeated until approximate errors were lower than a specified threshold (0.5% in this part of the study) for all variables.

Three pseudo-second order methods were used for each subset in this part of the study (Figure 2-3) by minimizing the residual sum of squares (RSS) between the modeled and collected data. Excel's Solver function was used to minimize the RSS between the data sets. Methods and values were then evaluated relative to each other. The “k first” and “L₀ first” methods notably only required four solver runs to yield approximate errors less than 0.5%.

The initial k value was set to the EPA default 0.020 year⁻¹ for arid landfills, which was also close to the value (0.023 year⁻¹) using the formula reported by Thompson et al. (2009) (Section 2.2.2). Based on the range of values in Table 1-1, a conservative initial value (100 m³/Mg) was selected for L₀. Opseth (1998) also used this value instead of the 170 m³/Mg default in LAAEM (precursor to LandGEM) in order to increase the accuracy of emission estimates.
Figure 2-3. Pseudo-second order iterative Solver methods used in part 1

- Change k and $L_o$
  - Repeat until Approximate Error < 0.5%

- Change $L_o$ first
  - Fix k
  - Change k
  - Fix $L_o$
  - Repeat until Approximate Error < 0.5%

- Change k first
  - Fix $L_o$
  - Change $L_o$
  - Fix k
  - Repeat until Approximate Error < 0.5%
Microsoft Excel's Solver function was used to minimize the RSS for the modelled methane. The Solver function changes one or more cells in order to optimize one other related cell. Certain constraints can be set such as boundary values, although this was not used in this study. The following conditions were used:

- Max solving time was set to 100 seconds
- Iterations were set to 5000 to ensure the solver would not require multiple runs
- Precision was set to 0.000001 (default)
- Tangent Estimates (default)
- Central Derivatives for lower error (k-first, and L₀-first methods)
- Forward derivatives to avoid divergence (Change k & L₀ together method)
- Newton Search (default)

Along with the high iteration setting, the approximate error was set to a low value of 0.5% in order to reasonably assume convergence using the optimization methods.

2.2 CO₂ Modelling Approaches and Sites

Part 2 of the study compiled and calculated data at four western Canadian landfills. Four approaches to CO₂ FOD modelling were compared to the existing approach, and are as follows:

i)  Default LandGEM assumption (Total LFG = CH₄ + CO₂) with field CH₄ content (reduced by high trace gases) (denoted “Default with Traces”)

ii) Default LandGEM assumption using adjusted CH₄ content (without trace gases) (denoted “Default without Traces”)

iii) Optimized k and L₀ values (denoted “Optimized”)

iv) CH$_4$ to CO$_2$ L$_{o}$ ratio of 1.2 (from lab studies) (denoted “1.2 Ratio”)

v) L$_{o}$ ratio of 1.4 (from field studies and the four study sites) (denoted “1.4 Ratio”)

The general methodology followed for determining the various input values (k, L$_{o}$, mass disposal) is displayed in Figure 2-4 to simplify replication of the study results.

### 2.2.1 Site Conditions and Characteristics

Three other landfills in western Canada with active LFG collection systems were studied in addition to Regina: Cache Creek (denoted CC) in British Columbia (B.C.), Saskatoon (denoted SK) in Saskatchewan, and Hartland (denoted HL) in B.C. Seasonal analysis was not performed for these sites because only monthly (SK) and annual (CC, HL) data were available.

All three sites, contrary to Regina, were active by the end of the study periods. A variety of site conditions exist at the sites. Three of the four sites tended to collect significant amounts of residual gases (Table 2-4). SK had the lowest range (0.4 to 6.5 % monthly), and HL had the highest (7.8 to 28.2%). CC (7 to 13%) and RE (5.8 to 19.8) were similarly high, increasing the uncertainty when using average collected CH$_4$ content in LandGEM. To verify this, LandGEM was run using average measured CH$_4$ content for each site (denoted “Default with Traces”), and compared to results which assumed total collected LFG was the sum of methane and CO$_2$ (denoted “Default w/o Traces”).
1. Collect and process available landfill waste disposal records
   Objective: $M_i, t_{ij}$ values

   Determine mass disposal (Mg, tonnes) within the wellfield’s area of effect
   Note when separation of certain materials begin

   Estimate missing waste records based on:
   (i) per capita disposal rates and population;
   (ii) known total tonnage

2. Collect and process disposed waste composition data
   Objective: $L_o$ value(s)

   Determine the proportions of wastes such as:
   Wood, grass clippings, leaves, food, pet waste,
   carcasses, nappies, textiles, and paper products

   (i) Calculate DOC using equation 6
   (ii) Calculate $L_o$ using equation 5 and appropriate
        estimates for parameters
   (iii) Consider a range of $L_o$ depending on available
        data and assumptions

3. Collect and process climate data, and/or reported $k$ for similar climates
   Objective: $k$ value(s)

   Using precipitation data, determine a range of $k$
   values to use in order to identify and reduce
   starting value bias

   (i) Consider reported $k$ if available
   (ii) Calculate $k$ via available empirical formulas
   (iii) Use suggested EPA defaults based on climate

Figure 2-4. General methodology for determining input values
Figure 2-5. Average landfill gas composition for four western Canadian sites
Waste disposal data was adjusted when available to account for the well field coverage. Average L₀ values were calculated when multiple waste composition reports were available. Gas data was gathered, and verified via annual site reports. The collection values were adjusted via quoted collection efficiencies, or assumed values based on available cover data. In CC, RE, and HL, the sites provided one more annual data point for methane than CO₂. No attempts were made to interpolate the absent CO₂ values via recorded total LFG data.

2.2.1.1 Cache Creek Landfill

The Cache Creek (CC) landfill opened in 1989, and accepts much of its waste from the Metro Vancouver region; only 3% of the waste disposed in 2014 came from other sources (Golder Associates, 2015). The waste mass is disposed in the side of a valley, and the refuse mass ranges from 502 to 675m elevation. Phases 1 to 4 cover 49.6 hectares, and the newest phase 5 covers 6.7 hectares. Phases 1-3 were topped with final cover, while approximately half of phase 4 had been covered as of 2014 (Golder Associates, 2015). The well field consisted of 42 wells as of 2005, and additional wells were added nearly each year during the study period to a final count of 154 wells (vertical and horizontal). Final cover is specified in the operation license as 1 m low permeability soil, and 15 cm topsoil with vegetation. Leachate recirculation was performed between 2005 and 2009. Leachate was applied to the surface cover to facilitate partial evaporation.

Annual waste disposal data was available in reports starting in 2005 and historical data was included in a 2010 report (AECOM, 2010). Composition data was gathered via
Metro Vancouver waste audit reports in 2010, 2011, 2013, and 2015. The resulting degradable organic carbon (DOC) values were averaged to calculate $L_0$.

Gas data was available in annual reports between 2005 and 2015 (no CO$_2$ data in 2009). Average methane and carbon dioxide for the study period were 53.6 and 36.9%, respectively. Due to well field expansions, the average daily flow rates ranged from 18,300 to 65,500 m$^3$/day. Gas collection efficiencies were reported each year between 2011 and 2015 (ranging 65 – 87%), and a conservative estimate of 50% was assumed between 2005 and 2010 (Ahmed et al, 2015; Spokas et al., 2006).

2.2.1.2 Saskatoon Landfill

The Saskatoon (SK) landfill opened in 1955 and is still undergoing disposal operations between old and new cells. Saskatoon is located in a cold, semi arid climate. The 29 vertical wells in the well field are spread along the central and northern sections of old cells, which ceased disposal operations between 2004 and 2010. Further disposal occurred in 2010 and 2013, with minimal disposal in 2015. 69 wells are required for overlapping horizontal coverage of the entire waste mass. A collection efficiency of 70% was used as the design basis report expected final cover to be constructed soon after the well field (Comcor Environmental Ltd., 2010). The old cells cover 27 hectares, and the expansion covers 9 hectares.

Waste data for the landfill's old and new cells (1955-2015) were collected and corroborated by annual reports. Composition data was processed from waste audits conducted in 2006, 2013, and 2014. Gas measurements were taken on average 2.3 days per month between January 2014 and December 2015, with some missing months. Each
well was sampled an average of once per month for methane, CO$_2$, oxygen, and LFG flow. Average flow rates ranged between 15,740 and 20,388 m$^3$/day, and the average methane and CO$_2$ for the study period were 57.7 and 40.1%, respectively.

2.2.1.3 Hartland Landfill

Hartland (HL) was privately owned and operated from the early 1950s (assumed 1951 in the present study) until 1985. 33 vertical LFG collection wells were installed in 1990. By 2013, the well field in the closed and open cells expanded to a total of 115 wells (73 vertical, 26 horizontal, 16 others), which was reflected in the increased collection efficiencies reported (range of 28.5 – 80.8%). This coastal site currently covers 36 hectares, with a projected final footprint of 46 hectares. Final cover consists of a geomembrane and clay cap, and interim covers are 0.5 m clay. The waste mass ranges from approximately 135 m at the base to 175 m at the summit. A leachate recirculation pilot project was initiated in 2002, but as of 2013, there were no full-scale bioreactor cells on site.

Waste disposal data from annual reports were limited to the period of 1980 to 2013. Estimates for the remaining period were based on an estimated waste in place value of 6.3 million Mg at the end of 2011 (Fillipone et al., 2012), and an exponential relationship derived using waste trends between 1980 and 1989 ($R^2 = 0.865$, data not shown). Composition data was available for five years between 1990 and 2010. Average methane and CO$_2$ for the study period were 48.1 and 34.0%, respectively, both lower than expected. Annual oxygen averages did not exceed 1% throughout the study period. Due to expansions in the well field, average daily flow rates ranged from 19,700 to 48,200 m$^3$/day.
2.2.2 Formulas for k and L₀

Some studies have supported formulas relating moisture content and the decay rate k in FOD models (Environment Canada, 2014; McDougall & Pyrah, 1999; Thompson et al., 2009). Thompson et al. (2009) used equation 3, developed by the Research Triangle Institute (RTI) based on U.S. EPA default k values, in their study:

\[ k = 3.2 \cdot 10^{-5} \cdot (x) + 0.01 \]  (3)

where: \( k \) = decay rate (year\(^{-1}\)); and \( x \) = annual average precipitation (mm). Environment Canada (EC) (2014) instead supported the use of a similar equation developed by the RTI:

\[ k = 7 \cdot 10^{-5} \cdot (x) - 0.0172 \]  (4)

The two equations were used to calculate annual k values with precipitation data gathered from Environment Canada’s weather archives. When main weather stations were discontinued, then the nearest station was used (HL in 2007, CC in 2015). All k’s were positive except for two years in CC using eq. 4 (2009 and 2015). The two negative k values were arbitrarily set to a low value (0.00001 year\(^{-1}\)) to use a positive number while representing the formula’s low estimate. The equations yielded a wide range of k’s (eq. 3: 0.016 to 0.048; eq. 4: 0.002 to 0.067 year\(^{-1}\)).

The IPCC provided a common formula for calculating L₀ from waste audit data, which has since been used in some studies (IPCC, 1996; Thompson & Tanapat, 2005; Aguilar-Virgen et al., 2014; Environment Canada, 2014):

\[ L₀ = F \cdot MCF \cdot DOC \cdot DOC_f \cdot 16/12 \]  (5)

where:
- $L_0$ [Mg CH$_4$/Mg of waste]
- $F =$ average CH$_4$ content in LFG [fraction]
- $MCF =$ CH$_4$ correction factor [1.0 for maintained landfills]
- $DOC_f = 0.50$ or $0.77$ [fraction, 2006 and 1996 defaults]
- $16/12 =$ molecular mass conversion factor [Mg CH$_4$/Mg C]
- $DOC =$ degradable organic carbon [Mg C/Mg of waste]

The DOC component is the variable based on available waste composition data:

$$DOC = (0.4 \cdot A) + (0.17 \cdot B) + (0.15 \cdot C) + (0.3 \cdot D)$$  \hspace{1cm} (6)

where:

- $A =$ paper and textiles (Mg/Mg of total sample)
- $B =$ garden and park waste (Mg/Mg of total sample)
- $C =$ food waste (Mg/Mg of total sample)
- $D =$ wood or straw (Mg/Mg of total sample)

Equation 3 was used with $DOC_f$ values of 0.50 and 0.77 as lower and upper bounds, respectively (Thompson et al., 2009). Research has been conducted to find more appropriate $DOC_f$ values based on site factors such as average volatile lignin content (Thompson & Tanapat, 2005), but such data was unavailable for the four sites. Table 2-3 summarizes calculated $k$ and $L_0$ data for the four sites, including the $L_0$-CO2 values used in the “Ratio” approaches (section 2.2.4).
Table 2-3. Site-specific data and values for CO$_2$ LandGEM modelling

<table>
<thead>
<tr>
<th>Site</th>
<th>Ave Prec. (mm/yr)</th>
<th>CH$_4$ $L_o$</th>
<th>L.4 Ratio $L_o$</th>
<th>L.2 Ratio $L_o$</th>
<th>$k$ (yr$^{-1}$)</th>
<th>CO$_2$ data (years)</th>
</tr>
</thead>
<tbody>
<tr>
<td>CC</td>
<td>328</td>
<td>82.8</td>
<td>127.6</td>
<td>59.2</td>
<td>91.1</td>
<td>69.0</td>
</tr>
<tr>
<td>SK</td>
<td>358</td>
<td>97.4</td>
<td>150.0</td>
<td>69.6</td>
<td>107.1</td>
<td>81.1</td>
</tr>
<tr>
<td>RE</td>
<td>411</td>
<td>86.2</td>
<td>132.7</td>
<td>61.6</td>
<td>94.8</td>
<td>71.8</td>
</tr>
<tr>
<td>HL</td>
<td>886</td>
<td>83.3</td>
<td>128.3</td>
<td>59.5</td>
<td>91.7</td>
<td>69.5</td>
</tr>
</tbody>
</table>

a. CH$_4$ $L_o$ used as starting values in the optimization algorithms to determine ideal $k$. 


2.2.3 Modified LandGEM formulas and Optimization procedures

Two groups of variations on the existing FOD, Scholl Canyon type formula were generated for study: Subsections, and Annual k’s (Table 2-4). The subsections group differ only by the time increment selected where j=1, 0.5, 0.25, and 0.1 (Thompson et al., 2009).

Annual k’s are used in place of the lifetime k values in order to account for annual differences in infiltration, using precipitation rates as an approximation. The same time increment “j” was kept as the current LandGEM formula to simplify comparison. It was assumed that k would be equal for both CH$_4$ and CO$_2$ given that it represents the rate of biodegradation, and not compound-specific measures. This assumption exists in the current LandGEM model. The non-optimized approaches (“Defaults w/ Traces”, “Default w/o Traces”, “1.2 Ratio”, and “1.4 Ratio”) tested the utility of the k equations (3 and 4) in simple, non-optimized annual k LandGEM.
Table 2-4. Default and Modified LandGEM Carbon Dioxide modelling formulas

<table>
<thead>
<tr>
<th>Formula Modification</th>
<th>Derivation (Names)</th>
<th>Formula</th>
<th>Terms</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Subsection</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>LandGEM 3.02 (Sub-10)</td>
<td></td>
<td>$Q_{CO2} = \sum_{i=1}^{n} \sum_{j=0.1}^{1} kL_{o} M_{i} e^{-kt_{ij}}$</td>
<td>$Q_{CO2}$ = CO$_2$ generation rate (m$^3$/yr)</td>
</tr>
<tr>
<td></td>
<td>Thompson et al. (2009) (Sub-4, Sub-2)</td>
<td>$Q_{CO2} = \sum_{i=1}^{n} \sum_{j=0.25}^{1} kL_{o} M_{i} e^{-kt_{ij}}$</td>
<td>$Q_{CO2}$ = CO$_2$ flow rate (m$^3$/yr)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>LandGEM 2.01 (Sub-1)</td>
<td></td>
<td>$Q_{CO2} = \sum_{i=1}^{n} kL_{o} M_{i} e^{-kt_{i}}$</td>
<td>$Q_{CO2}$ = CO$_2$ flow rate (m$^3$/yr)</td>
</tr>
<tr>
<td></td>
<td>This study (Annual k)</td>
<td>$Q_{CO2} = \sum_{i=1}^{n} \sum_{j=0.1}^{1} kL_{o} M_{i} e^{-kt_{ij}}$</td>
<td>$Q_{CO2}$ = CO$_2$ flow rate (m$^3$/yr)</td>
</tr>
<tr>
<td><strong>Current</strong></td>
<td>LandGEM 3.02 (Default)</td>
<td>$Q_{CO2} = Q_{CH4} \cdot \left(100 \cdot \frac{1}{P_{CH4}} - 1\right)$</td>
<td>$Q_{CO2}$ = CO$_2$ flow rate (m$^3$/yr)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

- $Q_{CO2}$ = CO$_2$ generation rate (m$^3$/yr)
- $i$ = 1 year time increment
- $n$ = (year of calculation) - (initial year of waste acceptance)
- $j$ = 0.1 year time increment
- $k$ = methane generation rate (yr$^{-1}$)
- $L_{o}$ = potential CO$_2$ generation (m$^3$/Mg)
- $M_{i}$ = mass of waste accepted in the $i^{th}$ year (Mg)
- $t_{ij}$ = age of the $j^{th}$ section of waste mass $M_{i}$ accepted in the $i^{th}$ year (decimal years)
- $Q_{CH4}$ = CO$_2$ flow rate (m$^3$/yr)
- $P_{CH4}$ = Methane content (%)
The algorithms used to determine optimized parameters in part 2 (Figure 2-6) minimized the residual sums of squares between model estimates and collected data using the Solver function in Excel. Multiple starting values were used to reduce starting value bias in optimized values. CH₄ data was used to calculate optimized k’s for two reasons: (i) site-specific Lₒ values were available, and (ii) all sites except SK had more data for CH₄ than CO₂, thus improving the reliability of optimized k. Method A used two starting sets to determine preferred starting k and DOCᵣ values for each site. Method B used one starting set, as the dual cell change algorithm was unaffected by starting values.

Methods C and D used four starting value sets. The approaches differed in which variable (k, Lₒ) was changed first to identify the more accurate algorithm. Methods B and D were subject to boundary constraints given difficulties with divergence in the literature (Amini et al., 2012), CH₄ Lₒ values were bound between 90% of Lₒ (0.50) and 110% of Lₒ (0.77), while k values were conservatively bound between 0.0005 and 0.4 year⁻¹, values appropriate for more extreme sites and climates (or leachate recirculation operations) than those in this study. For example, Environment Canada (2014) reported k’s between 0.001 and 0.199 year⁻¹ in Whitehorse, Yukon and Prince Rupert, B.C., respectively.
Figure 2-6. RSS optimization modelling algorithms for CO₂ modelling in part 2
2.2.4 Simple Ratios for $L_o$-CO$_2$

Jeong et al. (2015) defined anaerobic landfills as those with CH$_4$ to CO$_2$ ratios between 1.0 and 2.0, while semi-aerobic landfills are below 1.0. The four sites in this study, however, were designed and operated as anaerobic landfills.

The four sites’ ratios of CH$_4$ to CO$_2$ were similar save for RE’s (1.20:1), which was closer to the average predicted from empirical/lab methods (Table 1-2), and suggestive of aerobic activity. CC (1.45:1), SK (1.44:1), RE (1.20:1), and HL (1.42:1) had an average ratio of 1.38:1, a result close to the average field study ratio (1.4 from Table 1-2) in the literature. The formulas and starting values in section 2.2 were thus repeated using conservative ratios of 1.2 and 1.4 in order to determine whether these simple, non-optimized approximations yielded CO$_2$ estimates more accurate than raw content LandGEM estimates. The method thus consists of scaling down known $L_o$ to produce a new term, $L_o$-CO$_2$, which can be input into LandGEM formulas.
CHAPTER 3: RESULTS & DISCUSSION

3.1 Seasonal k CH₄ Modelling at the RE Landfill

The five datasets in part 1 of this study were compared using resulting RSS and results from previously published studies. The average methane content for the screened dataset during the study period was 48%. The model results for the default k and Lₒ terms were not included in any graphs because their mean percent error was 76.5%, consistently overestimating the actual RE data throughout the study. For example, the model outputs 3,225,000 m³/year using LandGEM defaults in 2010, while the actual data totals 1,840,000 m³/year (75.3% higher than field data). As such, the range of EPA default values is found not applicable for Regina’s landfill. It is hypothesized that the discrepancy is partly due to the differences in climatic condition between average U.S. landfills and the RE landfills. Comparisons between U.S. and Canadian LFG production is recommended for future studies.

3.1.1 Comparing Pseudo-Second Order Iterative Methods

Three iterative methods were run for all seasonal sets in part 1, and produced a wide range of k and Lₒ values. To identify the most appropriate method for future studies, the methods were compared on a dual basis of reported k and Lₒ values in the literature, and accuracy as represented by RSS values.

3.1.1.1 Change k & Lₒ

A few consistent observations occur across all the datasets (Table 3-1). The first is that the k and Lₒ results from the “Change k and Lₒ” iterative method are well outside the values reported in the literature for the winter, fall, and spring subsets. For example, the
for Winter 1 (0.0006 year\(^{-1}\)) is 9 times lower than the lowest value reported by Karanjekar et al. (2015), where the lowest reported k was 0.0054 year\(^{-1}\) in Israel. Winter 1’s corresponding L\(_o\) value (1226.3 m\(^3\)/Mg) is more than 5 times higher than the highest L\(_o\) value (220 m\(^3\)/Mg, for Quebec) reported by Thompson et al. (2009). The findings are consistent with Amini et al. (2012), who noted a similar result using an approach which calculated k and L\(_o\) at the same time. One of the five landfills calculated a L\(_o\) value of 3,844 m\(^3\)/Mg, the second highest value being 175 m\(^3\)/Mg using the same method. By comparison, Ishii and Furuichi (2013) measured L\(_o\) for different wastes and reported that paper has the highest L\(_o\) (214.4 m\(^3\)/Mg in their study). Subsets Summer 3 (46.2 m\(^3\)/Mg) and Spring 4 (38.3 m\(^3\)/Mg) each had L\(_o\) values drop by more than 18% below the EPA’s recommended boundary, although Karanjekar et al. (2015) observed even smaller L\(_o\) in 2 of their 27 laboratory reactors.
Table 3-1. Optimal k and $L_o$ from minimizing RSS using the Excel Solver

<table>
<thead>
<tr>
<th>Seasonal Set</th>
<th>Sub Set</th>
<th>Change k and $L_o$</th>
<th>Change $L_0$ first</th>
<th>Change k first</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>k (yr$^{-1}$)</td>
<td>$L_o$ (m$^3$/Mg)</td>
<td>RSS</td>
</tr>
<tr>
<td>Winter</td>
<td>Winter 1</td>
<td>0.0006</td>
<td>1,226.3</td>
<td>889</td>
</tr>
<tr>
<td></td>
<td>Summer 1</td>
<td>0.0164</td>
<td>67.6</td>
<td>652</td>
</tr>
<tr>
<td></td>
<td>Average</td>
<td>0.0085</td>
<td>647.0</td>
<td>770</td>
</tr>
<tr>
<td>Temp. Winter</td>
<td>Winter 2</td>
<td>0.0005</td>
<td>1,301.9</td>
<td>1,037</td>
</tr>
<tr>
<td></td>
<td>Summer 2</td>
<td>0.0209</td>
<td>58.6</td>
<td>512</td>
</tr>
<tr>
<td></td>
<td>Average</td>
<td>0.0107</td>
<td>680.3</td>
<td>775</td>
</tr>
<tr>
<td>Prec. Winter</td>
<td>Winter 3</td>
<td>0.0004</td>
<td>1,642.9</td>
<td>1,169</td>
</tr>
<tr>
<td></td>
<td>Summer 3</td>
<td>0.0346</td>
<td>46.2</td>
<td>382</td>
</tr>
<tr>
<td></td>
<td>Average</td>
<td>0.0175</td>
<td>844.6</td>
<td>775</td>
</tr>
<tr>
<td>Control</td>
<td>Fall 4</td>
<td>0.0010</td>
<td>806.0</td>
<td>894</td>
</tr>
<tr>
<td></td>
<td>Spring 4</td>
<td>0.1689</td>
<td>38.3</td>
<td>703</td>
</tr>
<tr>
<td></td>
<td>Average</td>
<td>0.0849</td>
<td>422.2</td>
<td>799</td>
</tr>
<tr>
<td></td>
<td>Full Set</td>
<td>0.0015</td>
<td>512.6</td>
<td>1,653</td>
</tr>
</tbody>
</table>
The resulting optimal values are thus considered invalid, in spite of the RSS values. The total RSS in this study ranged between 1,540 for the 5-month Winter set, and 1,653 for the Full Set (about 7.3% higher).

3.1.1.2 \( L_o \) first method

The \( L_o \) first method leads to reasonable k values compared to Thompson et al. (2009), although they did not deviate from the starting value by more than 0.5% in any subset. There is a lack of cold, semi-arid climate field and modelling studies to compare k values with; Thompson et al. (2009) and Environment Canada (2014) both use an empirical formula to determine k.

Similar to the “Change k and \( L_o \)” method, the “\( L_o \) first method” tended to calculate \( L_o \) values outside the recommended range set by the EPA. Tolaymat et al. (2010) also calculated values below the recommended range, although they used waste sampling to determine \( L_o \) rather than numerical modelling. The range of calculated \( L_o \) values in other studies (Amini et al., 2012; Karanjekar et al., 2015; Tolaymat et al., 2010) were within 5m³/Mg of the average \( L_o \) in this set, suggesting the EPA’s recommended range is not reliable under certain conditions. RSS for the “\( L_o \) first method” tended to be the highest of all three methods. In addition, \( L_o \) by definition does not change in a seasonal (or periodic) pattern, but rather steadily decreases as the waste mass decomposes as shown by Ishii and Furuichi (2013). Moreover, Thompson et al. (2009) noted that diversion programs affect landfill composition and thus \( L_o \). While this could apply to some landfills, it is unlikely that RE’s landfill composition changed appreciably during the study since it began diverting construction and demolition waste as late as 1994. The
next large change in the COR’s waste management program did not occur until 2013, when curb side recycling was implemented. Furthermore, Wang et al. (2016) noted that Saskatchewan's per capita diversion was much lower than the national average between 1998 and 2010.

The results of the $L_o$ first method are thus not applicable due to the method itself; additional steps and complexity are required to account for $L_o$’s theoretical basis, thus decreasing the method’s ease of use.

The total RSS ranged from 1,563 for the 5-month Winter set to 1,669 for the Full Set (about 6.7% higher). The results from changing $L_o$ first are thus considered less reliable than the results of changing $k$ first (section 3.1.3). $L_o$ depends primarily on waste composition, which is not subject to change nearly as much as precipitation and weather conditions during the year.

3.1.1.3 $k$ first method

The starting default values overestimated the methane generation, leading the first changed variable to scale down to approximate the recorded data. For the “$L_o$ first method”, $L_o$ decreased by an average of 42.8%, while the “$k$ first method” decreased $k$ by an average 55.1% from the starting value. The RSS range for the “$k$ first method” is more statistically significant than in the “$L_o$ first method”, and more reasonable from a theoretical basis than the results of changing $k$ and $L_o$ together. In Table 3-1, all $k_{\text{winter}}$ values are lower than their corresponding $k_{\text{summer}}$. On average, $k_{\text{winter}}$ values were about 12.6% less than their counterparts. The $k_{\text{winter}}$ values tend to be smaller than the Full set's $k$ values as well. This is probably due to (i) reduced moisture infiltration during the
winter, resulting in less LFG generation; and (ii) higher downtimes and lower collection efficiency during winter due to frozen wellheads and other condensate issues. The summer values ($k_{\text{summer}}$) were higher than the Full Set because they were not weighted down by low LFG output in winter months. Results suggest the use of seasonal collection values $k_{\text{winter}}$ and $k_{\text{summer}}$ may provide more accurate results by better representing the field conditions in cold, semi-arid climates.

None of the resulting $k$ values from “change k first” set were close to estimates by Thompson et al. (2009) at about 0.023 year$^{-1}$ based on the precipitation data alone. However, the values reported by Environment Canada (2014) and Opseth (1998) were comparable (0.006 - 0.011 year$^{-1}$). Other $k$ values in Table 1-1 were from more temperate or wet climates, so it was expected that they would be higher than the results in the present study.

The optimal $L_o$ values remained close to the starting default (100 m$^3$/Mg) for all 5 sets, indicating the significant impact the first changed variable has on the second using this iterative method. $L_o$ was selected as a conservative estimate from literature. The $L_o$ from all 5 sets was comparable to Alberta (100 - 178 m$^3$/Mg) and Ontario (90 - 160 m$^3$/Mg) (Thompson et al., 2009). The algorithm produced $L_o$ values with at most 0.1% changes from the starting values. The results reported in Table 3-1 come from only changing $k$ in order to represent the non-seasonal characteristic of $L_o$ (i.e. $L_o$ was not changed) RSS decreased minimally (at most 0.01) after the first step in all sets, meaning subsequent steps were useful only to ensure approximate errors were low and convergent.
Figure 3-1. Relationship between optimized k and $L_o$ using k first method

$k = -0.026L_o^2 + 5.129L_o - 256.2$

$R^2 = 0.60$
The optimized results for each seasonal subset and iterative method yield determined high k values paired with lower L₀ values, and vice versa. This is consistent with the results in Tolaymat et al. (2010). Figure 3-1 shows the slightly negative second order polynomial relationship between optimized k and L₀ for the RE landfill using k first method. Tolaymat et al. (2010) instead generated an inverse relationship between k and L₀ for a larger range of L₀ values (25 - 100 m³/Mg). The difference in observed ranges is due to using k first optimized values in Figure 3-1, while Tolaymat et al. (2010) fit k values based on several different starting L₀ values.

The values from the “k first method” stayed within ranges consistent with the literature for both k and L₀ for all subsets. It also consistently resulted in lower RSSs than the L₀ first method, although only by a small amount (average 0.8% difference). In addition, the range between the “change k first” method’s summer and winter k values has a greater theoretical basis. Preliminary results suggest that the “k first method” is the most appropriate, and is selected for further analysis.

3.1.2 Optimal Seasonal Set and k

The RSS results in Table 3-1 and Figure 3-2 indicate that the winter periods were systematically responsible for more than half of the total RSS throughout the study. For comparison, using Environment Canada's (2014) calculated parameters and the LandGEM default parameters result in much higher RSS values (8,211 for Environment Canada values and 2,605 for LandGEM default values).
Figure 3-2. Seasonal RSSs minimized using k first method in part 1
For every optimized seasonal set and iteration method, winter RSSs were greater than the summer RSSs. First, lingering low values in the dataset remained, resulting in large residuals in common winter months such as November and December. For example, the Summer 3 RSS from the “6-month Prec. Winter” set has the lowest subset RSS by 24% compared to second best (Summer 2), yet it only has 1% fewer data points (Table 2-1), suggesting the significant effect that sub-standard operating methane flow rate data can have on seasonal subsets. Second, annual maintenance shutdowns and equipment freezing may have contributed to the gradual declines in LFG collection during certain winters. For instance, 8.4 m$^3$/min of LFG was collected in RE on November 23, 2009, which fell below 0.2 m$^3$/min by November 26, and remained low throughout December.

The optimized “Prec. Control” seasonal set RSS was 2.4% lower than the Full Set. This suggests that subdividing k regardless of seasonal precipitation trends still leads to minor increases in accuracy; however, it is possible that the winter data was more prone to abrupt drops in LFG flow rates, thus increasing RSS in both “Prec. Control” subsets. Operational reasons could include scheduled shutdowns, or freezing in the collection system as reported by Hettiarachchi et al. (2013). Time lags between LFG generation and collection may contribute to the uncertainties, however direct evidences were not observed in the present study.

The results show that the sum of RSSs for each seasonal set is lower than the RSS for the Full set. This suggests that methane estimates may have higher accuracy when seasonal collection k values are used. Using field data from a cold and semi-arid landfill, the optimal $k_{\text{winter}}$ (0.0082 year$^{-1}$) and $k_{\text{summer}}$ (0.0095 year$^{-1}$) are obtained with about 15%
difference. The optimized “5-month Winter” set results in the lowest total RSS value (6.4% lower than Full Set RSS).

By taking the sum of daily values for each year (Figure 3-3) and comparing them to the recorded data, the average percent error for various sets was: 15.5% for the optimized Full Set; 45.1% using Environment Canada inputs (Table 1-1); 76.5% using EPA defaults; and 7.3% using the proposed 5-month Winter. Thus, this study's methods managed to obtain optimized k and L_o that lowered the percent error to just 15.5%, and found that seasonal sets are capable of halving that error.
Figure 3-3. Annual methane generated in RE using various sets
3.1.3 Application of Seasonal k using LFGcost

A simple application study was performed using the EPA's LFGcost-Web 3.0 model to highlight the importance of using optimized sets compared to the suggested defaults for Regina. Optimal seasonal subset k and Lₒ values were used in the model, and then weighted by the number of data points in each subset to determine the net present value for standard engine, small engine, microturbine, and compressed natural gas (CNG) projects (Table 3-2). Other project types were not selected because the required LFG flow rates were too high for the Regina landfill. For example, cost estimates for standard turbines (larger output than microturbines) required greater than 3 MW. Even at maximum LFG flow rates and a k of 0.06 year⁻¹, the Regina Landfill would output a maximum of only 1.1 MW. The selected input parameters included: 28 acre (11 ha) well field, average gas output, 48% methane content, no collection and flaring costs (assumed the existing network will be used), and local government-owned project financing as outlined in the user manual (Alexander et al., 2005).

The default LFG inputs overestimated the net present value compared to the optimized sets for the standard engine (86%), microturbine (107%), and CNG projects (57%). Differences between the optimized sets were negligible. For example, “5-month Winter” and “Prec. Control” had negative net present values of $37,200 and $37,900, respectively, for the microturbine project, while the default set had a positive worth of $541,274. However, seasonal variation in collection may limit potential projects due to varying resilience. For instance, transporting CH₄ to a nearby site for heating would not be acceptable to a client due to unreliable system conditions in winter, when heating is most crucial, rendering a LFG system unattractive.
Table 3-2. Net Present Value for 4 common LFG Energy Projects at RE

<table>
<thead>
<tr>
<th>Set</th>
<th>Standard Engine</th>
<th>Small Engine</th>
<th>Microturbine (10 year)</th>
<th>Compressed Natural Gas</th>
</tr>
</thead>
<tbody>
<tr>
<td>5-month Winter</td>
<td>($690,900)</td>
<td>($253,400)</td>
<td>($37,200)</td>
<td>$811,800</td>
</tr>
<tr>
<td>6-month Temp. Winter</td>
<td>($691,400)</td>
<td>($253,200)</td>
<td>($38,000)</td>
<td>$810,100</td>
</tr>
<tr>
<td>6-month Prec. Winter</td>
<td>($691,000)</td>
<td>($253,300)</td>
<td>($37,600)</td>
<td>$811,100</td>
</tr>
<tr>
<td>Prec. Control</td>
<td>($690,900)</td>
<td>($253,300)</td>
<td>($37,900)</td>
<td>$811,000</td>
</tr>
<tr>
<td>Annual</td>
<td>($690,600)</td>
<td>($253,400)</td>
<td>($37,900)</td>
<td>$811,700</td>
</tr>
<tr>
<td>Default</td>
<td>($370,700)</td>
<td>($413,700)</td>
<td>$541,300</td>
<td>$1,888,700</td>
</tr>
</tbody>
</table>

Note: All numbers rounded to the nearest hundred. Parentheses represent a negative net present value.
The default set underestimated the net present value for the small engine project. This may have been due to an expected capacity range used in LFGcost-Web, as over supplying LFG to the engine may have necessitated project expansions or increased maintenance. It is good practice for municipal planners to accurately estimate project costs over project lifetimes. In the case of three out of four LFG energy project alternatives, using default LFG values in LandGEM and LFGcost-Web would lead to significantly overestimated (57% - 107%) net present values.

3.2 CO₂ Modelling Results

Part 2 of the study had a significantly larger dataset scope. Between the five approaches, five formulas, two algorithms per formula group, and varied starting value sets, 80 CO₂ models were generated per site.

For ease of comparison, Table 3-3 presents only the best starting set results from each combination of approaches and governing equations relative to the LandGEM default Sub-10 approach using field CH₄ content (Default w/ Traces). Results were compared to this approach since it was expected to produce the highest RSSs due to overestimation, and did so for each site except SK, which had exceptionally poor accuracy with the 1.2 Ratio approach.
Table 3-3 Lowest RSSs from studied CO₂ approaches and models

<table>
<thead>
<tr>
<th>Site</th>
<th>Approach</th>
<th>CO₂ (%)</th>
<th>Optimal k ((\text{year}^{-1})^a)</th>
<th>(L_{\text{opt}-\text{CO₂}}) (m³/Mg)</th>
<th>Percent Decrease in RSS (%)(^b)</th>
<th>Sub-1</th>
<th>Sub-2</th>
<th>Sub-4</th>
<th>Sub-10</th>
<th>Annual k</th>
</tr>
</thead>
<tbody>
<tr>
<td>CC</td>
<td>Default w/o Traces</td>
<td>40.8</td>
<td>0.020</td>
<td>(82.8)(^c)</td>
<td></td>
<td>84.8</td>
<td>84.8</td>
<td>84.8</td>
<td>84.8</td>
<td>-23.2</td>
</tr>
<tr>
<td></td>
<td>Default w/ Traces</td>
<td></td>
<td>0.020</td>
<td></td>
<td>-7.8</td>
<td>-3.4</td>
<td>-1.3</td>
<td>0.0</td>
<td>98.7</td>
<td>-208</td>
</tr>
<tr>
<td></td>
<td>Optimized</td>
<td>36.9</td>
<td>0.011</td>
<td>95.5</td>
<td>88.3</td>
<td>88.2</td>
<td>88.2</td>
<td>88.2</td>
<td>68.8</td>
<td></td>
</tr>
<tr>
<td></td>
<td>1.4 Ratio</td>
<td></td>
<td>0.020</td>
<td>59.2</td>
<td>82.4</td>
<td>83.0</td>
<td>83.2</td>
<td>83.4</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>1.2 Ratio</td>
<td></td>
<td>0.020</td>
<td>69.0</td>
<td>22.7</td>
<td>26.1</td>
<td>27.8</td>
<td>28.8</td>
<td>-19.8</td>
<td></td>
</tr>
<tr>
<td>SK</td>
<td>Default w/o Traces</td>
<td>41.0</td>
<td>0.020</td>
<td>(150.0)</td>
<td>-40.5</td>
<td>-46.5</td>
<td>-49.7</td>
<td>-51.6</td>
<td>-1441</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Default w/ Traces</td>
<td></td>
<td>0.020</td>
<td></td>
<td>-5.9</td>
<td>-2.5</td>
<td>-0.9</td>
<td>0.0</td>
<td>-2613</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Optimized</td>
<td>40.1</td>
<td>0.007</td>
<td>115</td>
<td>13.1</td>
<td>13.0</td>
<td>13.0</td>
<td>13.0</td>
<td>95.4</td>
<td></td>
</tr>
<tr>
<td></td>
<td>1.4 Ratio</td>
<td></td>
<td>0.020</td>
<td>81.1</td>
<td>10.6</td>
<td>9.5</td>
<td>8.8</td>
<td>8.4</td>
<td>-2019</td>
<td></td>
</tr>
<tr>
<td></td>
<td>1.2 Ratio</td>
<td></td>
<td>0.008</td>
<td>125</td>
<td>-1182</td>
<td>-1148</td>
<td>-1132</td>
<td>-1122</td>
<td>-7033</td>
<td></td>
</tr>
<tr>
<td>RE</td>
<td>Default w/o Traces</td>
<td>45.5</td>
<td>0.012</td>
<td>(86.2)</td>
<td>83.1</td>
<td>83.5</td>
<td>83.7</td>
<td>83.9</td>
<td>16.7</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Default w/ Traces</td>
<td></td>
<td>0.012</td>
<td></td>
<td>-2.9</td>
<td>-1.3</td>
<td>-0.5</td>
<td>0.0</td>
<td>-30.8</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Optimized</td>
<td>40.0</td>
<td>0.005</td>
<td>113</td>
<td>94.1</td>
<td>94.1</td>
<td>94.1</td>
<td>94.1</td>
<td>99.8</td>
<td></td>
</tr>
<tr>
<td></td>
<td>1.4 Ratio</td>
<td></td>
<td>0.012</td>
<td>61.6</td>
<td>94.0</td>
<td>94.0</td>
<td>94.0</td>
<td>94.0</td>
<td>18.6</td>
<td></td>
</tr>
<tr>
<td></td>
<td>1.2 Ratio</td>
<td></td>
<td>0.012</td>
<td>71.8</td>
<td>83.3</td>
<td>83.7</td>
<td>83.9</td>
<td>84.0</td>
<td>16.8</td>
<td></td>
</tr>
<tr>
<td>HL</td>
<td>Default w/o Traces</td>
<td>41.4</td>
<td>0.045</td>
<td>(83.3)</td>
<td>40.0</td>
<td>38.9</td>
<td>38.2</td>
<td>30.7</td>
<td>47.4</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Default w/ Traces</td>
<td></td>
<td>0.040</td>
<td></td>
<td>-44.2</td>
<td>-37.5</td>
<td>-34.3</td>
<td>0.0</td>
<td>-13.8</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Optimized</td>
<td>34.0</td>
<td>0.049</td>
<td>67.1</td>
<td>97.2</td>
<td>97.2</td>
<td>97.2</td>
<td>96.8</td>
<td>98.0</td>
<td></td>
</tr>
<tr>
<td></td>
<td>1.4 Ratio</td>
<td></td>
<td>0.045</td>
<td>59.5</td>
<td>84.8</td>
<td>82.8</td>
<td>81.7</td>
<td>69.4</td>
<td>81.5</td>
<td></td>
</tr>
<tr>
<td></td>
<td>1.2 Ratio</td>
<td></td>
<td>0.045</td>
<td>69.4</td>
<td>96.9</td>
<td>97.2</td>
<td>97.2</td>
<td>94.8</td>
<td>95.4</td>
<td></td>
</tr>
<tr>
<td>X</td>
<td>Default w/o Traces</td>
<td>42.1</td>
<td></td>
<td></td>
<td>41.9</td>
<td>40.2</td>
<td>39.3</td>
<td>36.9</td>
<td>-350.0</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Default w/ Traces</td>
<td></td>
<td></td>
<td></td>
<td>-15.2</td>
<td>-11.2</td>
<td>-9.2</td>
<td>0.0</td>
<td>-717</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Optimized</td>
<td>37.7</td>
<td></td>
<td></td>
<td>73.1</td>
<td>73.1</td>
<td>73.1</td>
<td>73.0</td>
<td>98.0</td>
<td></td>
</tr>
<tr>
<td></td>
<td>1.4 Ratio</td>
<td></td>
<td></td>
<td></td>
<td>67.9</td>
<td>67.3</td>
<td>66.9</td>
<td>63.8</td>
<td>-463</td>
<td></td>
</tr>
<tr>
<td></td>
<td>1.2 Ratio</td>
<td></td>
<td></td>
<td></td>
<td>-245</td>
<td>-235</td>
<td>-231</td>
<td>-229</td>
<td>-1735</td>
<td></td>
</tr>
</tbody>
</table>

\(a\) Optimized approach k reported as average from all formulas, all other approaches were starting k

\(b\) Reported in terms of reduction from the Default w/ Traces approach (Sub-10 model)

\(c\) Values in parentheses are CH₄ L₀
Overall, the optimization approach led to the most accurate results with an average RSS reduction of 73.1% for subsection formulas (Sub-1, Sub-2, Sub-4 and Sub-10), and 98.0% reduction for “annual k” models. This was expected, although the application of optimized results is limited to improving collection forecasting. The “1.4 Ratio” approach is more applicable for collection system pre-design given reliable k and waste composition data: the mean RSS decrease between the subsection formulas was 66.5% for this approach.

The SK results produced low improvements (aside from annual k formulas) and tended towards high reductions in accuracy for “Default w/o Traces”, and “1.2 Ratio” approaches. This was likely due to it having the shortest available study period (2 years), and monthly LFG flow experienced more fluctuation in 2014 (6.6 to 17.8 m³/min) than 2015 (12.3 to 13.0 m³/min). By removing the SK results, the 1.2 Ratio approach (mean RSS reduction 61.2% for all formulas) becomes comparable with the “Default w/o Traces” approach (57.5%). Overall, the “optimized” and “1.4 Ratio” approaches were the first and second most accurate, and the annual k formulas yielded the lowest improvements rather than reductions in accuracy. Thus the “Ratio approaches” (1.2, 1.4) can be considered comparable or more accurate than assuming 0% trace gases. The results in Table 3-3 will be discussed in the following sections in more detail.

3.2.1 Default with Traces

As expected, the “Default with Traces” approach led to overestimated CO₂ for all sites except SK. The optimized Sub-10 formula for all sites (Table 3-3) ranged from 13.0 to 97.2% lower RSS than the existing approach. CC and RE yielded similar trends, wherein the “optimized” and “Default w/o Traces” approaches were close to each other and the
actual data for most of the study period. SK, however, yielded a higher amount of actual gas in 2015 due to low collection in 2014 (triangle symbols, Figure 3-4), thus skewing the results for optimized models due to the short study period.

As an exception, the “Default w/o Traces” approach consistently underestimated the actual gas data in HL (Figure 3-4), although the resulting RSS was 30.7% lower than the “Default with Traces”. The shift was likely due to the different preferred k’s (0.040 and 0.045 year^{-1}): “Default with Traces” used the lower bound k (Table 3-3) to offset the overestimation tendency of the approach. Unlike HL, all other sites had equal k’s for both approaches, possibly due to the shorter study periods.

For each site, the Sub-10 formula yielded the lowest RSS for the “Default with Traces” approach. The mean RSS (considering all sites) increased by 7.0 to 49.9% using the subsection and annual k modifications. This is consistent with Alexander et al. (2005), who reported that the higher divisor (Sub-10) yields 1-2% lower estimates than the using Sub-1 for CH_4. Since the “Default with Traces” tends to overestimate CO_2, the Sub-10 formula’s systematic lower estimates tend to be more appropriate.
Figure 3-4. Comparison of different modelling approaches with field data
3.2.2 Optimized Results

The optimization process resulted in identical k and L_o-CO2 values in preliminary work for starting sets with equal L_o values (k in the case of method C, Figure 2-6). This is the result of the pseudo-second order algorithm’s first step, where one parameter, usually k, was changed before the other. This affected both “subsection” and “annual k” models. The magnitude of the first changed parameter was insignificant, as the algorithm was dependent on the unchanged parameter. For instance, when k was changed first in Method D, optimized k’s were equal for both starting equations for the same L_o(DOC_f).

The Canadian field data (Figure 2-5) suggests that 1:1 CH4 to CO2 ratios are uncommon in LFG collection, resulting in optimized L_o-CO2 values consistently lower than L_o. This may be partly due to significant differences in solubility (Jeong et al., 2015), as CO2 is 22 times more soluble than CH4 at 35 °C (Gevantman, 2016). It is thus expected that landfills with higher precipitations and higher moisture content (such as HL, Table 2-3) will tend to produce gas with less CO2 (Figure 2-5).

3.2.2.1 Annual k models

Figure 3-5 shows the unique, comparable results for methods C and D. Method D (DOC_f = 0.50 and 0.77) tended to produce lower RSSs for sites with fewer data points, as SK (2 yrs) and RE (6 yrs) are not visible in Figure 3-5 with RSSs of 2020 and 6426, respectively. Method D represented the optimal annual k results in Table 3-3 for every site as a result, in addition to the most optimized results for all approaches and formulas.
Figure 3-5. Optimized Annual k RSSs in part 2
Results for equation 4 in method C are absent from Figure 3-5 due to the tendency to produce enormous RSSs (from 2 to 59 times higher) for all sites except SK. Equation 4 was less applicable for annual use than equation 3 in the present study, as it resulted in negative k values for two years with low precipitation in CC (2009 and 2015, corrected to k=0.00001yr\(^{-1}\)). The least accurate optimized results tended to come from method C overall, thus neither equations 3 or 4 are recommended for use in “annual k” models. Moreover, “annual k” models were less accurate than the “Default with Traces” approach with Sub-10 formula in 9 of 20 cases (Table 3-3), and the worst alternative models for 12 of 20 results. Thompson et al. (2009) however encountered mixed success using lifetime k in multiple models based on average precipitation, among other assumptions.

The D method runs consistently yielded the lowest RSS values, meaning that the success of the annual k governing formula rested on the iterative method rather than starting values. All other methods changed at most 2 reference cells at once (lifetime k and L\(_0\)), and those cells were calibrated based on their applied fit across the study period. Method D, however, was able to calibrate the k values on a per year basis, unencumbered by poor fits with one or more data points, an issue which is reduced using the multi-phase approaches in the newest Afvalzorg and IPCC models.

Method D has potential to determine new annual k formulas. Precipitation has been used as an independent variable in some studies to estimate k using linear relationships (Garg et al., 2006; Karanjekar et al., 2015; Thompson et al., 2009). Of three sites in Figure 3-6 (SK excluded due to its shortest study period), only RE yielded a significant relationship (R\(^2\) = 0.88 and 0.89) between k (optimized via collected CH\(_4\)) and precipitation. The
slopes of the linear formulas for RE in Figure 3-6 are lower than the slope and intercept in equation 3 (equation 4 has a negative intercept), though all four equations have the same slope magnitude. RE’s slope magnitudes are also higher than CC and HL when the k=0.4 year⁻¹ point in HL is ignored. RE’s stronger sensitivity between k and precipitation was surprising considering RE had a final cover during the study period, a feature intended to reduce moisture infiltration, although it can be highly susceptible to freeze-thaw processes common in the cold climate. Low precipitation is unlikely to be the lone reason for the relationship strength, as CC has lower precipitation rates than RE.

3.2.2.2 Subsection models

The optimized results of the subsection models yielded a smaller range of RSSs than the annual k runs. Percent difference between the lowest and highest RSSs among all subsections and starting sets ranged between 1.4% (RE) and 26.1% (CC), while the annual k methods ranged between 75.8% (HL) and 199.8% (SK) by comparison (data not shown). Considering only the sets with the lowest RSS results, the range of differences reduce to <0.01% (about 94.1% reduction in RE) to 8.84% (96.8 to 97.2% reduction in HL), with the other two sites lower than 0.3% (Table 3-3). This trend holds for other approaches aside from the “Default with Traces” approach: no single subsection formula had a consistent, significantly higher accuracy than another for any site (except HL). Differences in k and Lₒ, and thus approaches, had greater effects on RSS. For instance, RE’s “1.4 Ratio” results were all 94.0% lower RSS than “Default with Traces” with Sub-10 formula, while “Default without Traces” ranged between 83.1 and 83.9% reduction, and the “1.2 Ratio” results ranged from 83.3 to 84.0% lower RSS.
Figure 3-6. Method D-optimized k's and precipitation for three sites
Method B, which changes k and L_o together, yielded the best RSSs for both CH_4 and CO_2 datasets in CC and SK (data not shown). For RE and HL, the optimized CO_2 results for method A were more accurate in terms of RSS, despite method B resulting in the lowest CH_4 RSS in both cases. This may be because RE and HL notably have years where trace gases well exceed 15% (Figure 2-5), which may suggest conditions affecting CH_4 k such as aerobic pockets via intrusion.

3.2.3 Simple Ratio models

This approach’s lack of an optimization process resulted in higher RSSs across all formulas, starting values, and sites; however, they were more accurate than the “Default with Traces” approach in most cases (Table 3-3). This was expected given the uncertainty associated with default and calculated k's. The results from simple ratio approaches were comparable to the “Default w/o Traces” approach, thus they may be appropriate for simple pre-design estimates.

The “1.4 Ratio” may be more applicable to semi arid sites, as only HL (a coastal site with more precipitation, see Table 2-3) had higher accuracy using the 1.2 Ratio. The higher accuracy for the 1.4 ratio for three of the sites is supported by the field literature (Table 1-2) and average ratio in these sites’ collection data (1.38). Although the ideal empirical formulas and results in Barlaz et al. (1989a) support the use of a 1:1 ratio still used in some studies (Calabro et al., 2011; Chalvatzaki & Lazaridis, 2010; Goswami et al., 2011; Kumar & Sharma, 2014; Marroni et al., 2010; Rezaee et al., 2014), this ratio is unreliable and too low for most observed field gas composition in the literature (Table 1-2). This is likely due to the higher solubility of CO_2 in water.
The identified 1.2 and 1.4 CH$_4$ to CO$_2$ ratios may provide more accurate CO$_2$ generation forecasts during collection system design than the existing 1:1. However, these ratios may be less applicable to landfills with planned leachate recirculation as observed by Calabro et al. (2011).

### 3.2.4 Modelled CO$_2$

Figure 3-7 compares the formulas and approaches which yielded the lowest RSS for each site. Aside from HL, the “1.2 Ratio” approach tended to overestimate CO$_2$ generation. Slight differences in subsection models resulted in Sub-1 (HL, CC) and Sub-10 (SK, RE) yielding the most accurate of the optimized subsection models.

The actual gas data for HL and CC were subject to some fluctuations throughout operations due to variable upgrade schedules for the well fields, and variable cover over the well field. By comparison, the two sites (RE and SK) in Saskatchewan did not expand their well fields during the study periods. The British Columbia sites (HL and CC) are also active, with new wells installed in lifts with intermediate cover, and old wells mostly under final cover. For instance, final cover at CC overlaid phases 1-3, and the slopes of phase 4 below the active cells.
Figure 3-7. CO₂ modelling approaches and actual data for four western Canadian sites
The quoted collection efficiency for HL may have been underestimated in 2005 (33.0%), as the CO₂ peak occurs in 2005 well before 2013 (Figure 3-7), where the peak should appear due to the landfill’s operating status. The optimized annual k value for HL (Figure 3-6) was 0.4 year⁻¹ for 2005, and is a clear outlier compared to all other points at HL and other landfills. The value peaked at 0.4 year⁻¹ due to an upper boundary constraint, as preliminary work yielded k larger than 1 year⁻¹, well above values seen in the literature.

### 3.2.5 Trace Gases Affecting Default Approach

Nitrogen (N₂) sample data was available for RE (7 days total, 6 prior to 2010) and HL (8 days total, 2007 - 2012). All but 1 sample (HL) had higher N₂:O₂ ratios than the atmospheric average, suggesting partial oxidation via intrusion, and either over-pressurized collection systems or shallow cover (Guter & Nuerenberg, 1987; Hettiarachchi et al., 2013). Partial oxidation at HL is supported by Figure 3-8, wherein CH₄:CO₂ ratios decrease as N₂, an indicator of atmospheric intrusion, increases. HL’s plotted relationship was expected.

Oddly, the RE data suggests the opposite case where CH₄ increased as intrusion increased. This may suggest that most intrusions occurred at a leak in the collection pipes or headers, or that increased moisture content in anaerobic sections of the waste mass (due to higher summer precipitation and infiltration) out-produced the aerobic sections. Low oxidation rates in aerobic sections stemming from reduced ambient temperatures (winter months) may be a factor in similar relationships, but not the one observed in Figure 3-8: the independent gas sample data were collected between April and September (warm, summer months).
Figure 3-8. Independent Sample Gas testing with N₂ at RE and HL
Figure 3-9 shows the range of average monthly residual gas concentrations measured by the continuous gas monitoring system at RE between 2008 and 2013. With closer inspection of these gases, the traces increase before and after winter. The highest average values are all between November (10.1%) and April (12.3%), with a peak in March (17.9%). This may support the idea of frozen condensate blockages and flow overcompensation by operators (Hettiarachchi et al, 2013). Furthermore, there was a tendency in the monthly SK and daily RE data (same province, similar climate) to experience significant drops in LFG collection flow rates during winter months between November and March. Continuous N₂ concentration data was unavailable to further support this observation.
Figure 3-9. Residual gas concentration at Regina landfill, 2008 – 2013
CHAPTER 4: CONCLUSIONS & RECOMMENDATIONS

4.1 Seasonal k Study Conclusions

LFG production and emission mitigation technologies require improvements in estimation and measurement techniques. Modelling is an important tool for understanding and estimating LFG production, collection, and emissions, and so increasing evaluation for various models is important. This study concluded that:

- Default LandGEM values for $k$ and $L_o$ were invalid for the Regina landfill, located in a cold, semi-arid climate. The predicted methane collections were overestimated during the study period (average 76.5% error).

- Changing $k$ and $L_o$ at the same time resulted in $k$ and $L_o$ values inconsistent with the literature. The $L_o$ first method resulted in the highest RSS among the optimized seasonal sets, and an unsupported theoretical basis for seasonal $L_o$ differences. The most reliable method for optimizing LFG constants in LandGEM was the $k$ first iterative method, determined via process of elimination.

- Using separate seasonal collection $k_{\text{winter}}$ and $k_{\text{summer}}$ values increased the accuracy of LandGEM in terms of RSS by 2.3 to 6.4%. By summing daily data to determine annual methane generation values, the optimal seasonal set had just 7.3% error with the data compiled from available records, compared to 15.5% error with the optimized Full Set.
• From the real time gas data, the optimal $k_{\text{winter}}$ was 0.0082 year$^{-1}$, and $k_{\text{summer}}$ was 0.0095 year$^{-1}$ at the Regina landfill. The values were consistent with the other modelling studies using precipitation data alone.

• Optimized LFG constants using annual and seasonal sets produced insignificant differences between net present value estimates in LFGcost-Web 3.0 for four small LFG to Energy projects; however, seasonal variation in collection may limit potential projects due to varying resilience. Using LandGEM's default LFG constants for arid landfills, however, led to net present value overestimations ranging from 57 - 107% for three of the four projects, which would significantly impact municipal budgets.

4.2 CO$_2$ Modelling Approaches Conclusions

CO$_2$ estimates in LandGEM currently rest on the assumptions that CO$_2$ is a function of CH$_4$, and that the two gases make up nearly 100% of LFG content. This can lead to oversights in collection system design, and faulty input estimates for macro climate modelling. Five approaches to LandGEM based CO$_2$ modelling were compared in this study, and concluded the following:

• The current form of the LandGEM equation (Sub-10) yielded the most accurate results for the Default with Traces approach for CO$_2$ modelling for all four sites due to tendency to produce lower estimates. The mean RSS increased by 7.0 to 49.9% using modified formulas (Sub-1, Sub-2, Sub-4, and Annual k). However, this approach was the least accurate compared to nearly all other approaches and formulas at all four western Canadian sites studied.
• Using a CH₄ to CO₂ ratio of 1.4 yielded more accurate results for CO₂ in LandGEM than the existing approach (mean RSS reduction of 66.5% for all sites and subsection models), and to a lesser extent so did a ratio of 1.2 for three of the four western Canadian landfills. This may be due to significant differences in solubility between CH₄ and CO₂ affecting field gas, and high trace gas content (range 0.4 - 28.2%) affecting the existing approach. The method may therefore serve as a better pre-design assumption when limited to traditional landfills with no leachate recirculation. The use of 1:1 ratio in gas modelling is not recommended.

• Annual k models yielded the lowest accuracy in 12 of 20 approaches for the four sites, although they yielded the highest accuracy in all four sites using the optimized approach. Optimized annual k’s yielded the lowest RSS values, however, strong relationships with precipitation were not observed in the present study.

• Method D produced the lowest RSSs for models with optimized parameters, and greatest percent decrease across all approaches and formulas (mean 98.0%) due to per year calibration. Methods A and B were dependent on starting values. Method B’s CO₂ RSS may have been affected by the assumption of equal k for CH₄ and CO₂. Sampling data suggested significant air intrusion in HL and RE, thus decreasing the field CH₄ L₀.

• Two existing empirical formulas (equations 3 and 4) relating k and precipitation were produced worse estimates for model CO₂ when applied to a modified
“annual k” LandGEM. The models were worse than the existing approach and formula in 9 of 20 cases, and the worst alternative model in 12 of 20 cases.

- The best approach for CO₂ modelling was optimization, which had the greatest reduction in RSS over the default approach (73.0 to 98.0%). However this approach depends on available data for gas, waste, and collection efficiency. For pre-design, the 1.4 Ratio may be preferable, depending on anticipated cover system construction and air intrusion.

4.3 Recommendations for Future Study

4.3.1 Validating Seasonal Generation and Governing Formula

- It was specifically assumed that the combination of extended periods of subzero temperatures and low precipitation rates (an assumed factor of infiltration rates) led to seasonal reductions in methane collection rates. Thus, further numerical studies could isolate for cold or arid conditions to determine which factor is more significant, although lab studies would more accurately quantify their effects.

- In addition, the seasonal part of the study was limited to concluding reduced seasonal collection rates, as opposed to generation rates. Generation rates are still difficult to estimate due to the high cost of accurate fugitive emission measurement, LFG migration off-site, and variable methane oxidation rates in cover soils. Some studies approximate generation rates using collection data and assumed oxidation and emission rates. Field studies are required to verify reduced infiltration in winter, and the typical frequency of freeze-induced shutdowns.
• Daily CH$_4$ flow rates were assumed as linear interpolations between annual values output by LandGEM in part 1 of this study. This assumption is common in existing literature (triangle method), and more sophisticated equations, such as variations of the exponential Scholl Canyon formula, were out of scope for the present study. The study also subdivided annual data into only two seasonal periods. Future studies could thus examine variations of a sub-annual exponential formula to further verify the application of seasonal collection k values.

4.3.2 Mass Input Uncertainties for Multi-Site Studies

• An often unobserved, and thus assumed detail in LFG modelling studies is that disposal records denote waste disposal solely into cells and lifts within the well field's area of effect. For example, disposal records from the SK landfill qualitatively noted years when the active face temporarily shifted from older to newer sections of the landfill (and vice versa). During such years, however, separate tonnage disposal data for each area were not recorded. This can result in uncertainties and assumptions in the mass input term in models, especially in landfills where the well field may not expand for several years (RE, SK). Moreover, studies which cover a large number of landfills may be more likely to simply assume all waste disposed over the landfill operating life contributed to gas collection. Thus future field studies should be conducted to compare the difference in model accuracy between recorded, sub-annual chronological disposal data and coarse annual records.

• The modelling approaches presented in this study may be applied to future studies on carbon tax forecasting, a practice more common in European nations.
Carbon tax policies are in development by Canada's Federal government, and have been enacted in B.C. since 2008. The topic is still controversial in some circles, and so improvements in LFG model forecasting may ease the transition for some; this study compared the approaches on a numerical basis, but the implications may be best understood in economic applications.
References


Comcor Environmental Ltd. (2010). *Design basis memorandum: Landfill gas collection system and compressor/flare station project*. Saskatoon, Saskatchewan, Canada.


IPCC. (2007). Climate change 2007: The physical science basis. Cambridge: Cambridge University Press.


Appendix A

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### Cache Creek, B.C. Landfill Gas Records

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*a*) Parentheses indicate assumed collection efficiencies. Lack of parentheses indicates source-reported collection efficiency.

*b*) N/A denotes unavailable data.