Quantifying the Transition to Low-Carbon Cities

by

Eugene A. Mohareb

A thesis submitted in conformity with the requirements for the degree of Doctor of Philosophy
Department of Civil Engineering
University of Toronto

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Doctor of Philosophy

Department of Civil Engineering
University of Toronto

2012

Abstract

Global cities have recognized the need to reduce greenhouse gas (GHG) emissions and have begun to take action to balance of the carbon cycle. This thesis examines the nuances of quantification methods used and the implications of current policy for long-term emissions.

Emissions from waste management, though relatively small when compared with building and transportation sectors, are the largest source of emissions directly controlled by municipal government. It is important that municipalities understand the implications of methodological selection when quantifying GHG emissions from waste management practices. The “Waste-in-Place” methodology is presented as the most relevant for inventorying purposes, while the “Methane Commitment” approach is best used for planning.

Carbon sinks, divided into “Direct” and “Embodied”, are quantified using the Greater Toronto Area (GTA) as a case study. “Direct” sinks, those whose sequestration processes occur within urban boundaries, contribute the largest share of carbon sinks with regional
forests providing a significant proportion. “Embodied” sinks, those whose sequestration processes (or in the case of concrete, the processes that enable sequestration) are independent of the urban boundary, can contribute to the urban carbon pool, but greater uncertainty exists in upstream emissions as the management/processing prior to its use as a sink are generally beyond the consumer’s purview.

The Pathways to Urban Reductions in Greenhouse gas Emissions (or PURGE) model is developed as a means to explore emissions scenarios resulting from urban policy to mitigate climate change by quantifying future carbon sources/sinks (from changes in building stock, vehicle stock, waste treatment and urban/regional forests). The model suggests that current policy decisions in the GTA provide short-term reductions but are not sufficient in the long term to balance the pressures of economic and population growth. Aggressive reductions in energy demand from personal transportation and existing building stock will be necessary to achieve long-term emissions targets.
Acknowledgements

I must start of by acknowledging my family, as they are my foundation. To my parents; you’ve instilled a conservationist mindset in me. You instructed me from the very beginning to appreciate the free gifts that the natural environment provides and to use all resources sparingly. Your love and support, however, have been limitless resources from which I can draw upon. My brothers; Justin, you have always kept me on guard, pushing me to be able to clearly rationalize my thoughts and have made me more curious as a result. Adrian, it’s difficult to quantify all the ways you’ve assisted me in this process. You’ve been my sounding board all my life and I can’t imagine how I would have found this particular path to a doctorate degree without your guidance and being about to share in your passion for sustainability. As well, to my newest parents and brother, the Bhattacharyas; as I have joined your family during this process, you have treated me like I was a son / brother of your own, with all the support and attentiveness that entails. How truly privileged I am to be doubly blessed in this way.

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To my advisor, Prof. Chris Kennedy; thanks for all your encouragement, optimism, and, when necessary, thoughtful redirection. Your patience and the skilled manner by which you draw out ideas from the fog of initial conception were fundamental to my academic achievements. Due to your remarkable insight and vision, I could rely on your guidance to recognise what was important, interesting research and what was not. You are the ultimate motivator, and my words of thanks could never suitably express my depth of gratitude for the opportunities you’ve given me. As I pursue a position in academia, I know I have been given an exceptional model by which to conduct myself.

To my friends, new and old; whether we were meeting for coffee, conversing about world issues over dinner, quarrelling about current affairs around a campfire or debating municipal policies over a foosball table, I have been greatly enriched by my time spent with you. I want to particularly thank the past and present members of the Sustainable Infrastructure Group; I can’t imagine my academic and social experience during my doctoral degree without all the stimulating conversations and laughter we’ve shared. I feel so fortunate to have met so many great people here, and I hope to stay connected with you for the rest of my life.

Finally, to my wife Mahua; your patience in proof-reading my writing and listening attentively to my presentations (both of which you’d likely seen a dozen times in some other iteration) is remarkable. On a serious note, having you as a partner made this thesis possible. Whenever something was out of reach, you gave me the necessary extra inches. I look forward with excitement to the path that we will now set upon together. This thesis is dedicated to you.

I must also acknowledge NSERC for providing me with the financial wherewithal to pursue research of my choosing.
## Table of Contents

List of Tables ................................................................................................................................... x  
List of Figures ................................................................................................................................... xii 
List of Acronyms ............................................................................................................................. xiv 
List of Appendices ........................................................................................................................... xvii 

1 Introduction .........................................................................................................................................1  
   1.1 Sources of Urban GHG Emissions ...................................................................................... 2  
   1.2 Addressing Emissions from Cities ....................................................................................... 3  
   1.3 Objectives and Contributions ............................................................................................... 4  

2 Effects of selection established methods - Waste sector quantification ........................................ 10  
   2.1 Background ........................................................................................................................... 12  
   2.2 Methodology .......................................................................................................................... 14  
      2.2.1 Greater Toronto Area (GTA) ...................................................................................... 14  
   2.3 Waste GHG Emissions Models .............................................................................................. 17  
      2.3.1 FCM-PCP ..................................................................................................................... 18  
      2.3.2 IPCC 1996 .................................................................................................................... 19  
      2.3.3 IPCC 2006 .................................................................................................................... 19  
      2.3.4 USEPA WARM .......................................................................................................... 22  
      2.3.5 Life Cycle-based Approach ......................................................................................... 22  
      2.3.6 Sensitivity Analysis ....................................................................................................... 25  
   2.4 Results & Discussion ............................................................................................................... 27  
      2.4.1 Model Comparison - Landfill Waste ............................................................................. 27  
      2.4.2 Life Cycle-based Approach to Waste Emissions ......................................................... 30  
      2.4.3 Comparison of Net GHG Emissions ............................................................................ 32  
      2.4.4 Uncertainty & Sensitivity Analysis ............................................................................... 34
6.3.1 Potential for Retrofits

6.3.2 Upstream Emissions

6.3.3 Broader Application of PURGE

6.3.4 Adaptation

6.4 Summary

Appendices

Appendix A: Summary Tables
List of Tables

Table 1-1 Contributions of Direct Urban-Related GHG Emissions, Compared with Federal and Provincial Scale

Table 2-1: Waste disposal data applied to the IPCC 1996, 2006, FCM-PCP and/or USEPA WARM for 2005 Waste GHG emission quantification (Sources: City of Toronto, 2005; Barton, 2009; Darnell-Omotani, 2009; Watson, 2009; Durham Region, 2009)

Table 2-2: Parameters applied to the IPCC 1996, 2006 and/or USEPA WARM for 2005 Waste Emission GHG quantification

Table 2-3: Relevant parameters applied in calculating GHG emissions from waste collection

Table 2-4: Gross and Net 2005 Emissions from Waste Management Activities using IPCC 2006 Method of Calculation

Table 2-5: Sensitivity to Uncertain Values of 2005 GHG Emissions from Landfill

Table 2-6: Estimates of 2005 Landfill GHG emissions for parameter estimates

Table 2-7: Comparison of Features of Four Models for Quantifying GHGs from Landfills

Table 3-1: Carbon Sink Enhancement Activities Recognized under Article 3.4 of the Kyoto Protocol (from IPCC, 2000)

Table 3-2: Summary of Concrete Production and Uptake for Concrete Poured in 2003 (Adapted from Pade and Guimaraes, 2007)

Table 3-3: Emission Factor and Parameters Used in Regional Forestry Calculations

Table 3-4: Agricultural Data in GTA (Statistics Canada, 2002; 2007)

Table 3-5: Stock Change Factors and Key Assumption used in Cropland Carbon Sink Calculations (IPCC, 2006)
Table 3-6: Summary of 2005 Direct and Embodied Carbon Sinks in the GTA ......................... 66

Table 3-7: Unquantified Sources Associated with Carbon Sinks Quantified for the GTA ........ 68

Table 4-1: Proportions of Toronto GHG associated with the Four Major Sectors to be Assessed (City of Toronto, 2007a) ........................................................................................................ 86

Table 4-2: Comparison between Actual Inventory Data and PURGE Model Results .......... 114

Table 5-1: Parameters applied to the PURGE model under various scenarios .................... 135

Table 5-2: High and low emissions scenarios for the GTA from the PURGE model ............. 139
List of Figures

Figure 2-1: Waste disposed in landfills from the GTA between 1955-2005................................. 16

Figure 2-2: Plot of Methane Emissions from 2005 GTA Landfill Waste (IPCC 2006 MC)....... 21

Figure 2-3: Flowchart displaying boundaries for IPCC 2006 LC............................................. 23

Figure 2-4: 2005 GHG Emissions (t CO2e) from LFG Released from Sites Handling GTA Waste Quantified using Six Distinct Approaches................................................................. 27

Figure 2-5: Gross & Net Annual GHG Emissions (2005) from Various Treatment Options for IPCC 2006, compared with USEPA WARM ................................................................. 33

Figure 3-1: Graphical representations of carbon fluxes associated with direct (a) and embodied carbon sinks (b, c, and d) ...................................................................................... 50

Figure 4-1: Typical Technological Diffusion Curve .................................................................... 88

Figure 4-2: Sigmoidal Adoption of Various Technologies in Cars (Source: USEPA, 2010)....... 95

Figure 4-3: Unofficial OPA Projections for Electricity Grid Emissions Intensity to 2050, with a Comparison to Actual Intensity Changes................................................................. 102

Figure 4-4: Projections of Market Diffusion of Alternative Vehicles Using Current and Government Rates of Adoption ........................................................................................... 104

Figure 4-5: Number of Retrofit Exit Audits Registered Nationally Over Time (Source: OEE, 2011) ........................................................................................................................................ 105

Figure 4-6: LEED-Registered Non-Residential New Construction Projects in the GTA, 2003-2010 (CaGBC, 2011) ........................................................................................................... 107

Figure 4-7: GTA GHG Emissions from sectors quantified using the PURGE model.............. 109

Figure 4-8: Changes in Transportation Sector Over Time: a) Vehicle Stock Composition; b) VKT travelled annually; c) Fuel Efficiency of Vehicle Types......................................................... 111
Figure 4-9: Contributions of Residential Energy Consumption from Existing and Future Building Stock ............................................................................................................................ 112

Figure 4-10: Commercial Building Emissions by Fuel Type and Era of Construction ............ 112

Figure 4-11: Emissions from the Management of Residential Solid Waste, plotted with Electricity Generation from Incineration, Landfill Gas Collection and Anaerobic Digesters (AD). MC=Municipal Composting and BC = Backyard Composting ........................................ 113

Figure 5-1: a) Scenario T1, b) Scenario T2 and c) Scenario T3 for Vehicle Technology Adoption Applied to the PURGE Model; ICE - C = Internal Combustion Engine - Conventional; ICE – T = Internal Combustion Engine – Turbo; HEV = Hybrid Electric Vehicle; PHEV = Plug-in Hybrid Electric Vehicle; BEV = Battery Electric Vehicle .............. 130

Figure 5-2: a) Diffusion of Home Retrofits; b) Building Code Changes into the GTA Housing Stock ........................................................................................................................................... 132

Figure 5-3: a) Scenarios for Energy Intensity of New Office Construction and b) Energy Intensity of Existing Buildings ................................................................................................... 134

Figure 5-4: GTA GHG Emissions in 2050 from a) Transportation; b) Residential Buildings (Aggressive Building Code); c) Commercial/Institutional Buildings (Aggressive Retrofits) .... 141

Figure 5-5: Transportation GHG Emissions using the Metrolinx Demand Reduction Scenario 142
## List of Acronyms

<table>
<thead>
<tr>
<th>Acronym</th>
<th>Associated Term</th>
</tr>
</thead>
<tbody>
<tr>
<td>AD</td>
<td>Anaerobic Digestion</td>
</tr>
<tr>
<td>AD</td>
<td>Anaerobic Digesters (Chapter 4)</td>
</tr>
<tr>
<td>AFOLU</td>
<td>Agriculture, Forestry and Land-Use (Inventorying category)</td>
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<tr>
<td>ALHB</td>
<td>Asian Long-Horned Beetle</td>
</tr>
<tr>
<td>BAU</td>
<td>Business-as-Usual</td>
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<tr>
<td>BC</td>
<td>Backyard Composting</td>
</tr>
<tr>
<td>BEV</td>
<td>Battery Electric Vehicle</td>
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<tr>
<td>C</td>
<td>Carbon</td>
</tr>
<tr>
<td>C/I</td>
<td>Commercial &amp; Institutional (Building Categories)</td>
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<tr>
<td>CAFE</td>
<td>Corporate Average Fuel Economy</td>
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<td>CBM-CFS3</td>
<td>Carbon Budget Model of the Canadian Forest Sector V3 (forestry carbon storage model)</td>
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<td>CCS</td>
<td>Carbon Capture and Storage</td>
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<tr>
<td>CF</td>
<td>Carbon Fraction</td>
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<tr>
<td>CH₄</td>
<td>Methane</td>
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<tr>
<td>CO₂</td>
<td>Carbon Dioxide</td>
</tr>
<tr>
<td>CO₂ₑ</td>
<td>Equivalent of Carbon Dioxide</td>
</tr>
<tr>
<td>CRW</td>
<td>Crown Cover Area-based Growth Rate of Woody Perennial</td>
</tr>
<tr>
<td>CSF</td>
<td>Carbon Storage Factor</td>
</tr>
<tr>
<td>DDOC</td>
<td>Decomposable Degradable Organic Carbon</td>
</tr>
<tr>
<td>DOC</td>
<td>Degradable Organic Carbon</td>
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<td>DOM</td>
<td>Dead Organic Matter</td>
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<td>EAB</td>
<td>Emerald Ash Borer</td>
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<td>EC</td>
<td>Energy Usage (consumed)</td>
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<td>Emissions Factor</td>
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<td>Electricity Generated</td>
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<td>EI</td>
<td>Emissions Intensity</td>
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<td>EUI</td>
<td>Energy Use Intensity</td>
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<td>Electric Vehicle</td>
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<td>FAO</td>
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<td>Fossil Carbon Fraction</td>
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<td>FCM</td>
<td>Federation of Canadian Municipalities</td>
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<tr>
<td>FCM-PCP</td>
<td>Federation of Canadian Municipalities - Partners for Climate Protection</td>
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<td>GDP</td>
<td>Gross Domestic Product</td>
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<td>GHG</td>
<td>Greenhouse Gas</td>
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<td>GIS</td>
<td>Geographic Information System</td>
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<td>GTA</td>
<td>Greater Toronto Area</td>
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<tr>
<td>GtC</td>
<td>Gigatonnes of Carbon</td>
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<td>GWP</td>
<td>Global Warming Potential - 100-year time interval</td>
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<tr>
<td>Ha</td>
<td>Hectares</td>
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<tr>
<td>Abbreviation</td>
<td>Full Form</td>
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<td>--------------</td>
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<tr>
<td>HEV</td>
<td>Hybrid Electric Vehicle</td>
</tr>
<tr>
<td>HFCV</td>
<td>Hydrogen Fuel Cell Vehicle</td>
</tr>
<tr>
<td>HWP</td>
<td>Harvested Wood Products</td>
</tr>
<tr>
<td>ICE</td>
<td>Internal Combustion Engine</td>
</tr>
<tr>
<td>ICI</td>
<td>Industrial, Commercial &amp; Institutional</td>
</tr>
<tr>
<td>ICLEI</td>
<td>Local Governments for Sustainability</td>
</tr>
<tr>
<td>IEA</td>
<td>International Energy Agency</td>
</tr>
<tr>
<td>IEEE</td>
<td>Institute of Electrical and Electronic Engineers</td>
</tr>
<tr>
<td>IESO</td>
<td>Independent Electricity System Operator</td>
</tr>
<tr>
<td>IPCC</td>
<td>Intergovernmental Panel on Climate Change</td>
</tr>
<tr>
<td>IPCC - NGGIP</td>
<td>Intergovernmental Panel on Climate Change - National Greenhouse Gas Inventory</td>
</tr>
<tr>
<td>KG</td>
<td>Kilogram</td>
</tr>
<tr>
<td>KM</td>
<td>Kilometres</td>
</tr>
<tr>
<td>kN</td>
<td>Kilo Newtons</td>
</tr>
<tr>
<td>KT</td>
<td>Kilotonnes</td>
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<tr>
<td>L</td>
<td>Litres</td>
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<tr>
<td>LC</td>
<td>Life Cycle</td>
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<tr>
<td>LEED</td>
<td>Leadership in Energy &amp; Environmental Design</td>
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<tr>
<td>LFEE</td>
<td>Laboratory for Energy and Environment</td>
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<td>LFG</td>
<td>Landfill Gas</td>
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<td>LL</td>
<td>Line Loss</td>
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<td>MC</td>
<td>Methane Commitment (Landfill GHG emission quantification approach)</td>
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<td>MC</td>
<td>Municipal Composting (Chapter 4)</td>
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<tr>
<td>MCF</td>
<td>Methane Correction Factor</td>
</tr>
<tr>
<td>MIT</td>
<td>Massachusetts Institute of Technology</td>
</tr>
<tr>
<td>MJ</td>
<td>Megajoule</td>
</tr>
<tr>
<td>Mm$^3$</td>
<td>Million cubic metres</td>
</tr>
<tr>
<td>MPG</td>
<td>Miles per Gallon</td>
</tr>
<tr>
<td>MSW</td>
<td>Municipal Solid Waste</td>
</tr>
<tr>
<td>Mt</td>
<td>Megatonnes</td>
</tr>
<tr>
<td>MUR</td>
<td>Multi-Unit Residential</td>
</tr>
<tr>
<td>MW</td>
<td>Mega Watts</td>
</tr>
<tr>
<td>MWh</td>
<td>Mega Watt-hour</td>
</tr>
<tr>
<td>N2O</td>
<td>Nitrous Oxide</td>
</tr>
<tr>
<td>NG</td>
<td>Natural Gas</td>
</tr>
<tr>
<td>NPP</td>
<td>Net Primary Production</td>
</tr>
<tr>
<td>NRCan</td>
<td>Natural Resources Canada</td>
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<tr>
<td>OECD</td>
<td>Organization for Economic Cooperation and Development</td>
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<td>OEE</td>
<td>Office of Energy Efficiency</td>
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<td>OPA</td>
<td>Ontario Power Authority</td>
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<td>PHEV</td>
<td>Plug-in Hybrid Electric Vehicle</td>
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<td>PURGE</td>
<td>Pathways to Urban Reductions in Greenhouse gas Emissions (GHG quantification</td>
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<tr>
<td>Abbreviation</td>
<td>Full Form</td>
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<td>--------------</td>
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</tr>
<tr>
<td>PV</td>
<td>Photovoltaic</td>
</tr>
<tr>
<td>SFH</td>
<td>Single-Family Housing</td>
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<tr>
<td>SOC</td>
<td>Soil Organic Carbon</td>
</tr>
<tr>
<td>t CO₂e</td>
<td>Tonnes of Equivalent Carbon Dioxide</td>
</tr>
<tr>
<td>TCSA</td>
<td>Toronto City Summit Alliance</td>
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<td>TJ</td>
<td>Terajoules</td>
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<td>UFORE</td>
<td>Urban Forest Effects Model</td>
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<td>UN</td>
<td>United Nations</td>
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<td>Dollars - United States</td>
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<tr>
<td>USEPA</td>
<td>United States Environmental Protection Agency</td>
</tr>
<tr>
<td>USGBC</td>
<td>United States Green Building Council</td>
</tr>
<tr>
<td>VKT</td>
<td>Vehicle Kilometers Travelled</td>
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<tr>
<td>WARM</td>
<td>Waste Reduction Model (Landfill GHG emission quantification approach)</td>
</tr>
<tr>
<td>WIP</td>
<td>Waste-in-Place (Landfill GHG emission quantification approach)</td>
</tr>
</tbody>
</table>
List of Appendices

Appendix A: Summary Tables

Table A.1: Values Applied to the USEPA WARM Model

Table A.2: Emissions Factors for Various Landfill Options as Applied to the U.S. EPA WARM Model, as well as Net Emissions from Landfilling

Table A.3: Marginal Emissions Factors from the Electricity Grid (OPA, 2011)

Table A.4: Emissions from Landfilled Residential Waste for Various Waste Stream Components from the GTA in 2005 as Calculated Using the WARM Model


Table A.7: Parameters Applied to PURGE Model for BAU Scenario

Table A.8: Summary of Scenario Options Applied to PURGE Model
1 Introduction

If humanity is to maintain the high quality of life enjoyed in the developed world and the increases in prosperity occurring within developing nations, two great challenges must be met: mitigating climate change and achieving energy security. It is generally agreed that global fossil fuel consumption is contributing to a warming climate, with potentially destabilizing effects on our natural support systems (Oreskes, 2004; Doran and Zimmerman, 2009). Meanwhile, increasing demand for, and uncertain supply of, fossil primary energy suggests price volatility, especially in the transportation sector (Tsokounoglou et al., 2008; Hirsch et al., 2005; Kerschner and Hubacek, 2009). Addressing these issues will require new systems for the provision of energy services, systems that are more efficient and less carbon-intensive. With 50% of the world’s population currently dwelling in cities, and over 70% of people projected to be living in cities by 2050, it is logical to focus on urban energy systems as a principal target for technological change (UN, 2008).

As a result of the urbanization of the global population, the case can be made that cities will dictate whether total global greenhouse gas (GHG) emissions growth or decline. Through consumption patterns and the provision of necessary energy services that populations demand (transportation, space conditioning, internal/external lighting), GHG emissions will inevitably be proportionally related to population given a fixed technological mix. Given the expected increase in population, it will henceforth be necessary to address urban sources of GHG emissions if reductions on the scale of the 80% target suggested by IPCC (2007) are to be achieved. The paths to these reductions will be complex and require significant new investment in changing the infrastructure that transports, feeds, houses and provides employment to urban residents.

It is also of value to focus on the urban scale due to the relative agility to make decisions, relative to higher levels of government. While North American federal governments have been slow to initiate policy addressing climate change, there has been active coordination of global cities to quantify and reduce GHG emissions (ICLEI, 2011; US Conference of Mayors, 2011; FCM, 2011). Due to the recognition of the need for action by policy makers at the urban scale, it is likely that they will be key lobbyists for action at the provincial/state and federal levels. The
starting point for being able to decide where emissions can be reduced is through quantification, which is the focus of the discussion below.

1.1 Sources of Urban GHG Emissions

Urban GHG emissions can predominantly be broken down into four major sources: transportation, buildings, electricity generation and waste. Of the 734 Mt of CO₂e, these sources directly contributed a total of 48.2% of all emissions according to the 2008 Canadian National Inventory Report (2010; see Table 1.1). However, at the Provincial scale, these particular direct sources were found to contribute 60.8%. This demonstrates the effect of reducing the spatial boundary on emissions inventories; inevitably, upstream/indirect emissions are excluded resulting in smaller emissions totals and larger proportions attributable to direct GHG releases. A much greater emissions total would result within cities if upstream emissions from the consumption of materials (such as steel and concrete) and fuels (such as gasoline and coal) were included in emissions inventories. With this finer resolution, however, one can begin to more accurately isolate what the exact sources of emissions are and more readily address them.

Table 1-1 Contributions of Direct Urban-Related GHG Emissions, Compared with Federal and Provincial Scale

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Electricity</td>
<td>16.2%</td>
<td>14.4%</td>
<td>25.4%</td>
</tr>
<tr>
<td>Buildings (Thermal)</td>
<td>10.6%</td>
<td>17.5%</td>
<td>35.5%</td>
</tr>
<tr>
<td>Road Transportation</td>
<td>18.4%</td>
<td>25.0%</td>
<td>35.2%</td>
</tr>
<tr>
<td>Waste</td>
<td>3.0%</td>
<td>3.9%</td>
<td>4.0%</td>
</tr>
<tr>
<td>% of Total Emissions</td>
<td>48.2%</td>
<td>60.8%</td>
<td>100.0%</td>
</tr>
</tbody>
</table>

1 Environment Canada, 2010; 2 City of Toronto, 2007

In addition to source quantification, carbon sinks also exist within urban boundaries and merit consideration. While source reduction is the most important course of action, it is of interest to examine how much carbon is stored as a result of decisions made within urban jurisdictions. A number of urban centres have attempted to quantify carbon sinks associated with their urban forests. The City of Toronto (2009) is one such example, where an Urban Forestry Effects (UFORE) study in 2008 estimated that the City’s trees sequester 36.5 kt C (net), annually. To
provide context for this sequestration total, is 0.5% of the 23.4 Mt CO$_2$e emitted from energy use in 2004. This type of study is not uncommon for urban forestry; however, the quantitative estimates of other types of sinks (such as urban soils or regional forests) are not included in inventories. The focus has generally remained on GHG emissions sources, whose magnitude dominates the imbalance of the carbon cycle on the regional scale, rather than the absence of sinks.

1.2 Addressing Emissions from Cities

The percentage of emission sources from different sectors (residential, industrial, transportation) varies considerably between global cities. Kennedy et al., (2009a) found that cities in colder climates and lower gross domestic product (GDP) generally have lower industrial/heating fuel consumption. As well, urban density and GDP also demonstrated an inverse correlation with transportation fuel consumption. The Greater Toronto Area (GTA), like other North American cities assessed, demonstrates a strong dependency on transportation fuels. The exception is New York City, which has a density three times that of the GTA. Additionally, the GTA is situated in a cold climate, resulting in a reliance on heating fuels – third highest amongst cities surveyed. Considering these factors, GHG emission reductions will require redevelopment of existing infrastructure to maintain energy services while reducing GHG emissions.

Recent years have seen significant reductions in the per capita emissions in the GTA. Between 2005 and 2009, per capita emissions have declined nearly 18% (Civic Action, 2011). However, much of this decline is attributable to the closure of coal-fired generating stations in the wider province. The decline in the carbon intensity of energy is an important approach towards overall GHG emission reduction; however, demand reduction would meet this end while also improving resilience to energy cost/supply issues. It is the combination of these two approaches that will lead to a deep, long-term reduction in emissions. An important part of both carbon intensity and energy demand reduction is the transition to alternative technologies to replace those currently providing energy services in the GTA.

Cities have been recognized as national economic engines, acting as the centres of wealth that drive national demand, as well as providing technological capacity to the wider economy (Jacobs, 1984). Bettencourt et al., (2007a) found that a power law relationship exists for various attributes of cities and population, namely,
$Y(t) = Y_0 N(t)^\beta$ \quad (1.1)

where $Y$ is the urban attribute, such as GDP, $N$ is population and $\beta$ is the scaling exponent. Inventors, research and development jobs and patent applications all have scaling exponents greater than 1, suggesting that large cities result in greater innovation per capita than smaller settlements. As well, urban GDP and average wages demonstrate a power law correlation with urban population, with a scaling exponent that is also greater than unity (Bettencourt et al., 2007b). This implies that cities have an advantage in their technological redevelopment, as the entrepreneurs and innovators who are likely provide the alternative technology systems (and the individuals with the means to adopt these solutions) already dwell within their boundaries.

### 1.3 Objectives and Contributions

This document is a compilation of four works of research, each examining components of the broader question of how to quantify GHG sources/sinks and how to quantify our current measures for reducing these. As stated above, cities are major contributors to climate change and can be important agents of change in the transition to a low-carbon future. The first two works focus on quantification issues, while the final two examine the dynamics of GHG emissions through the implementation of technologies that are more energy-efficient and/or have a lower carbon intensity. The objective of each of these works, as well as their contribution, is described below.

The first step required when commencing an assessment of climate change mitigation strategies for cities is quantification. International standards for national inventories are provided by the Intergovernmental Panel on Climate Change’s National Greenhouse Gas Inventories Programme (IPCC-NGGIP), with the first standards document published in 1996. However, as municipalities have become interested in emissions reductions, the need for addressing the void in urban scale inventory methodologies has become apparent.

Various early inventoring schemes for cities are found in the literature on urban GHG quantification. Early inventories published from cities use differing methodologies, inhibiting the ability to compare and contrast emissions between urban areas (Kennedy et al., 2009); temporal and spatial boundaries are inconsistently selected across many inventories, rendering any sector-specific emissions comparison invalid. It is hence important that methodologies be
selected that are appropriate for the ultimate application (inventorying vs. planning). In Chapter 2, research on different methods for waste GHG emissions quantification is presented to determine their impact on inventories. Currently, there is variability between approaches that cities take to enumerate the emissions from waste. In North America, landfills contribute the greatest proportion of waste sector emissions. Landfills are sources of methane (CH$_4$), a potent GHG that has a 100-year global warming potential of 25 (IPCC, 2007). In quantifying CH$_4$ emissions from landfill, two approaches are generally used to determine the rate of decay of biogenic carbon: the waste-in-place (WIP) and the methane commitment (MC) approaches. These approaches, examined in the context of four different widely applied methodologies, are utilized to quantify emissions in a case study of the GTA.

The four methodologies used in the case study are the IPCC 2006 guidelines, the IPCC 1996 guidelines, the United States Environmental Protection Agency (USEPA) waste reduction (WaRM) model and the Federation of Canadian Municipalities Partners for Climate Protection (FCM-PCP) tool. The latter three methods listed use a MC approach, where a projection is made to estimate all future emissions resulting from waste deposited in the inventory year. The IPCC 2006 examines waste deposited in landfills in previous years, and estimates emissions occurring in the inventory year. This study provides guidance to policy makers in their selection of waste emissions, and demonstrates the nuances of methodologies of varying complexity.

The focus of Chapter 3 is the quantification of carbon sinks within the urban boundary. As mentioned earlier, carbon sinks are generally not quantified in urban inventories. If the ultimate goal toward sustainability in urban systems is to balance the metabolic processes and their related energy, nutrient and material flows, then it is important to examine both sides of the carbon cycle. Sinks are first classified into two categories: direct and embodied. Direct sinks include natural sinks where processes that store carbon are attributed to activities occurring within urban borders, whereas embodied sinks are those where demand for goods and services results in carbon storage due to sequestration (or processes which enable sequestration) upstream. Examples of direct sinks include forests and agricultural soils, while embodied sinks include harvested wood products (HWPs) and concrete.

The direct and embodied sinks are then quantified for the GTA. IPCC 2006 methodologies are used for direct sink inventories, while embodied sinks are quantified using literature sources.
The chapter is useful in that it provides potential data sources available to urban sinks and then demonstrates calculations of these sinks. This chapter closes with a discussion of means by which sinks can be enhanced within urban boundaries, using new technological means (such as artificial trees) or removing restrictions on the use of HWPs in building construction.

After examining quantification issues related to GHG sources and sinks within the urban environment, this thesis focuses on mitigation in Chapters 4 and 5. Specifically, the impact of technological change is assessed through the presentation of a model developed in this research, where current policies to reduce GHG emissions in the GTA are applied (Chapter 4) and scenarios for alternative technological transition pathways are then explored (Chapter 5). This model, entitled Pathways to Urban Reductions in Greenhouse gas Emissions (or PURGE), provides a means to quantify emissions dynamics, based on an existing technology set. The model allows for the projection of GHG emissions from private transportation, buildings and waste, while also being able to quantify carbon stored in street trees and regional forests.

The PURGE model is applied using emission reductions targets for technological change in the GTA; these include the wider adoption of battery-based vehicles, retrofits to existing buildings and changes to the existing building code. This is a novel approach to examining GHG fluxes over time based on existing data on building, vehicle, waste treatment technology and forest stocks. Through the PURGE model, urban policy makers can explore what actions will be necessary to reduce urban GHG emissions to match their stated goals.

A multitude of scenarios are then applied to the PURGE model, examining changes in the private vehicle, commercial building and residential building stocks. These sectors are examined specifically, since they are responsible for over 90% of urban emissions (excluding emissions embodied in materials, food and energy consumed by cities). Parameters tested for their impact on future emissions include population, GDP, new construction energy intensity, electrical grid carbon intensity, and private vehicle technology. This scenario analysis is instructive concerning the impacts of actions, whether business-as-usual or aggressive technological change, on future emissions. This also provides insight concerning the depth of emissions reductions that can be achieved through redevelopment of the technology stock.

The ultimate goal of these collected works is to further the understanding of how cities presently contribute to climate change and energy demand, while providing useful tools and case studies.
for policy makers to plan for a low-carbon future. In the chapters that follow, the following questions will be addressed:

- What impact do boundary and/or methodological selection have on the quantification of GHG emissions from waste? (Chapter 2)
- How significant are carbon sinks within the urban environment? (Chapter 3)
- How can carbon sinks be classified, based on the temporal and spatial boundaries applied? (Chapter 3)
- What are the current options to increase the magnitude of carbon sinks within the urban boundary? (Chapter 3)
- How can future emissions from cities by quantified by examining existing/future technology stocks? (Chapter 4)
- What magnitude of emissions reductions can be expected based on current technological adoption trends and policies/targets suggested by various levels of government with jurisdiction in the GTA? (Chapter 4)
- Based on a literature review that suggests possible technological change pathways, how do different scenarios in the GTA impact GHG emissions to the year 2050? (Chapter 5)

This work is summarized in a concluding chapter, outlining significant findings that address each of the above questions. Opportunities for further improvement and refinement of the work presented here are then discussed. Finally, appendices outlining key data sources and parameters used in the PURGE model are provided for repeatability.

**Note:** All units used in this thesis are assumed to follow the Systeme Internationale, unless explicitly stated.

**List of Publications (Submitted and Expected to be Submitted)**


References


2 Effects of selection established methods - Waste sector quantification

The release of landfill gas (LFG) resulting from anaerobic decomposition of municipal solid waste (MSW) is generally quantified in greenhouse gas (GHG) emissions inventories conducted by cities. For 2007, this emissions source represented 21 Mt (roughly 3%) of total emissions tabulated in the Canadian national GHG inventory and 127 Mt (2%) in the US inventory (Environment Canada, 2009; USEPA 2010). Municipalities, who have been vocal advocates for addressing climate change, play the principal role in managing these GHGs since their decisions dictate diversion, treatment and mitigation (such as LFG capture) practices. The opportunity for reductions is large; an example from the 2004 City of Toronto inventory suggests that solid waste contributed 3% of community-wide emissions; however, its proportion of corporate emissions (those stemming strictly from municipal government activities) was 45% (City of Toronto, 2007). Additionally, waste emissions generally contribute a larger proportion of community-wide municipal emissions in the developing world (e.g. up to 40% in Rio de Janeiro; Kennedy et al., 2009a). The method selected for quantifying waste-related emissions is important, as projects to mitigate MSW-related GHG emissions are likely to be a high priority; Kennedy et al., demonstrated that waste emissions reduction strategies tend to be the most cost-effective of municipal projects targeting GHGs regardless of region, underscoring the importance of proper quantification for planning purposes (Kennedy et al., 2009b).

Greenhouse gas emissions are released through a number of waste management treatment options. However, the greatest source of waste-related GHGs in the 2007 Canadian National Inventory is anaerobic digestion (AD) in landfills, contributing 95% of all Waste sector emissions (Environment Canada, 2009). When biogenic carbon is deposited in landfills, degradation processes become anaerobic after oxygen is depleted in the fill material, producing LFG that is roughly 50% methane (CH$_4$). This GHG is 25 times more potent over a 100-year timeframe than if the same biogenic carbon were aerobically degraded to CO$_2$, which would presumably be a carbon-neutral process (IPCC, 2007). Hence, whenever landfill CH$_4$ is oxidized through combustion or a specially-engineered landfill cover, a reduction in radiative forcing is achieved (compared to a case where CH$_4$ emissions are not controlled). Other possible GHG sources from solid waste include (IPCC, 2006):
• Combustion of fossil fuel-derived carbon in incineration systems resulting in the release of CO₂
• Production of CH₄ from anaerobic conditions within composting operations
• Release of N₂O during nitrification in compost piles
• Leakage of CH₄ from anaerobic digestion reactors
• Collection and transportation of waste to transfer & treatment sites (indirect).

While policy measures to reduce GHG emissions from MSW appear straightforward (such as improved recycling of wood products and diversion of food wastes), inaccurate quantification of these may distort the issue’s scale (and economic feasibility, if carbon pricing is part of the rationale for a mitigation project). Comparison of emissions totals is complicated due to the fact that two different temporal boundaries have been applied to MSW emissions studies; GHGs can be quantified using either the methane commitment (MC, or Theoretical Yield Gas) method or the waste-in-place (WIP) method. The MC method requires the forecast of any future methane emissions associated with MSW deposited in the inventory year, basing this estimation on a projection of future landfill operation practices. The WIP method attempts to quantify methane released within the inventory year from all MSW waste previously deposited in landfills.

The objective of this paper is to quantify and compare GHG emissions associated with waste management using various methodologies that are currently employed for inventorying purposes, as different approaches are being used by cities globally (generally using MC approaches including USEPA WARM and IPCC 1996; Kennedy et al., 2009a). The importance of this exercise stems from the potential for comparisons between global cities’ emissions, which are likely to be made even though boundaries used in their inventories may differ. Quantification of a single case study provides insight into the effect of inconsistent methodological selection between these cities. Additionally, comparing different methodologies to quantify GHG emissions from MSW and analysis of the effects of parameter selection is useful for waste planners/managers. WIP and MC approaches are examined, looking at both direct and indirect emissions associated with different MSW management practices. Once the details of the nuances of quantification methodologies are clearer, policy makers will be able to select the approach that best suits their needs in a particular application (i.e. inventorying vs. waste management planning) and apply it with knowledge of its strengths and weaknesses.
Landfill, incineration, AD and/or composting GHG emissions are calculated, using the Greater Toronto Area (GTA) as a case study, by applying four commonly-used models: Intergovernmental Panel on Climate Change (IPCC) 1996, IPCC 2006, USEPA Waste Reduction Model (WARM) and the Federation of Canadian Municipalities – Partners for Climate Protection (FCM-PCP) quantification tool for communities (IPCC, 1996; IPCC, 2006; USEPA, 2009; FCM, 2009). Additionally, two modifications of the IPCC 2006 model are made to allow for further analysis: one to provide a MC calculation (henceforth termed IPCC 2006 MC) and one to provide a limited life cycle-based inventory (IPCC 2006 LC). The IPCC 2006 LC predominantly includes emissions/credits that would not be included in the IPCC 2006 MC, but that would occur within the municipal boundary and may be relevant to a municipal emissions inventory. This results in a comparison of six different approaches.

2.1 Background

Some information must be provided on the methodologies used in this study to provide an understanding of where they originated, how they are designed and their intended uses. In 1991, the IPCC initiated the National Greenhouse Gas Inventories Programme to commence work on methodologies for quantifying GHG dynamics for member countries (IPCC, 1996). The program aimed to attain consensus with its members by developing emissions/sink inventories and established a task force to aid nations in the quantification of their GHG emissions (IPCC-NGGIP, 2009). The result has been two guidelines (henceforth referred to as IPCC 1996 and IPCC 2006) which have two important differences: The IPCC 1996 model uses a MC calculation while the IPCC 2006 revision uses a WIP method (using > 10 years of detailed landfill disposal data).

The other main difference between the two models is the data requirement. As the 1996 method uses the MC approach, it is based on a simple calculation which employs an estimate of waste carbon content that is dissimilated to methane over an infinite time period (assuming no changes in landfill conditions). Only the tonnage deposited within the year of inventory is required, while default data can be applied to fill in any missing information. The IPCC 2006 WIP method requires the use of a more complex first-order decay model that estimates the degree of decomposition of accumulated carbon in landfilled waste based on half-life data of materials.
under given landfill conditions, which has a greater data requirement (waste deposited from up to 50 years prior is suggested).

*Partners for Climate Protection (PCP)*, the Federation of Canadian Municipalities (FCM) program on climate change action (in association with ICLEI’s global Cities for Climate Protection program), have developed a spreadsheet tool that can be used by municipalities to complete a community and corporate GHG inventory (FCM, 2009). This tool employs a MC approach, as it simply requires an estimate of waste landfilled in a given year, based on a fixed emissions factor (t CO$_2$e (t landfill waste)$^{-1}$). It should be noted that at the time of writing, there are plans to update the FCM-PCP municipal quantification tool (Conner, Personal Communication, June 2010).

The USEPA WARM model was created to assist municipal waste planners in making better decisions with respect to GHG emission mitigation from waste (USEPA, 2009). The model allows the quantification of emissions from landfills (using a MC approach), composting, incineration and recycling. Due to the life-cycle perspective taken, emissions credits are provided using a system expansion approach that incorporates offsets. By expanding the system boundary to include an estimated quantity of emissions avoided due to a component of the waste management activity (i.e., electricity generation from LFG), the USEPA model reduces emissions allocated to the waste activity by that quantity (i.e., emissions that would have otherwise occurred had, for example, the electricity been produced from fossil-based electricity generation). Sources of credits in the WARM model include: 1) using recycled (rather than virgin) content; 2) electricity generated from waste management practices; 3) carbon stored in soil from compost; 4) sequestration of biogenic carbon in landfills. These all have varying degrees of uncertainty associated with them; for example the model assumes an infinite timeframe for the landfill credit though future disturbances to landfill sites, such as landfill mining, that may oxidize this carbon (such as through combustion or biodegradation). Additionally, from a management perspective, credits can shift the focus away from current CH$_4$ emissions, which is problematic as CH$_4$ is a potent GHG with a strong, short-term effect on radiative forcing (IPCC, 2007).
The methodologies examined allow varying amounts of flexibility for considering jurisdiction-specific conditions. Generally speaking, average/default values are applied for the comparison of the models, leaving some uncertainty in the results.

2.2 Methodology

2.2.1 Greater Toronto Area (GTA)

The GTA is comprised of five regional municipalities: City of Toronto, Peel, Halton, Durham and York. The GTA is selected as the study region in contrast to solely examining the City of Toronto, for two reasons; firstly, waste is a regional issue with waste management operations being utilized by multiple municipalities within the region. Secondly, this complements a study performed by Kennedy et al., (2009a) on regional GHG emissions and follows their methodology of examining a major urban centre along with its neighbouring communities whose economies are interdependent (Kennedy et al., 2009a). In 2006, the population of the GTA was estimated at 5,556,182, with 45% of residents centrally located in the City of Toronto. It is estimated that, on average, GTA residents sent 210 kg of MSW to landfill per capita in 2005, compared to the national and provincial residential averages of 290 and 305 kg, respectively (see Table 2.1) (Statistics Canada, 2010).

<table>
<thead>
<tr>
<th>Waste Disposal Method</th>
<th>Tonnage</th>
<th>Tonnes Per Capita</th>
<th>Comments</th>
</tr>
</thead>
<tbody>
<tr>
<td>Waste Landfilled (MC approaches)</td>
<td>1,154,981 (^{a,b,c,d})</td>
<td>0.210</td>
<td></td>
</tr>
<tr>
<td>Waste Composted</td>
<td>188,700 (^{a,d})</td>
<td>0.034</td>
<td></td>
</tr>
<tr>
<td>Waste Incinerated</td>
<td>91,000 (^{a,d})</td>
<td>0.016</td>
<td></td>
</tr>
<tr>
<td>Waste AD’d</td>
<td>72,448 (^{d})</td>
<td>0.013</td>
<td>(^{a})Allocated to “Waste Composted”</td>
</tr>
<tr>
<td>Backyard Compost</td>
<td>69,888 (^{d})</td>
<td>0.013</td>
<td>(^{d})Assumed to be carbon neutral</td>
</tr>
<tr>
<td>Recycled</td>
<td>446,719</td>
<td>0.080</td>
<td></td>
</tr>
</tbody>
</table>

\(^{a}\) Applied to WARM Model, \(^{b}\) Applied to FCM-PCP Model, \(^{c}\) Applied to IPCC 1996, \(^{d}\) Applied to IPCC 2006

Prior to the mid 1960s, waste management strategies were guided by a mélange of municipal policies across the GTA (City of Toronto, 1980; Anderson, 1997). Incineration was the primary
means of waste management up until the mid-1960s; however, incinerator capacity was frequently less than waste production. Up until 1965, emergency landfills set up in public ravines were used in the City of Toronto to handle the excess waste. In addition, private dumps, which often partook in open burning, were prevalent. In order to address this patchwork disposal system, large peri-urban landfills were planned and commenced operation in 1967. However, as a result of the diverse waste management schemes across the GTA, obtaining accurate waste disposal data from the era prior to large scale landfill sites is difficult.

The earliest landfill waste figures are from City of Toronto archives, where waste disposed in the four major regional landfills between 1971-1979 are available (Beare Road, South Thackeray, Brock West and Brock North; City of Toronto, 1980). Landfilled waste data between 1955 and 1970 are extrapolated based on per capita waste produced in 1971 and census data. This is an acceptable approach as emissions from waste deposited prior to 1971 will be relatively small; as a result, error in this term will have a minimal impact on the 2005 emissions. Waste data for odd numbered years between 1981 and 1989 are obtained from Metro Toronto Planning Dept publications (1981, 1983, 1985, 1987, 1989) and gaps between these data and 1989 – 1999 are linearly interpolated (Figure 1). Data from 1999 – 2005 are obtained from Kennedy and others, as well as from regional data (Kennedy et al., 2007; City of Toronto, 2005; Barton, 2009; November, 2009; Watson, 2009; Durham, 2009). All data obtained prior to 1999 include industrial, commercial and institutional (ICI) waste; hence, a correction factor of 0.36 (representing the proportion of ICI waste reported in Ontario in 2006) is applied to these (Statistics Canada, 2008).
A drop in the quantity of Peel landfill waste in 1993 is assumed, due to the introduction of a Waste-to-Energy incineration operation (assumed to have a capacity of accepting 80 kt of waste yr\(^{-1}\)). A steep upward slope in the growth of waste is observed during the 1980s. This is likely attributable in part to population growth, coupled with the closure of incinicators during that period. However, given that incinerators accounted for 200,000 tonnes of waste in 1981 and the rate of population growth does not seem to differ much from other decades, this may not provide a complete explanation. A similar spike is observed by Anderson, looking at Metro Toronto and industrial waste; however, waste production from the former City of Toronto (which represented the central component of the former Metropolitan area) did not rise as quickly, suggesting that this increase is mainly attributable to the Industrial, Commercial, and Institutional (ICI) sector, perhaps due to the closure of incineration facilities previously accepting this sector’s waste (Anderson, 1997). This would imply that the correction factor of 0.36 applied to pre-1999 waste would no longer be valid; however, since a more suitable ICI correction factor for waste

**Figure 2-1: Waste disposed in landfills from the GTA between 1955-2005**
deposited before 1990 is unknown, and the contribution to LFG emissions from this waste is relatively small, this change is not incorporated. Waste GHG Emissions Models

Landfilling waste is the dominant treatment method in the GTA, followed by recycling, composting, incineration and AD. The proportion of waste that is from single family housing compared with multi-unit dwellings is obtained from census data (Statistics Canada, 2010). Parameters applied to the four methodologies and the two variations on IPCC 2006 are displayed in Table 2.2, along with applicable sources (some of which are discussed further in the specific methodologies below). Calculation methods for incineration- and composting-related emissions were only available in IPCC 2006 and USEPA WARM, while AD emissions calculations are only possible for the former; IPCC 1996 & FCM-PCP do not provide a means of quantifying these. For the waste composition calculations, differentiation is made between single family and multi-unit dwellings; these are taken from City of Toronto data and assumed to be uniform across the region (except for the Region of Peel incineration calculation; see GTA background above) (Stewardship Ontario, 2009). The smaller contribution of waste > 10 years old to current emissions (see Results & Discussion) validates this assumption.

**Table 2-2:** Parameters applied to the IPCC 1996, 2006 and/or USEPA WARM for 2005 Waste Emission GHG quantification

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Value</th>
<th>Sources / Comments</th>
</tr>
</thead>
<tbody>
<tr>
<td>Degradable Organic Carbon (DOC) fraction</td>
<td>0.169(^b), 0.161(^c)</td>
<td>Using IPCC 1996, 2006 DOC defaults weighted based on waste audits (Stewardship Ontario, 2009); Carbon content based on IPCC defaults (IPCC, 1996; 2006)</td>
</tr>
<tr>
<td>DOC dissimilated (DOC(_f))</td>
<td>0.5(^b,c)</td>
<td>IPCC (1996); IPCC (2006)</td>
</tr>
<tr>
<td>Fraction of CH(_4) in LFG (F)</td>
<td>0.5(^b,c)</td>
<td>IPCC (1996); IPCC (2006)</td>
</tr>
<tr>
<td>Fraction of LFG Recovered (R)</td>
<td>0.75(^a,b,c)</td>
<td>USEPA (2006)</td>
</tr>
<tr>
<td>Half-life of DOC in Waste, years (t(_{1/2}))</td>
<td>9.58(^c)</td>
<td>Weighted based on waste stream calcs (see DOC), using IPCC (2006) defaults for waste half-lives</td>
</tr>
<tr>
<td>Fraction of Landfill CH(_4) Oxidized (OX)</td>
<td>0.1(^b,c)</td>
<td>IPCC (1996); IPCC (2006)</td>
</tr>
<tr>
<td>CH(_4) Global Warming Potential</td>
<td>25(^b,c)</td>
<td>IPCC (2007); GWP(_{100})</td>
</tr>
<tr>
<td>Grid Emissions Factor (g / kWh)</td>
<td>210(^c)</td>
<td>Environment Canada (2009)</td>
</tr>
<tr>
<td>N(<em>2)O GWP(</em>{100})</td>
<td>298(^c)</td>
<td>IPCC (2007)</td>
</tr>
<tr>
<td>Incineration Electricity Generation (kWh / t)</td>
<td>480(^c)</td>
<td>Dennison, (1996)</td>
</tr>
<tr>
<td>CH(_4) Leakage, AD Facilities</td>
<td>5(^c)</td>
<td>IPCC (2006)</td>
</tr>
</tbody>
</table>

\(^a\)Applied to WARM Model, \(^b\) Applied to IPCC 1996, \(^c\) Applied to IPCC 2006
Of the parameters listed above, default data are generally used with the exception of those relating to emission reduction credits discussed in the IPCC 2006 LC approach (specifically, average grid emissions and incineration electricity generation, which are calculated for GTA-specific conditions). Electricity generation from waste treatment options assumes a 46% conversion efficiency of total methane captured (using a reciprocating engine) and a lower heating value of 50 MJ / kg (Harvey, 2010). Methane production is multiplied by capture efficiency to provide the figure for total weight of CH$_4$ captured, with landfills that have received GTA waste are assumed to be equipped with LFG capture systems (assumed to be collecting 75% of LFG) with electricity generation. GHG emission reduction credits (or offsets) are applied for electricity produced from treatment options and generation is assumed to be continuous (Watson, 2009), allowing a 2005 provincial average emissions factor to be used. Specific transportation-related related parameters are described in Table 2.3.

Table 2-3: Relevant parameters applied in calculating GHG emissions from waste collection$^3,7$

<table>
<thead>
<tr>
<th>Truck Capacity (t)</th>
<th>Fuel Consumption (L/100 km)</th>
<th>Energy Density of Diesel (MJ/L)</th>
<th>Diesel Emissions Factor (t/TJ)</th>
</tr>
</thead>
<tbody>
<tr>
<td>34$^a$</td>
<td>42$^a$</td>
<td>35.8$^b$</td>
<td>74.1$^b$</td>
</tr>
</tbody>
</table>

$^a$City of Toronto (2007); $^b$IPCC (2006)

Regarding specific treatment methods, it is assumed that no emissions result from backyard composting (assumed aerobically degraded). All green bin waste is assumed to be anaerobically digested at central processing facilities.

2.2.2 FCM-PCP

In order to assist municipalities in compiling GHG inventories, a spreadsheet tool is provided by the FCM entitled “Inventory Quantification Support Spreadsheet”, which is based on PCP GHG software (FCM, 2009). The calculation for annual GHG emissions is based on an emissions factor (see Equation 2.1 below), and is assumed to be based on national average data, though this could not be confirmed. Using a simple “emissions factor” calculation in a GHG emissions system as complex as waste cannot provide the flexibility of the other more detailed methodologies described below. However, this is simply a preliminary figure for municipalities to use and it is of interest for comparison with other more rigorous methodologies.

\[
GHG \text{ emissions (} t \, CO_2e) = t \, \text{of waste landfilled} \cdot 0.4817 \, t \, CO_2e / t \, \text{of waste landfilled (2.1)}
\]
2.2.3 IPCC 1996

As stated earlier, the IPCC 1996 uses a MC approach for GHG emissions quantification. Emissions can be calculated using (adapted from IPCC, 1996):

\[
CH_4 \text{ emissions} = (W \times MCF \times DOC \times DOC_F \times F \times \frac{16}{12}) \times (1 - R) \times (1 - OX) \quad (2.2)
\]

where 
- \(W\) = total weight of waste deposited in landfills (Gg yr\(^{-1}\));
- \(MCF\) = Methane Correction Factor (for sanitary landfills = 1);
- \(DOC\) = Degradeable Organic Carbon;
- \(DOC_F\) = Fraction \(DOC\) dissimilated;
- \(F\) = Fraction of \(CH_4\) in LFG;
- \(\frac{16}{12}\) is the stoichiometric conversion of carbon to methane;
- \(R\) = Fraction of \(CH_4\) Recovered (i.e. LFG capture efficiency);
- \(OX\) = Fraction \(CH_4\) Oxidation.

2.2.4 IPCC 2006

The IPCC 2006 method involves the most complex calculation of the four landfill methodologies examined. A first-order decay model (Tier 2) approach is employed, using default parameters and region-specific landfill data. The WIP calculation uses sequential calculations each year, employing the equations outlined below (IPCC, 2006):

\[
DDOCm = W \times DOC \times DOC_F \times MCF \quad (2.3)
\]

where \(DDOCm\) = mass of decomposable \(DOC\) deposited in the landfill

\[
DDOCma_T = DDOCmd_T + (DDOCma_{T-1} \times e^{-k}) \quad (2.4)
\]

where \(DDOCma_T = DDOCm\) remaining after a given year (T); \(DDOCmd_T = DDOCm\) deposited in year T; \(k\) = reaction constant \([\ln(2)/t_{1/2}\) (yrs\(^{-1}\))]; \(t_{1/2}\) = half-life of waste (yrs)

\[
DDOCm decomp_T = DDOCma_{T-1} \times (1 - e^{-k}) \quad (2.5)
\]

where \(DDOCm decomp_T = DDOCm\) decomposed in year T

\[
CH_4 \text{ generated} = DDOCm decomp_T \times F \times \frac{16}{12} \quad (2.6)
\]

\[
CH_4 \text{ emitted} = (CH_4 \text{ generated} - R) \times (1- OX) \quad (2.7)
\]
CO₂ emissions from landfill are associated with the degradation of biogenic carbon, resulting in a carbon neutral process. Consequently, these are not included in emissions calculations here, assuming that biogenic carbon stored in paper or harvested wood products would have been tabulated as an emission in any upstream inventories for their materials. Additionally, their contribution would be relatively small compared with CH₄. Emissions of CO₂ would be (in Gg of CO₂)

\[
\text{CO}_2 \text{ generated} = DDOC_{\text{decomp}} T (1-F) \cdot \frac{44}{12} + (CH_4 \text{ generated - R}) \cdot OX \tag{2.8}
\]

Waste composition is assumed to be constant for historical data, and hence, the degradable organic carbon (DOC) content is the same for all years used for the WIP calculation. Disposal is assumed to be at the beginning of the year, with methane emissions calculated at year’s end. DOC is weighted according to the IPCC 2006 fractions for waste components. Wastewater sludge deposited in landfills is assumed to be in the form of biosolids that are stabilized to the extent where further decomposition is negligible.

In year 0 (i.e., 1955), \( DDOC_{\text{ma} T-1} \) and \( DDOC_{\text{m.decomp}} T \) are assumed to be nil, giving a value of \( DDOC_{\text{ma} T} \) that is simply the amount of decomposable DOC deposited in 1955. This was used as a basis for calculations in all following years. The reaction constant \((k)\) is estimated assuming that waste is deposited in a dry, boreal region, using Environment Canada data on Toronto’s mean annual precipitation (MAP) and relating it to potential evapotranspiration (PET; MAP/PET<1), both obtained from Environment Canada (2003a; 2008).

A MC calculation is completed in the same manner (denoted IPCC 2006 MC), using the above equations, except in this case year 0 is 2005, with a 100-year forecast for resulting methane emissions (a 75% LFG capture efficiency is assumed for the lifetime of the waste). This is assumed to be a sufficiently long forecast since by the year 2105, methane emissions are estimated to be below 25 tCO₂e year⁻¹ (Figure 2.2).
Calculations of IPCC 2006 for other disposal methods only require knowledge of total waste tonnage and composition (Tier 1). Equation 2.8 provides an estimate of emissions from fossil carbon from incineration. As only the Region of Peel treats waste through combustion, and this waste is generally sourced from multi-residential units, waste audits for Peel are used to quantify the relevant waste composition (Barton, 2009).

\[
CO_2 \text{ emissions (t CO}_2\text{e)} = MSW \cdot \sum_j (WF_j \cdot dm_j \cdot CF_j \cdot FCF_j \cdot OF_j) \cdot \frac{44}{12}
\]  

where \(MSW\) = total wet weight incinerated, Gg yr\(^{-1}\); \(WF_j\) = fraction of component \(j\) in the MSW; \(dm_j\) = dry matter in component \(j\) (fraction); \(CF_j\) = fraction of carbon in dry matter of component \(j\); \(FCF_j\) = fossil carbon fraction in of component \(j\); \(44/12\) = conversion factor from C to CO\(_2\). All calculations for dm, CF and FCF used IPCC 2006 defaults.

The IPCC 2006 methodology suggests that both CH\(_4\) and N\(_2\)O are released during the composting process (specifically large-scale operations, inferred from references cited within the IPCC guidelines), while emissions from residential-scale compost (i.e. home composting units)
do not receive any explicit mention. Composting and AD calculations for CH$_4$ and N$_2$O emissions require only tonnage treated and IPCC defaults and are described in Equation 2.9. According to this methodology, N$_2$O emissions from anaerobic digesters are assumed to be negligible; IPCC 2006 cautions that more data on these emissions are needed. IPCC 2006 suggests using a 5% leakage rate for AD facilities. Thus, CH$_4$ or N$_2$O emissions are given by

$$\text{GHG emissions (t CO}_2\text{e)} = [(M \cdot EF) \cdot 10^{-3} \cdot R] \cdot GWP_{100}$$

(2.10)

where $M$ = wet weight of waste treated (t); $EF$ = emissions factor (kg (t waste treated)$^{-1}$; 4 for CH$_4$ compost; 0.3 for N$_2$O compost; 1 for CH$_4$AD); $R$ = gas recovered (0 for composting; 95% for AD); $GWP_{100}$ = Global Warming Potential based on a 100-year timeframe (25 for CH$_4$; 298 for N$_2$O; IPCC, 2007).

### 2.2.5 USEPA WARM

Tonnage, composition, and diversion rate details are integral to the usage of the WARM model. Using waste audits and diversion rates for 2005, data are entered for the various required component streams. Stewardship Ontario data are categorized according to the waste inputs available in the USEPA model (USEPA, 2009; Stewardship Ontario, 2009). The data and method of application are available in Appendix A (see Tables A.1, A.2 and A.4).

Key assumptions of the WARM model include a global warming potential for CH$_4$ of 21, a 75% LFG collection system efficiency, and average national electricity grid emissions factor of 0.17 kg / kWh (or 0.17 t / MWh).

### 2.2.6 Life Cycle-based Approach

A life cycle-based approach (IPCC 2006 LC) is used to include more of the upstream (life cycle) GHG emissions associated with waste management practices in the GTA for 2005 using the IPCC 2006 MC method, with the functional unit being waste managed in 2005. While a larger proportion of life cycle emissions associated with waste management is included in this method than with the IPCC 2006 MC, a full life cycle inventory analysis is not completed. The boundaries for the IPCC 2006 LC approach are presented in Figure 2.3, using credits/emissions applicable to a municipal inventory (use of incineration residues for fertilization in forestry has been reported by Toller et al., 2009). Specifically, emissions included are those related to the
collection and transportation of waste to treatment sites and those associated with the treatment options themselves. The exclusion of upstream emissions of fuels will have a negligible impact on results, since transportation of waste materials is generally a lower proportion of total waste-related emissions (Mohareb et al., estimate a contribution of 8% of gross emissions or 15% of net emissions including credits for recycling) and combustion is the primary source of these emissions when diesel is used as a fuel (Finnveden et al., 2005; Mohareb et al., 2008; MacLean & Lave, 2003). Emission reductions from co-products directly resulting from on-site activities of treatment methods (i.e. electricity production from incineration) are included within the LC boundary as well.

**Figure 2-3:** Flowchart displaying boundaries for IPCC 2006 LC

IPCC 2006 is selected for this approach, as it allows for the quantification of emissions from landfill, AD, incineration and large-scale composting. Using an IPCC method with some scope for life cycle emissions also allows comparison of a designated emissions inventoring with USEPA WARM (which is explicitly stated to be incompatible with emissions inventoring), as it uses a life cycle approach. A point of note is that WIP cannot be used as the means to quantify emissions from landfilled waste, as this would not conform to the temporal boundary set by examining waste collected within 2005.
No GHG emissions reduction credits for recycling are allocated to GTA municipalities. While recycling credits may be suitable on a national level, there is little certainty that materials diverted for recycling will actually be processed and used within the same spatial boundary being assessed (Cleary, 2009a). From a life cycle inventory perspective, the location of an activity would not, in itself, provide justification for exclusion, but this was deemed appropriate in the context of municipal inventories; since this study focuses on emissions and credits applicable to municipalities based on policy decisions, exclusion of these credits is reasonable as the decision on reuse of recycled material is beyond municipal jurisdiction. In addition to the uncertainty associated with where the co-products will be used, Finnveden (1999) illustrates complications that arise when materials are not recycled directly into the same product (termed open-loop recycling). Allocation procedures differ for the emissions related to the original product and those incorporating recycled content. Some estimates on potential credits associated with recycling are provided in the “Results & Discussion” section.

Emissions reductions from co-products serving as fertilizer/soil conditioner are also excluded due to the uncertainty in their destination and final use (i.e. potential contamination may prevent their usage). Finnveden et al. (2005) suggest that GHG emission benefits from fertilizer displacement from anaerobic digestion and composting are also likely small.

Emissions from capital infrastructure are ignored; there is precedence for this as Cleary (2009b) states that only three of the 20 waste LCA studies he reviewed included these emissions. However, energy requirements from operations are considered. Denison (1996) provides a figure for net energy generated for incineration, while landfill operations utilize roughly 15% of energy generated for internal operations, which is applied to the IPCC 2006 methodology (Franklin Associates, 1994). It is assumed that the latter figure is likely a mixture of diesel, electricity and natural gas in the GTA; however for simplicity, a 15% penalty is applied to landfill gas electricity generation and is also applied to electricity generation at AD facilities (it should be noted that this penalty would be much greater if diesel had been used exclusively). Composting operations energy requirements are assumed to be negligible.

The IPCC 2006 LC approach examined in this work includes transportation for waste and grid emissions factors (applied during system expansion to include for offsets for electricity production. Transportation distance calculations follow the methodology used by Mohareb,
using distances from the approximate geographic centre of an urban area (as opposed to city hall) to landfills, incinerators, anaerobic digesters and material recovery facilities (for recycling) (Mohareb et al., 2005).

Grid emissions factors applied in the system expansion approach for landfill, AD and incineration operations represent the marginal emissions that would have otherwise occurred from the electricity generation. Finnveden et al., (2005) suggests that a marginal source of electricity (coal) is displaced by electricity from waste, whereas Cleary (2009b) observed an even split in 12 studies between the use of marginal and average electricity source emissions factors (Finnveden et al., 2005; Cleary, 2009b). In a situation where \( \text{CH}_4 \) storage is possible (or \( \text{CH}_4 \) is flared when demand does not exist) and is used only to meet a fluctuating load or as spinning reserve for the electrical grid, use of the emissions factor for the displaced marginal generation is logical. Conversely, if LFG is combusted as produced then it supplies baseload generation and use of the average grid emissions factor is preferred.

2.2.7 Sensitivity Analysis

Selection of the parameters described in Table 2 is made based on default data used in other literature, but regional specifications (such as factors related to the GTA’s climatic zone) are applied where available. However, there is some uncertainty in many of these quantities and this is addressed in a sensitivity analysis.

Uncertain treatment-specific factors considered in this study include oxidation of \( \text{CH}_4 \) (landfill), concentration of \( \text{CH}_4 \) in LFG, carbon content of waste (landfill, incineration), fraction of carbon dissimilated (degraded in landfills), reaction constant \( (k; \) relevant to first-order decay models for landfills) and methane leakage (AD, landfill with LFG capture). Oxidation of \( \text{CH}_4 \) in LFG due to use of specialized covers (other than clay) has reduced emissions from 10 – 100%, varying due to site and climatic conditions (Lou & Nair, 2009). An Alberta, Canada study suggested that the rate of oxidation is dependant on \( \text{CH}_4 \) flow rate, suggesting that the value of \( k \) may influence oxidation (Stein & Hettiaratchi, 2001). The USEPA and IPCC (1996 and 2006) both make the assumption of 10% oxidation using aerating covering material. While this may seem low in light of the range suggested above, the more conservative estimate is prudent in the absence of site-specific data.
LFG CH$_4$ fractions are also somewhat uncertain, with the IPCC default being 50% while the fraction recorded at Brock West, Beare Rd and Keele Valley landfills in 2001 were roughly 40%, 45% and 47% (Environment Canada, 2003b). Impacts of modifying LFG CH$_4$ concentrations are assessed in the sensitivity analysis.

The leakage rate of LFG is also a point of contention in literature. In the WARM model, a default assumption of 75% capture rate is assumed as the national average efficiency. The Keele Valley landfill site (C40 Cities, 2009) estimates a collection efficiency of between 85-90% (high, but not infeasible according to Barlaz et al., 2009), while Mohareb (2008) reports 40% for the Trail Rd landfill in the Ottawa Region. A value of 50% is selected for sensitivity analysis versus the 75% baseline suggested by the USEPA (USEPA, 2006).

The carbon content of waste is region-specific and can be approximated using waste audits (such as those provided by Stewardship Ontario) and default values of carbon contents of various waste components (IPCC, 1996; IPCC, 2006; Stewardship Ontario, 2009). The range of the IPCC (1996) North American values is used for the sensitivity analysis.

The fraction of biogenic carbon that can actually be dissimilated is also a matter of debate. Barlaz (1998) suggests that roughly 40% of carbon in MSW does not decompose under anaerobic conditions, while the IPCC default suggests using a value of 50% of total degradable carbon (IPCC, 2006).

The reaction constant, $k$, is sensitive to the climatic conditions and composition of the waste deposited in landfill, amongst other factors. For example, some landfills have been operated as bioreactors, with recirculation of leachate in order to increase the reaction constant (Benson, 2007). This parameter has not been assessed since the latter has no impact on total emissions (such as for the MC method).
2.3 Results & Discussion

2.3.1 Model Comparison - Landfill Waste

As the principle source of GHG emissions, it is of most interest to compare the results for landfill emissions from the six approaches examined and shown in Figure 4. Four MC calculations are provided, as well as the IPCC 2006 LC and the IPCC 2006 WIP calculation. FCM-PCP, IPCC 1996 & IPCC 2006 (MC & WIP) figures given below are gross site emissions (without transportation emissions or offsets for electricity generation), while WARM and IPCC 2006 LC calculations are net emissions. This is because WARM and IPCC 2006 LC incorporate the offsets, as well as transportation emissions. The estimates for landfill GHG vary from an emission of ~556 kt (FCM-PCP) to a net carbon sink of 53 kt (WARM).

![Figure 2-4: 2005 GHG Emissions (t CO2e) from LFG Released from Sites Handling GTA Waste Quantified using Six Distinct Approaches](image)

Figure 2.4 can be used to illustrate some of the strengths, weaknesses and applications of each model. Firstly, while the FCM-PCP model likely overestimates GHG emissions due to its
inflexibility and relatively high landfill waste emissions factor (0.4817 t CO\textsubscript{2}e (t waste))\textsuperscript{-1}, compared with 0.302 t CO\textsubscript{2}e (t waste)\textsuperscript{-1} from the IPCC 2006 MC) it can be considered a reasonable “first guess”, given that emissions from the IPCC 2006 MC method are within the 30% of this estimate.

Secondly, IPCC MC methodologies provided similar results (with the 1996 calculation being 5% greater than the 2006 approach), suggesting that professional judgment be used in considering whether to employ the slightly more detailed waste stream quantification required in the IPCC 2006. As well, if one were to simply apply the median value of the default DOC range provided for North America in IPCC 1996 (0.18-0.21; i.e. using a DOC value of 0.195), the difference compared with the IPCC 2006 MC method increases to 18%. Using the median value could provide an acceptable approximation in this case if one were willing to tolerate a difference of this magnitude and there were no other known factors that would cause the value for the city in question to differ. This allows the quantification of the waste MC emissions without having to quantify waste stream components using audit data, if municipal waste audit data are unavailable or difficult to obtain. Assurance can be taken from greater diligence; however, the degree of accuracy that is necessary and cost limitations should be factored into the decision if a waste audit will be required to obtain waste stream information.

Thirdly, differences are evident in the IPCC 2006 WIP and MC estimates, and though WIP can be given more weight from an inventorying perspective as it quantifies emissions occurring in the inventory year, rather than projecting future emissions (there is uncertainty in the historic mass of waste and its composition applied to WIP, as well). It must be noted that the correspondence of these two values is case specific (as it would be in any of these approaches); landfilled waste tonnage has been relatively stable during the past decade (a slight decline in recent years gives a lower MC value), coupled with other parameters being assumed constant (such as LFG capture for MC or DOC for WIP), resulting in the similar quantities obtained. Uncertainty in the future landfill management practices clouds the accuracy of emissions quantified by MC.

From an economic perspective (i.e., discounting), future emissions may have less value than GHGs released at present. From a climatic perspective, and within in the context of a municipality with an increasing organics diversion rate, using the MC projection for an inventory
underestimates CH$_4$ emissions occurring at present. However, developing countries that increasingly use sanitary landfills for waste disposal will experience a rise in waste-related GHG emissions (IPCC, 2006); if a MC method is selected for inventoring purposes rather than WIP, a greater emission estimate will result. It follows that WIP-approach quantification would give a lower estimate when compared to MC, due to the diminished contribution from waste deposited in previous years that would have occurred otherwise if open dumping or a semi-aerobic disposal were used.

The USEPA WARM model is a clear outlier of the models assessed. This is principally due to the provision of carbon credits for the sequestration of organic carbon. Under aerobic conditions, it is assumed that biogenic carbon breaks down completely, releasing atmospheric CO$_2$ which had been previously captured during photosynthetic processes. However, as stated previously, not all carbon is dissimilated in the anaerobic environment present in an undisturbed landfill (IPCC, 2006; USEPA, 2006 – See Appendix A, Tables A.1, A.2, & A.4). Consider a tonne of biogenic waste of which 50% is carbon; assuming 50% of that carbon is degraded anaerobically to CH$_4$, 10% of it is oxidized in the landfill cover, and 75% of the remainder is captured and flared. This gives a figure of 5.6% of the landfilled carbon being released as methane. Accounting for the molar weights (which would require the multiplication of 12/44 by that released fraction) and assuming that methane is 25 more potent than CO$_2$, it is theoretically possible using these assumptions that more resultant carbon storage is greater than the release of methane ($5.6\% \times 12/44 = 0.015 < 1/25$). Hence a significant carbon sink, compared to the aerobic degradation base-case, is created in landfills. When this is coupled with emissions offset by electricity generation from captured LFG, greater net carbon storage results.

The concern regarding difficulty in obtaining accurate historic waste data may be of little importance. The IPCC suggests that waste data from at least 10 years prior are required for use of the 2006 method. Looking at the contribution from waste deposited prior to 1995, this is roughly 12% of 2005 WIP emissions, given the methane generation rate calculated for the GTA. This contribution will increase for regions where the reaction constant ($k$) is lower (drier climates or where greater proportions of slower degrading materials such as wood and paper waste are landfilled). For example, if using upper estimates for half-life of waste in landfills located in boreal/dry climates, the contribution of waste older than a decade would increase to 16%. For warmer, wetter climates the effect of these earlier data will diminish, adding greater incentive for
cities to use the WIP approach when used for inventorying purposes. In cases where obtaining historic waste disposal data is difficult, estimations for waste deposited based on population trends (using per capita waste) will likely meet the requirements of most applications.

There is certain value for all of the LFG models assessed above, such as ease of use (FCM-PCP) or increased rigor (IPCC 2006 WIP). The simplicity provided by the MC models can definitely be appreciated in circumstances where time or resources are constraints; however, greater adherence to inventorying goals (i.e., consistent emissions temporal boundaries) is achieved with the IPCC 2006 WIP model since there is more parameter flexibility and fewer assumptions inherent in its design.

### 2.3.2 Life Cycle-based Approach to Waste Emissions

The IPCC 2006 LC approach is used in order to quantify some key credits that are within municipal spatial boundaries and further emissions attributable to each waste management activity. This approach underlines the relative importance of landfill emission quantification, as LFG emissions provide the greatest share of the total.

Under the IPCC 2006 LC approach, gross emissions from waste management practices in the GTA are shown in Table 2.4, using the MC calculation for landfill. Total emissions in 2005 using this methodology were estimated to be 509 kt CO$_2$e. When applying a credit for carbon emissions offset by electricity generation from waste, net emissions are reduced to 441 kt CO$_2$e, although this would not be included in standard GHG emission inventorying practice (not to be confused with life cycle inventory practice); while emissions may indeed be reduced, credits for emissions offsets are not applied toward totals in GHG inventories, such as those provided in national inventory reports (IPCC, 2006).

Table 2.4 details the specifics regarding gross and net emissions for each treatment option. The data are in agreement with Finnveden (2005) and Mohareb et al., (2008) as transportation-related emissions have a relatively minor impact on the total (contributing less than 10% to total emissions). Even if total transportation distance is doubled to account for any underestimation made in distance travelled to waste facilities, it would only contribute slightly more than 13% to total net emissions.
Table 2-4: Gross and Net 2005 Emissions from Waste Management Activities using IPCC 2006 Method of Calculation

<table>
<thead>
<tr>
<th>Treatment Option</th>
<th>Gross Emissions (t CO₂e)</th>
<th>Per Tonne Disposed</th>
<th>Emissions Offset¹ (t CO₂e)</th>
<th>Net Emissions (t CO₂e)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Landfill (MC)</td>
<td>348,300</td>
<td>0.302</td>
<td>57,000</td>
<td>291,000</td>
</tr>
<tr>
<td>AD</td>
<td>100</td>
<td>0.001</td>
<td>320</td>
<td>-220</td>
</tr>
<tr>
<td>Incineration</td>
<td>29,800</td>
<td>0.327</td>
<td>9,200</td>
<td>20,600</td>
</tr>
<tr>
<td>Composting</td>
<td>75,100</td>
<td>0.398</td>
<td>N/A</td>
<td>75,100</td>
</tr>
<tr>
<td>Transportation</td>
<td>30,500</td>
<td>N/A</td>
<td>N/A</td>
<td>30,500</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td><strong>508,600</strong></td>
<td></td>
<td><strong>67,600</strong></td>
<td><strong>441,100</strong></td>
</tr>
</tbody>
</table>

¹Calculated using average emissions; if marginal emissions offsets from OPA (2011) estimates were used, total landfill offsets would be 142 kt, 1 kt, and 25 kt for electricity produced from landfill, AD and incineration, respectively.

AD is the only management option that produces net negative emissions (direct minus electricity offsets); if transportation emissions were disaggregated and added to AD facilities emissions, net emissions would be revised to roughly zero. While one might expect higher emissions due to the relatively high leakage rate suggested by IPCC 2006 guidelines, the low emissions values resulting from default parameters are likely due to the relatively high moisture content of the waste deposited in AD (predominantly source separated organics) when compared to landfilled waste which includes components with higher carbon contents (e.g. greater proportion of forestry products). Composting provides a very high emissions result in relation to incineration and landfilling, especially when comparing net emissions. Composting emissions could be even greater when considering that backyard composting is suggested to result in N₂O and CH₄ emissions that would not be negligible; Amlinger et al., (2008) suggest that each tonne of wet waste could result in the emission of 76 – 187 kg CO₂e (or up to 0.45 kg N₂O and 2.2 kg CH₄ per tonne of wet waste deposited in backyard composting units). It is also possible that properly managed composting systems would have lower GHG emissions than have been estimated using IPCC default emissions factors.

Additionally, relatively high GHG emissions are associated with incineration. When one considers that, for direct (excluding transportation and electricity generation) emissions, 90 kt of incinerated waste resulted in 29.8 kt of gross GHG emissions and 1,150 kt of landfilled waste resulted in 348 kt of GHG emissions according to the IPCC 2006 MC calculation, emissions per unit of waste treated are 9% higher for incineration compared with landfill. When including offsets for energy generation for both landfill and incineration, the net emissions from landfills
are only 11% higher per tonne of waste treated. This is a conservative estimate given that the Ontario government has proposed the replacement of all coal-fired generating stations with renewable and natural gas-fired generation by 2014, with 40% of the 2003 coal generation capacity being taken offline by the end of 2010 (Government of Ontario, 2010). Using a lower emissions factor (i.e. reducing the emissions factor by 1/3), landfill emissions are only 4% higher than incineration per tonne of waste treated.

It may be of interest to briefly examine the emissions reductions potential from recycling, although this was beyond the scope of the LC approach. Mohareb et al., (2008) suggest a virgin material displacement credit of approximately 1.04 t CO$_2$e per tonne of mixed material recycled, while the USEPA (2006) suggest 0.85 t CO$_2$e (excluding transport and process non-energy), giving a credit of 464 and 380 kt CO$_2$e, respectively, for the nearly 447,000 tonnes of waste diverted from the GTA for 2005.

2.3.3 Comparison of Net GHG Emissions

IPCC 2006 MC and the WARM model were both used to calculate net annual GHG emissions (including offsets from electricity generation and emissions from transportation) for different waste treatment options (Figure 2.5). Net emissions from landfills increase slightly when using a more conservative figure for the efficiency of the reciprocating engine used to generate electricity from LFG; Lombardi et al., (2006) suggests an efficiency of 35% (vs. the efficiency of 46% applied here; see Table 2.2), which would cause the net efficiency for the IPCC 2006 methods to increase by 5%.
Figure 2-5: Gross & Net Annual GHG Emissions (2005) from Various Treatment Options for IPCC 2006, compared with USEPA WARM

The reduction in net emissions is far more substantial for the WARM model than IPCC. WARM provides further credits from the following: 1) A larger credit for electricity offsets is assumed due to the prevalence of coal-fired generation in the US (average emissions factor of 1014 g CO$_2$e (kWh)$^{-1}$ is used and cannot be adjusted), while the IPCC calculation for the GTA scenario examined uses the 2005 Ontario average emission factor (210 g CO$_2$e (kWh)$^{-1}$ where carbon-free electricity (e.g., nuclear, hydro) contributes a greater proportion; Environment Canada, 2009); 2) A significant credit is applied to landfills due to undegraded biogenic carbon; 3) Soil carbon credits are provided for composting (and no CH$_4$ or N$_2$O emissions penalty). If credits for LFG electricity generation are removed, USEPA WARM suggests an 80 kt CO$_2$ emissions source for landfill waste disposal will result.

As stated in section 2.3.5, both marginal and average grid emissions have been applied in previous studies to calculate electricity offsets. Marginal emissions factors used by the OPA have
been provided by in the Appendix (Table A.3; Personal Communication, OPA, 2011). If these emissions factors were applied in place of the average grid emissions, the offset would increase to nearly 142 ktCO$_2$e for the IPCC 2006 MC approach; the net emission that would result would be 206 ktCO$_2$e. The IPCC inventoring approaches calculate methane emissions by assuming that only a portion biogenic carbon deposited in landfills is degraded under anaerobic conditions (using the fraction of carbon dissimilated, DOC$_f$; IPCC, 2006; IPCC, 1996). If one were to assume that all undegraded biogenic carbon from IPCC scenarios would have been oxidized under aerobic conditions, the carbon sink provided by the anaerobic landfill conditions for waste deposited in 2005 is calculated to be 170,300 t CO$_2$e using the IPCC 2006 MC method; this would result in a net emissions value of 120,700 t CO$_2$e, still greater than the WARM figure. Greater flexibility on which sinks to incorporate and parameter values used in the WARM model would improve accuracy and applicability.

The discrepancy in compost emissions also comes from the high default values of the IPCC 2006 CH$_4$ and N$_2$O emissions factors, in addition to the application of carbon credits in the WARM scenario. As stated earlier, Hobson et al., (2005) suggest that GHG production is likely when household waste is deposited in windrows, especially CH$_4$. Quantities of N$_2$O may be lesser; however, due to its greater global warming potential over a 100-year time frame, its effect is more prominent (75% of composting-related GHG emissions). More research is needed in quantifying the emissions of two important GHGs through the composting of MSW in windrows in order to determine the most suitable waste treatment approach.

### 2.3.4 Uncertainty & Sensitivity Analysis

As outlined in the background section, many variables in the quantification of GHG emissions from waste are uncertain. Table 2.5 provides a number of uncertain variables within the methodology, along with the corresponding sensitivity of ranges for these variables according to literature or IPCC ranges (see Methodology for explanation of parameter selection). There is a focus on landfill-related emissions due to their relative significance compared to other emissions sources and the ubiquity of their quantification across multiple methodologies. The FCM-PCP equation does not allow any modifications of parameters other than waste deposited in landfills, which is a relatively certain quantity, and hence is not examined.

**Table 2-5:** Sensitivity to Uncertain Values of 2005 GHG Emissions from Landfill
LFG capture efficiency has the greatest impact on landfill GHG emissions of those demonstrated above, with at least a doubling of emissions from a $\frac{1}{3}$ reduction in LFG collected.

By applying waste audits from the City of Toronto, degradable carbon content was estimated to be 16.1% and 16.9% using default data from IPCC 2006 and 1996, respectively, for carbon content for waste stream fractions. This figure varies based on waste composition (i.e., greater organic content gives a greater degradable carbon content). The IPCC (1996) provides a range of DOC in North American waste of 18 – 21%. The high end of this range would provide an increase in landfill GHG emissions by nearly 25%.

Variation of oxidation potential of landfill cover is examined using data provided by Stein and Hettiaratchi (2001), who report a methane oxidation rate of 20% at a flow rate of 400g CH\(_4\) (m\(^2\)-day)\(^{-1}\). An increase of 100% in the amount of CH\(_4\) oxidized reduced overall GHG emissions by 10%. Lou & Nair (2009) suggest that oxidation of CH\(_4\) in landfill cover can range from negligible to 100%, so importance should be placed on quantifying this value accurately. It is hence of interest to use site specific measurements of these parameter for reliable inventorying.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Values</th>
<th>IPCC 1996 (tCO(_2)e)</th>
<th>IPCC 2006 (WIP) (tCO(_2)e)</th>
<th>WARM (tCO(_2)e)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>LFG Capture</td>
<td>0.75</td>
<td>365,518</td>
<td>373,120</td>
<td>- 52,841</td>
</tr>
<tr>
<td></td>
<td>0.5</td>
<td>731,037</td>
<td>746,239</td>
<td>270,346</td>
</tr>
<tr>
<td></td>
<td>% Change</td>
<td>100%</td>
<td>100%</td>
<td>612%</td>
</tr>
<tr>
<td>Degradable Carbon</td>
<td>0.17</td>
<td>368,150</td>
<td>394,381</td>
<td>N/A</td>
</tr>
<tr>
<td></td>
<td>0.21</td>
<td>454,774</td>
<td>487,177</td>
<td>N/A</td>
</tr>
<tr>
<td></td>
<td>% Change</td>
<td>24%</td>
<td>23%</td>
<td>N/A</td>
</tr>
<tr>
<td>Oxidation</td>
<td>0.1</td>
<td>365,518</td>
<td>373,120</td>
<td>N/A</td>
</tr>
<tr>
<td></td>
<td>0.2</td>
<td>324,905</td>
<td>331,662</td>
<td>N/A</td>
</tr>
<tr>
<td></td>
<td>% Change</td>
<td>-11%</td>
<td>-11%</td>
<td>N/A</td>
</tr>
</tbody>
</table>

LFG capture efficiency, degradable carbon content, oxidation rate, fraction dissimilated and CH\(_4\) content of LFG are examined in Table 6, based on the uncertainty demonstrated from literature and methodologies. Values are grouped into quantities that increase emissions and those that reduce emissions, providing a high and low case of each. The range of values vary substantially, as demonstrated by the high case for the IPCC 2006 model which is more than 450% that of the low case.
Table 2-6: Estimates of 2005 Landfill GHG emissions for parameter estimates

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Values</th>
<th>IPCC 1996 (tCO₂e)</th>
<th>IPCC 2006 (WIP) (tCO₂e)</th>
<th>WARM (tCO₂e)</th>
</tr>
</thead>
<tbody>
<tr>
<td>LFG Capture</td>
<td>0.75</td>
<td>365,518</td>
<td>373,120</td>
<td>-52,841</td>
</tr>
<tr>
<td></td>
<td>0.5</td>
<td>731,037</td>
<td>746,239</td>
<td>270,346</td>
</tr>
<tr>
<td>% Change</td>
<td></td>
<td>100%</td>
<td>100%</td>
<td>612%</td>
</tr>
<tr>
<td>Degradable Carbon</td>
<td>0.17</td>
<td>368,150</td>
<td>394,381</td>
<td>N/A</td>
</tr>
<tr>
<td></td>
<td>0.21</td>
<td>454,774</td>
<td>487,177</td>
<td>N/A</td>
</tr>
<tr>
<td>% Change</td>
<td></td>
<td>24%</td>
<td>23%</td>
<td>N/A</td>
</tr>
<tr>
<td>Oxidation</td>
<td>0.1</td>
<td>365,518</td>
<td>373,120</td>
<td>N/A</td>
</tr>
<tr>
<td></td>
<td>0.2</td>
<td>324,905</td>
<td>331,662</td>
<td>N/A</td>
</tr>
<tr>
<td>% Change</td>
<td></td>
<td>-11%</td>
<td>-11%</td>
<td>N/A</td>
</tr>
</tbody>
</table>

2.3.5 Assessment of Models

A summary of key model features is presented in Table 2.7. As stated in the introduction, those involved in urban emissions inventorying use a variety of models in their efforts to quantify GHG emissions attributed to activities of residents within their municipalities. After examining the issues associated with the methodologies presented above, a principal categorization can be made; MC models are most valuable from a planning standpoint due to their predictive nature while the WIP model used in IPCC 2006 is most appropriate for conducting emissions accounting (emissions reduction credits for electricity generation must be neglected when reporting for the purpose of inventorying as inventories aim to quantify direct sources and sinks, not assumed derivative impacts). Since planning decisions can alter the values provided by MC models, they have limited usefulness from a reporting standpoint due to their greater degree of uncertainty. However, MC models can be helpful in quantifying the effects of certain landfill management decisions (i.e., measures to reduce LFG emissions) and for evaluating impacts on waste diversion from a global warming perspective (i.e., impacts from diverting waste to incineration). A WIP model can be used to provide similar information to planners, however it is temporally constrained to emissions in the inventory year rather than the entire lifespan of waste deposited in a given year. An additional attraction towards the MC approach comes from its relative simplicity, as data requirements for the WIP model can seem onerous.
Table 2-7: Comparison of Features of Four Models for Quantifying GHGs from Landfills

<table>
<thead>
<tr>
<th></th>
<th>PCP-FCM</th>
<th>USEPA WARM</th>
<th>IPCC 1996</th>
<th>IPCC 2006</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Stated Purpose</strong></td>
<td>Inventorying</td>
<td>Planning</td>
<td>Inventorying</td>
<td>Inventorying</td>
</tr>
<tr>
<td><strong>Model Type</strong></td>
<td>MC</td>
<td>MC</td>
<td>MC</td>
<td>WIP</td>
</tr>
<tr>
<td><strong>Scope</strong></td>
<td>Direct Emissions</td>
<td>LC Emissions</td>
<td>Direct Emissions</td>
<td>Direct Emissions</td>
</tr>
<tr>
<td><strong>LFG Capture Efficiency</strong></td>
<td>Fixed</td>
<td>Variable</td>
<td>Variable</td>
<td>Variable</td>
</tr>
<tr>
<td><strong>Waste DOC Composition</strong></td>
<td>Fixed</td>
<td>Variable</td>
<td>Variable</td>
<td>Variable</td>
</tr>
<tr>
<td><strong>Carbon Sinks</strong></td>
<td>Not Quantified</td>
<td>Quantified</td>
<td>Quantifiable</td>
<td>Quantifiable</td>
</tr>
<tr>
<td><strong>Waste Data Required</strong></td>
<td>1 year</td>
<td>1 year</td>
<td>1 year</td>
<td>10-50 years</td>
</tr>
</tbody>
</table>

As discussed in the “Model Comparison” section, there are circumstances where WIP and MC may coincide; if waste deposited in landfills has been relatively stable for recent years and similar parameters are used, the two methods will tend to agree with one another. If however, there has been a marked decline in waste being landfilled (examples being the introduction of an incinerator or new diversion programs to process organics), the WIP model will exceed MC. Conversely, if there is an increase in waste deposited in landfills (possible causes being the closure of an incinerator or reduced usage of aerobic waste treatment options), emissions from the MC method would exceed WIP. Greater complication in this relationship will be observed if projected values for parameters in the MC model such as oxidation, LFG capture efficiency and electricity generation (if considering offsets) differ from those employed in a WIP model.

USEPA WARM is unique in its consideration of both carbon emissions and sinks. This provides a simplified method for gaining insight into the carbon balance of waste operations. The developers of the model directly state that the tool should not be used in inventorying or accounting activities “since the life-cycle approach is not appropriate …(due) to the diffuse nature of emissions and emission reductions contained in (the) emissions factors applied” (USEPA, 2011). While providing interesting information, various constraints limit rigour, such as those on recyclable material inputs (% virgin: % recycled), efficiency of energy conversion to electricity, oxidation from landfill cover and grid emission factor. Examining the WARM method for composting emissions quantification, N₂O/CH₄ emissions are ignored, which is contrary to research presented in other literature (Amlinger et al., 2008; Hobson et al., 2006; Anderson, 2010; Brown et al., 2008). Considering these limitations and the credits provided for
undegraded carbon, it is unbalanced to compare absolute quantities obtained from WARM with other landfill MC and composting approaches. It may still be of interest to compare the variation in WARM with other models, keeping in mind that the results are relative to the limitations imposed by each model.

The other three MC methodologies (FCM-PCP, IPCC 1996 & 2006) vary in thoroughness. As stated earlier, due to the rigidity of the FCM-PCP model, it can only be considered a simplified first step to LFG emissions quantification. Additionally, the FCM-PCP tool calculates only emissions associated with landfill disposal and provides no allowance for including those from composting or incineration, which added over 100kt CO$_2$e of emissions to the IPCC 2006 total in the GTA example. Allowing for the input of other waste-related variables, such as those mentioned above, will improve this approach. The IPCC 1996 MC landfill calculation is simpler than what was performed for IPCC 2006, as the former aggregated various organic components of waste streams to a greater degree than latter. The difference in the results from the two methods was roughly 7%, which may be acceptable for purposes where such a disparity in approximations is sufficient.

The IPCC 2006 methodology can be improved through greater research on emissions factors and by the inclusion of guidelines on emissions from small-scale composting, however the pursuit of higher tier methods by cities would also address some of the uncertainty. Whether or not this endeavour is relevant to cities that may not have the means to pursue higher tiers is a matter for debate. An ideal approach for municipalities would include climate-specific emissions factors or methane generation reaction constants, site-specific recovery efficiency and oxidation data, and region-specific waste composition. The IPCC 2006 method could also be improved through further research on the fraction of carbon dissimilated in landfills and composting emissions.

Ultimately, the use of the MC methods for GHG inventory work must be avoided. It is suggested that 10-years of historical data with default IPCC 2006 coefficients be used to provide the most accurate picture of emissions in an inventory year, rather than quantifying future emissions which are far more uncertain. If 10-years of data are not available, landfilled waste can be extrapolated using an average waste per capita figure (or the oldest figure available) for city/region.
2.4 Conclusions

Empirical data are always ideal in quantifying GHG emissions from waste. However, if measured data are unavailable, modeling approaches can provide an estimate of emissions within the inventory year. In instances where data and parameters are more uncertain for a WIP approach, MC models can be used in GHG inventorying, though they are more appropriate when used for planning purposes. It is important to obtain the earliest possible annual landfill disposal data (composition and tonnage) to ensure greater accuracy of IPCC 2006 WIP calculations; however, this should not be a barrier to attempting WIP quantification.

As landfilled waste often represents the largest single urban emissions source managed by municipal governments, it is also an important opportunity for GHG reductions. In proper accounting of these emissions, the best approach would be to use the IPCC 2006 methodology for quantification and gauging the impacts of waste management decisions. This approach also provides the means to assess emissions from all waste management options examined here, unlike the other methodologies assessed.

Without standardizing the methodology selected for corporate waste GHG emissions inventorying, it is inappropriate to compare these emissions between cities. If it is assumed that IPCC 2006 WIP provides the most accurate estimate for LFG emissions inventorying, deviations by the other models for landfills would be 13%, 114% and 49% for IPCC 1996, USEPA WARM and FCM-PCP, respectively. When comparing waste emissions between cities, care must be taken to assess the methodology used and the selection of major parameters in each case. The same can be for decision-making related to treatment options.

In selecting a model for waste GHG measurement, five primary considerations affect the decision making process: 1) Assessment of disposal versus diversion practices (WIP vs. MC); 2) Motivation behind quantification (formal inventorying vs. planning); 3) Data quality / availability; 4) Acceptance and applicability of model assumptions / key inputs; and 5) Proportion of total (direct and indirect) emissions categories to be included.

Cities will likely continue to be leaders in efforts to address anthropogenic climate change, especially in the absence of binding international agreements or strong, unilateral action by national or state/provincial governments. Through diligent examination of the various
quantification methods for municipal emissions, the most appropriate tool may be selected for successfully targeting important emissions sources on the path to a low carbon future.
References


City of Toronto, 2008. Personal Communication [Irene Ford].


Region of Peel, 2009. Personal Communication, [Trevor Barton].


http://www.epa.gov/climatechange/wycd/waste/calculators/Warm_home.html, accessed Nov 2, 2009
3 Quantification of Direct & Embodied Carbon Sinks from Cities

As the acceptance and understanding of climate change science has spread globally, it has been recognized that the majority of current greenhouse gas (GHG) emissions can be attributed to urban dwellers. This is principally because more than 50% of the global population now lives in cities, with more than 75% of populations in more developed regions being urban by 2010 (UN, 2008). In their efforts to act on climate change, cities have taken up the task of emissions quantification to set emissions targets and identify reduction opportunities.

Quantifying emission sources allows a municipality to identify its major emitting sectors and set policies, fund projects and provide incentives for GHG reduction. However, the focus is placed on the sources of GHGs; potential carbon sinks are not typically assessed. Kennedy et al., (2009) examined GHG emissions from 40 global cities and found that agriculture, forestry and land use (or AFOLU) are generally neglected, with only a few exceptions such as the cities of Calgary, Sao Paulo and Rio de Janeiro. Nowak and Crane (2002), Kenney et al., (2001) and Pouyat (2002) have attempted to quantify carbon storage and/or carbon stocks in municipalities for forestry (former two) and soils (latter). As well, Pataki et al., (2006) have provided estimates on several carbon sinks within the urban setting, though through a broad review of literature and not an inventorying approach for a specific city.

The Intergovernmental Panel Climate Change (IPCC) suggests that the 3.1 GtC yr\(^{-1}\) captured by the biosphere through terrestrial and ocean sinks annually (averaged between 2000-2005) is far from being balanced with the 7.2 GtC yr\(^{-1}\) emitted from fossil sources and cement production and the 1-3 GtC yr\(^{-1}\) from land use change (IPCC, 2007). They suggest that sink management activities could increase global terrestrial sequestration by an additional 2.5 GtC yr\(^{-1}\) by 2040 using the range of options shown in Table 3.1 (IPCC, 2000). Even though many of these practices are rurally-based, they can be motivated through urban consumption.
Table 3-1: Carbon Sink Enhancement Activities Recognized under Article 3.4 of the Kyoto Protocol (from IPCC, 2000)

<table>
<thead>
<tr>
<th>Improved Management Practices</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Cropland</strong></td>
</tr>
<tr>
<td><strong>Rice Paddies</strong></td>
</tr>
<tr>
<td><strong>Agroforestry</strong></td>
</tr>
<tr>
<td><strong>Grazing Land</strong></td>
</tr>
<tr>
<td><strong>Forest Land</strong></td>
</tr>
<tr>
<td><strong>Urban Land</strong></td>
</tr>
<tr>
<td><strong>Land Use Change</strong></td>
</tr>
<tr>
<td><strong>Agroforestry</strong></td>
</tr>
<tr>
<td><strong>Restoration of Degraded Lands</strong></td>
</tr>
<tr>
<td><strong>Grassland</strong></td>
</tr>
<tr>
<td><strong>Wetland Restoration</strong></td>
</tr>
</tbody>
</table>

| Off-Site Carbon Storage                           | Harvest and usage in long-lived applications |

It is of interest to explore how the strategies described above relate to the urban environment. However, it is necessary to divide sinks into two broad groups; direct and embodied. A *direct* sink is one which ultimately results from carbon sequestered through biomass production within jurisdictional boundaries. An *embodied* sink is one that results from the consumption of a good or service where embodied carbon is involved (such as biogenic carbon stored in landfills).

This chapter has three related objectives. First, a description of the concepts of direct and embodied carbon sinks within the urban context is provided, and distinguishes between the two. Secondly, a methodology for quantifying components of these two categories of sinks for cities is described based on peer-reviewed literature and IPCC guidelines, with the Greater Toronto Area (GTA) in the year 2005 used as a case study. The methodology allows cities to quantify annual increases in carbon stocks, which is beneficial in that it allows the establishment of baselines for future reference and comparisons between cities on sink stewardship (especially relevant to direct sinks). This is followed by the third objective, which is to discuss the potential for carbon sink enhancement.
Methodologies used in the direct sinks quantified in this article are based on the IPCC 2006 guidelines to provide a means of quantifying both GHG sources and sinks. As with all IPCC (2006) methodologies for carbon inventory, carbon stock growth is quantified using one of three tier levels, with Tier 1 methodologies being the most basic and Tier 3 the most complex. IPCC Tiers 1 and 2 are generally used in this study.

One may argue that sinks within the urban boundary are insignificant compared to urban emissions (e.g. 15.8 MtC emitted from the GTA in 2005; Kennedy et al., 2010) and are relatively small when compared to national-scale carbon sinks. However, there are three reasons why quantification is of value: 1) If the objective is for cities to ultimately move towards a closed-loop carbon system, sinks accounting is inevitable; 2) Sinks often have many indirect benefits such as increased shading, evapotranspirative cooling (for urban trees; Akbari, 2002) and improved soil fertility and greater yields (for soil sequestration, Robert, 2001); 3) Urban-scale quantification has the potential to provide greater resolution for modelling efforts on the national scale if a sophisticated methodology is employed, reducing uncertainty.

3.1 Background – Direct and Embodied Sinks

It is important to define and distinguish between direct and embodied sinks prior to examining illustrations. Direct sinks are those that result from carbon storage through management activities for relevant biological resources within the inventory’s spatial boundary. Embodied sinks differ from these in that their existence within the spatial boundary is attributable to the consumption activities, independent of its origin; the embodied sink exists in the urban boundary due to consumer demand occurring within it. An example is the carbonation of concrete. In the methods presented below, direct sinks expand under natural sequestration, while embodied sinks expand as materials are consumed by urban residents.

The differences between direct and embodied sinks are illustrated by examples shown in Figure 3.1. Carbon uptake via photosynthesis results in direct biomass growth that yields an increasing carbon stock over time (Figure 3.1a). The quantity of the direct carbon sequestered is represented by the negative change in carbon emissions year over year (the bars in the figure, labeled “Annual Carbon Sequestered”). This would be similar to the carbon stored in soil, with biomass directly transferred to increase the carbon content of soil.
Direct carbon sinks, measured annually, can be compared to the embodied carbon sinks, which are estimated by their potential for long-term carbon stock creation. The embodied sinks considered here include harvested wood products (HWPs), carbonation of concrete and landfilled waste (containing biogenic carbon). HWPs represent an embodied sink as they contain the biogenic carbon from forestry products as long as they are prevented from decaying (Figure 3.1b); this is comparable to the cumulative carbon stored in the direct forestry sink. Concrete naturally undergoes long-term carbonation after setting, a degradative process which results in carbon storage (fulfilling its embodied potential to sequester carbon; Figure 3.1c). Lastly, due to prevention of complete decomposition of embodied biogenic carbon under anaerobic conditions, carbon stocks increase as long as organic wastes are deposited in sanitary landfills (Figure 3.1d).
The direct sink concept (shown in Figure 3.1a for forestry biomass) is relatively straightforward, as sequestration from biomass occurs in forests and soils within the urban boundary; as long as biomass stocks remain in the growth phase, the direct carbon sink (e.g. forest stands) expands (though the rate of sequestration changes as forests mature). The incremental annual growth would represent the inventoried stock gain. Similarly, if soils have not reached their capacity for carbon storage (related to soil type, climate, land-use, and management practices), growth in carbon stocks will occur.
Embodied sinks have more complexity associated with their spatial boundaries. Whereas direct sinks occur strictly within urban boundaries, the creation of the carbon stock (or the potential for a stock) for embodied sinks can occur on either side of the jurisdictional boundary. More precisely, industries that manufacture these means of carbon sequestration may or may not be within the urban boundary, but the consumer demand which drives their production, consumption and resultant disposal must be. Embodied sinks that are exported to other jurisdictions would not be included. Emissions associated with use, installation, processing, harvesting or disposal are not counted against the gross sinks total (refer to Figures 3.1 b, c and d). If a comprehensive consumer-side emissions inventory is conducted, emissions associated with the production and/or use of these products will be captured and can then be contrasted against carbon stock growth. However, the focus of this chapter is solely on gross and not net sinks; upstream quantification will only be briefly examined in supplementary material (see Section “3.5.2 Unquantified Sources and Sinks”). Efforts in taking a consumers approach in emissions inventorying show that greater accountability by the end-use can be achieved when a broader view of emissions associated with consumption is taken. As suggested in Figure 3.1, when relevant emissions are not considered, the perception of a larger carbon sink is created. Both direct and embodied sinks neglect emissions; however, we suggest that, using a lifecycle perspective, embodied sinks are likely to be a net emissions source.

Concrete and HWPs are the most consumed construction materials by weight according to Brunner and Rechberger (2002), leading them to be two of the most important material stocks. Additionally, landfills are a common disposal destination for biogenic carbon materials. This is not meant to be a comprehensive inventory of embodied sinks; books and products containing natural rubber are a few other examples of biogenic carbon stocks in the urban environment. Instead, the aim here is to illustrate the quantification of some key embodied sinks.

Embodied sinks described here are allocated to the city that consumes/handles the associated means of storage. For example, HWPs (Figure 1b) may or may not be extracted from within an urban boundary; however, our interest is in the principal motivation behind the creation of the sink (the consumer) and the sink is allocated accordingly. Pre-harvest, the biomass provides a direct carbon sink within the jurisdiction that it exists.
Miner and Perez-Garcia (2007) estimate that global CO$_2$ storage in HWPs for 2005 was 200 Mt CO$_2$e yr$^{-1}$, roughly 0.5% of global net primary production. As one would presume, carbon in the form of biomass harvested for use in forestry products does not immediately return to the atmosphere from which it was drawn (the assumption under initial IPCC inventory methodology in 1996). Hashimoto et al., (2002) show that significant carbon storage potential exists, amounting to a carbon sink that is currently 2% of the 1990 emissions baseline for a group of Annex I countries. It should be noted that this rate of carbon storage should not be assumed to be static; rather, it is associated with forest age and the disturbance cycles. Post-harvest, any carbon remaining in the HWP consumed by a city represent an increase in the embodied carbon stock within the urban boundary.

Cement represents a significant industrial sector emitter of GHGs. Through the calcination of limestone and combustion of fossil fuels, cement production contributes roughly 5% to global (and 4% of GTA) CO$_2$ emissions (Pade and Guimaraes, 2007; Kennedy et al., 2010). When combined with aggregates, additives and water in an urban environment, the ubiquitous urban material stock concrete is formed (predominantly calcium hydroxide, Ca(OH)$_2$; Haselbach and Ma 2008). Over time, carbon dioxide diffuses into concrete, creating the more stable compound calcium carbonate (CaCO$_3$), a process described by the reaction:

$$\text{Ca(OH)}_2 + \text{CO}_2 \rightarrow \text{CaCO}_3 + \text{H}_2\text{O}$$

(3.1)

The degree of carbonization is dependant primarily on concrete thickness (i.e. exposed surface area) and clinker concentration; the greater the clinker concentration and surface-to-volume ratio, the greater the degree of carbonization (Galan et al., 2010).

Pade and Guimaraes (2007) suggest that the average carbon uptake from concrete stocks may be between 14-25 kgC sequestered per cubic meter over a 100-year timeframe (Table 3.2). All carbon anticipated to be absorbed by concrete used in construction in a given year represents an increase in the urban carbon stock (though taking a lifecycle perspective would result in a net source; see Section 3.5.2.4).
Table 3-2: Summary of Concrete Production and Uptake for Concrete Poured in 2003 (Adapted from Pade and Guimaraes, 2007)

<table>
<thead>
<tr>
<th>Country</th>
<th>Norway</th>
<th>Denmark</th>
<th>Sweden</th>
<th>Iceland</th>
</tr>
</thead>
<tbody>
<tr>
<td>Concrete Produced, Mm³</td>
<td>3.3</td>
<td>3.9</td>
<td>4.0</td>
<td>0.4</td>
</tr>
<tr>
<td>C Uptake, tonnes C (100 year timeline)</td>
<td>60,000</td>
<td>93,000</td>
<td>65,000</td>
<td>8,000</td>
</tr>
<tr>
<td>C uptake, kgC m⁻³ (100 year timeline)</td>
<td>18</td>
<td>23</td>
<td>16</td>
<td>14</td>
</tr>
</tbody>
</table>

Finally, disposal of biogenic materials in landfill sites represents a waste management practice that results in long-term storage of biogenic carbon (Figure 1d). The US annual national inventory report includes a section detailing the contribution of food scraps and yard trimmings to this type of sink (USEPA, 2009). Though the landfill site may not be operated by the municipality or be located within the urban boundary, waste deposited is a result of a city’s consumptive activities. As well, selection of the waste treatment option lies within municipal jurisdiction and as a result of these considerations, the increased carbon stock is credited to the consumers who use the materials.

IPCC methodologies for waste management are primarily concerned with year-over-year emissions due to methane release. This approach is rational when addressing the immediate concern of GHG emission reduction, but work by Barlaz (1998) demonstrates that decomposition within landfills is generally incomplete under anaerobic conditions. Moreover, work by the USEPA (2006) applying Barlaz’s work suggests that organics deposited in landfills will result in a net carbon sink over the life-cycle, even after considering methane emissions (assuming 75% LFG collection and 10% oxidation). From a consumer standpoint, when neglecting all emissions associated with production, distribution and usage of materials deposited in landfills, municipal solid waste (MSW) represents a carbon sink.

It should be noted that temporal boundaries differ for embodied and direct sinks; embodied sinks assume an extended timescale starting from when materials are consumed by the inventoried jurisdiction, whereas direct sinks use a timescale of a single year of carbon removal. Considering lumber as an example, a tree during the course of growth is classified as a direct sink within the jurisdiction in which it is growing, with the annual increment of carbon stock change quantified for the direct sinks. However, once the tree is harvested, it ceases to
contribute to the direct sink for that jurisdiction and the direct sink would be zero, as it no longer is sequestering carbon. (It is worth noting that if the net carbon storage in the tree due to slower growth and losses in dead organic matter were zero, the annual change in carbon stock change, as well as the direct carbon sink, would also be zero). If the tree is processed into lumber, it then becomes a stock of carbon, providing an embodied sink within the jurisdiction in which it is consumed. The contribution of this sink is assumed to be over an extended period, assuming the application in which it is used is permanent (i.e. building construction). A stock-change method could be applied, though historical data would be required to determine stock magnitude and age, which would be difficult to obtain or estimate accurately. For example, if one were to inventory current carbon stocks in harvested wood products, an accounting of all buildings, their era / method of construction and any renovations occurring would need to be known. Additionally, direct sinks quantified here generally use the IPCC (2006) guidelines for quantification whereas embodied sink quantification methods are adapted from literature.

One final point on the bounding of the study is on the choice of using regional versus city bounds. When selecting boundaries of GHG sinks, it seems appropriate to use as broad a spatial urban boundary as possible. GHG emissions are ultimately a sustainability issue and their reduction will require the application of closed-loop analysis. As suggested by Rees (1997), the sustainable city will likely be a more self-reliant one; when attempting to classify a city as sustainable, the systems which support it must come into consideration. Using the broader urban geographical boundary (such as a regional one rather than municipal) facilitates this quest somewhat, providing a greater resource base to draw from in the transition to a balanced urban carbon budget.

### 3.2 Methodology

Using IPCC (2006) methodologies, approaches for quantifying direct carbon sinks are presented below. This is followed by the quantification of embodied carbon sinks, the calculation of which is based on various data and literature sources.

#### 3.2.1 Direct Sinks

Under national inventories, all lands designated as “managed lands” are applicable for GHG source/sink quantification. The IPCC (2006) defines these as “lands where human interventions
and practices have been applied to perform production, ecological or social functions”. Typically, lands within urban boundaries would hence be considered managed lands.

Under the IPCC (2006) inventory protocol, trees/forests contribute to five carbon pools:

1) Above Ground Biomass
2) Below Ground Biomass
3) Dead Wood – Non-living, standing woody biomass or fallen biomass that is generally greater than 10cm in diameter
4) Litter – Non-living biomass (2mm < diameter < 10cm)
5) Soil Carbon

Above and below ground biomass increase carbon storage through photosynthetic processes (Net Primary Production, NPP), whereas dead wood and litter, classified as dead organic matter (DOM), generally act as carbon sources. DOM can accumulate above ground (due to its slow degradation) and add to the soil carbon stock during decay.

As these are gross carbon storage inventories, DOM and litter contributions to emissions are not estimated (though in Tier 1 methodologies, litter fall is assumed to have no net impact on carbon fluxes; see discussion in Section 3.5.2.1). The first two biomass pools listed will be described below, detailing calculations of the annual soil carbon flux.

3.2.2 Forests

Forests provide a carbon storage opportunity by means of sequestration in live biomass, soil organic carbon, litter and dead organic matter. Even old growth forests have the capability to increase carbon stocks in the long term (though this does appear to diminish over time), with century old forests storing more carbon per annum than is released through respiration (Luyssaert et al., 2008). The means of quantification here are based on the IPCC Tier 1 and 2 methods; a Canadian Tier 3 approach is applied to the PURGE model (Chapter 4), which can be assumed to be more rigorous.

3.2.2.1 Urban Forests

Work on carbon sequestration estimation for urban areas has been previously undertaken, such as by McPherson (1998) and Novak and Crane (2002; using the Urban Forest Effects Model or
UFORE). The crown cover area-based inventoring method used in this study (and by the IPCC, 2006) is based on the work of Nowak and Crane (2002).

Wright (2000) provides land use data based on GIS information sourced from regional conservation authorities which allows for Tier 2 quantification of Urban Forests in the GTA. The total settlement area is given as 2361 km$^2$. Urban canopy data were obtained from the City of Toronto’s Urban Forestry Department (Pickett, personal communication, March 19, 2008), and the figure of 17.5% crown coverage is applied to all settlement areas across the GTA (giving a total canopy area of 401 km$^2$). Canopy data are not available for all municipalities in the GTA; however, an urban forestry professional at the Toronto Region Conservation Authority suggests that this is a reasonable assumption (Eastwood, personal communication, April 8, 2010). As stated earlier, the target year for carbon sink calculations was the year 2005. However, the most recent GIS data available are for 2000, and Toronto’s urban canopy data are from the year 2000. It is assumed that the change in urban canopy and settlement area between 2000 and 2005 is negligible.

For urban forests, carbon storage ($\Delta C_G$; tonnes C yr$^{-1}$) was calculated using the IPCC (2006) Tier 2a methodology. Removal of trees is not covered explicitly in this approach but is assumed to be captured in the data on crown cover area.

$$\Delta C_G = \Sigma A_{i,j} \cdot CRW_{i,j}$$  \hspace{1cm} (3.2)

where $A_{i,j}$ = Total Crown Cover Area of Class $i$ Woody Perennial Type $j$ (ha); $CRW_{i,j}$ = Crown Cover Area-based Growth Rate of Class $i$ in Woody Perennial Type $j$, tC (ha crown cover)$^{-1}$ yr$^{-1}$; a default value of 2.9 is applied in this study.

3.2.2.2 Regional Forest Biomass

The IPCC (2006) calculation of regional forests carbon sequestration is more complex than for urban forests, as its magnitude on the national level far exceeds that of forests within settlements. Data for IPCC (2006) Tier 1 (Gain-Loss method) quantification of regional forests is given through GIS data provided by Wright (2000), with default values for IPCC (2006) methodology used for emission factors (Table 3.3).
Table 3-3: Emission Factor and Parameters Used in Regional Forestry Calculations

<table>
<thead>
<tr>
<th>Average Annual Above Ground Biomass Growth (Gw) (t/ha-yr)</th>
<th>Above-Ground : Below-Ground Biomass (R) – Conifer / Broad Leaf</th>
<th>Carbon Fraction of Dry Matter (CF) – Conifer / Broad Leaf</th>
<th>Land Area (A) Conifer/Broad Leaf in GTA (km²)</th>
</tr>
</thead>
<tbody>
<tr>
<td>4 (0.5-7.5)(^1)</td>
<td>0.29 / 0.23(^1)</td>
<td>0.51 / 0.48(^1)</td>
<td>215 / 430(^2)</td>
</tr>
</tbody>
</table>

\(^1\) Default IPCC Value (Tables, 4.4.9, 4.4.4 and 4.4.3 - North American, Temperate); \(^2\) From Wright (2000)

An above-ground biomass density of 130 t ha\(^{-1}\) estimate is used based on IPCC (2006) estimates for North American temperate forests greater than 20 years of age. Calculation of annual regional forest carbon storage is made using Equations 3.3 and 3.4.

\[
\Delta C_G = \sum (A \cdot G_{\text{TOTAL}} \cdot CF)
\] (3.3)

where \(CF\) = Carbon Fraction of dry matter (d.m.); \(G_{\text{TOTAL}}\) = Mean annual above and below ground biomass growth (t d.m. ha\(^{-1}\) yr\(^{-1}\)); \(A\) = Land Area (ha)

\[
G_{\text{TOTAL}} = \sum \{G_w \cdot (1 + R)\}
\] (3.4)

where \(G_w\) = average annual above ground biomass growth (t d.m./ ha-yr) (\(G_w = 4.0\)); \(R\) = Ratio of below ground : above ground biomass for specific type

3.2.3 Perennial Crops

Another woody biomass carbon sink can be found in agricultural land use. With over 5000 ha of perennial crops in the GTA in 2005, it is reasonable to quantify the related carbon storage (modified from Statistics Canada, 2002; 2007).

A Tier 1 approach is demonstrated using default IPCC (2006) data for a temperate climate. It is assumed that perennial crops have not reached maturity (i.e. carbon is accumulating in biomass) and that tree removal is not taking place. In the case of Christmas trees, it is assumed that 10% are harvested annually, leaving only 90% to contribute to annual carbon stock (Christmas Tree Farmers of Ontario, 2010). From a stock perspective, the net flux would be zero, as the total carbon stock of these perennials would not change as long as the total land area sequestering carbon does not change; however, using a stock change approach, there is an annual increase.
When conducting an inventory, a careful quantification of the ultimate disposition of the perennial crops must be accounted for (i.e., if Christmas trees were disposed of in landfills, how much of the stored carbon was released back into the atmosphere from degradation and how much was retained within the landfill). Carbon sequestration from perennial crops ($C_G$; in tonnes C yr$^{-1}$) is determined by

$$\Delta C_G = \sum_i (A_i \cdot G)$$

(3.5)

where $A_i =$ Total area of cropland type $i$ (ha); $G =$ Biomass accumulation rate (tonnes C ha$^{-1}$ yr$^{-1}$); 2.1 given as temperate region default (IPCC 2006, Table 4.5.1)

As with other methodologies in this article, gross quantification of carbon storage is performed and emissions from agricultural operations are not calculated. Data from 2005 are interpolated linearly for GTA perennial crops (fruits, berries, nuts, nursery crops and Christmas trees) using 2001 and 2006 census data, and are provided in Table 3.4. This table also provides agricultural data that is applied in the soil carbon quantification methodology (seeding practices and manure application).

It is important to note that uncertainty of carbon storage in agricultural crops regarding variation in annual weather conditions, the maturity of crops and the storage capacity of biomass/soil needs to be addressed. Obtaining detailed information on crop type/age, region-specific relationships between weather and biomass growth, and harvesting rates would be necessary to reduce this uncertainty.
Table 3-4: Agricultural Data in GTA (Statistics Canada, 2002; 2007)

<table>
<thead>
<tr>
<th>Perennial Crops (ha)</th>
<th>Year</th>
<th>Halton</th>
<th>Peel</th>
<th>York</th>
<th>Durham</th>
<th>Toronto</th>
</tr>
</thead>
<tbody>
<tr>
<td>Christmas Trees</td>
<td>2006</td>
<td>82</td>
<td>55</td>
<td>291</td>
<td>303</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>2001</td>
<td>74</td>
<td>104</td>
<td>362</td>
<td>420</td>
<td>0</td>
</tr>
<tr>
<td>Nursery</td>
<td>2006</td>
<td>1,023</td>
<td>173</td>
<td>363</td>
<td>967</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>2001</td>
<td>1,034</td>
<td>277</td>
<td>279</td>
<td>927</td>
<td>0</td>
</tr>
<tr>
<td>Fruit Crops</td>
<td>2006</td>
<td>348</td>
<td>226</td>
<td>486</td>
<td>683</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>2001</td>
<td>323</td>
<td>211</td>
<td>264</td>
<td>851</td>
<td>0</td>
</tr>
</tbody>
</table>

Seeding Practices (ha)

<table>
<thead>
<tr>
<th></th>
<th>Year</th>
<th>Halton</th>
<th>Peel</th>
<th>York</th>
<th>Durham</th>
<th>Toronto</th>
</tr>
</thead>
<tbody>
<tr>
<td>Full Tillage (ha)</td>
<td>2001</td>
<td>7,845</td>
<td>10,222</td>
<td>21,463</td>
<td>30,892</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>2006</td>
<td>5,496</td>
<td>10,151</td>
<td>14,515</td>
<td>24,803</td>
<td>0</td>
</tr>
<tr>
<td>Low-Till (ha)</td>
<td>2001</td>
<td>6061</td>
<td>8,144</td>
<td>7,945</td>
<td>17,518</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>2006</td>
<td>5,925</td>
<td>6,349</td>
<td>10,677</td>
<td>21,227</td>
<td>0</td>
</tr>
<tr>
<td>No-Till (ha)</td>
<td>2001</td>
<td>8,997</td>
<td>4,067</td>
<td>8,961</td>
<td>13,087</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>2006</td>
<td>8,814</td>
<td>4,710</td>
<td>8,818</td>
<td>19,200</td>
<td>0</td>
</tr>
</tbody>
</table>

Farms Applying Manure (units)

<table>
<thead>
<tr>
<th></th>
<th>Year</th>
<th>Halton</th>
<th>Peel</th>
<th>York</th>
<th>Durham</th>
<th>Toronto</th>
</tr>
</thead>
<tbody>
<tr>
<td>Farms Applying</td>
<td>2006</td>
<td>233</td>
<td>242</td>
<td>360</td>
<td>943</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>2001</td>
<td>248</td>
<td>301</td>
<td>411</td>
<td>1262</td>
<td>0</td>
</tr>
<tr>
<td>Total Farms</td>
<td>2006</td>
<td>566</td>
<td>483</td>
<td>972</td>
<td>1686</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>2001</td>
<td>550</td>
<td>481</td>
<td>931</td>
<td>1540</td>
<td>0</td>
</tr>
</tbody>
</table>

3.3 Soils

Soils represent an enormous potential organic carbon sink. Post and Kwon (2000) suggest that in many terrestrial ecosystems, more carbon is stored in soils than the biomass it supports. Soil carbon fluxes are generally a function of climate, which affects net primary production and their resultant inputs through DOM (Kirschbaum, 2000). Current land use and management practices have lead to the degradation of soils and the release of carbon through decomposition and mineralization. However, improved management practices can lead to increased carbon stocks in soils through the assimilation of carbon from biomass sources such as tree litter, crop residues, and root systems.

Soils are classified into two broad categories under IPCC (2006) methodology: mineral and organic soils. Mineral soils are those that are moderately-to-well drained and represent the most significant soil type for the manipulation of carbon fluxes through land management practices. Organic soils are found in poorly drained locations (such as wetlands) and are able to store much greater quantities of carbon due to the anaerobic environment provided.

In 2000, the GTA land area was found to contain roughly 86 km² of wetlands (or roughly 1% of the total area; Wright, 2000). As this is relatively low and little sequestration is expected (IPCC...
Tier 1 methodology is primarily concerned with carbon releases due to drainage, organic soils will be neglected from this sink calculation.

The IPCC (2006) general equation for calculating the annual soil carbon flux is:

\[
\Delta C_{Soils} = \Delta C_{Mineral} - L_{Organic} + \Delta C_{Inorganic}
\]  

where \( \Delta C_{Soils} \) = annual change in soil carbon stock, tonnes C yr\(^{-1}\); \( \Delta C_{Mineral} \) = annual change in organic carbon in mineral soils, tonnes C yr\(^{-1}\); \( L_{Organic} \) = annual loss of carbon due to organic soil drainage, tonnes C yr\(^{-1}\); \( \Delta C_{Inorganic} \) = annual change in inorganic carbon from soils, tonnes C yr\(^{-1}\).

IPCC (2006) states that calculations for changes with inorganic carbon (such as chalk or limestone grasslands) changes are site dependent and require examination of soil mineralogy, but will be ignored here (which is assumed reasonable considering that Canadian national inventories exclude this category). A tier 1 approach for forest soils assumes no change in carbon stocks year over year (higher tiers would also require site-specific measurements; see Supplementary Material under “Soils” for estimates in the GTA).

Pouyat et al., (2002) have studied soil organic carbon pools in three urban areas (New York City, Chicago, and Moscow). Through direct soil measurements, a wide variety of soil carbon density (kg m\(^{-2}\)) was observed in urban ecosystems. Carbon densities in residential zones were on par with those observed in the forested zone in the same climatic region, though undeveloped areas within urban centres could also be lower than forested areas. In a more recent study, Pouyat et al., (2009) demonstrated that urban turfgrass and remnant forests in Denver, CO and Baltimore, MD, respectively, contain greater carbon stocks than natural shortgrass prairie and rural hardwood deciduous forests, respectively, which may be attributable to higher inputs in the former.

These studies provide evidence that a great deal of complexity and uncertainty in quantifying carbon fluxes for soils in urban area, due to variation in inputs, management practices, land cover and micro climate. Impervious cover (roads, buildings, parking lots, etc.) are not undergoing measurable changes in soil carbon as the inputs of litter and root systems are not available, which leaves turfgrass and urban forest (Pouyat et al., 2006). Additionally, they generally examine stocks and compare these with undeveloped areas; it would be valuable to examine stock
changes within a specific land use type. It is evident that further study is required before
generalized methodologies are available on quantification of urban soils. For similar reasons, the
IPCC 2006 methodology neglects carbon flux within settlements without direct measurement.

3.3.1 Agriculture

The IPCC 2006 guidelines use a stock change methodology to describe changes in mineral soil
carbon storage. For a given climate zone, soil type and management system, soil organic carbon
(SOC) stocks in mineral soils are calculated using Equations 3.7 and 3.8.

\[ \Delta C_{\text{Mineral}} = \frac{(SOC_0 - SOC_{(0-T)})}{D} \]  
(3.7)

\[ SOC = SOC_{\text{REF}} \cdot F_{LU} \cdot F_{MG} \cdot F_I \cdot A \]  
(3.8)

where \( C_{\text{Mineral}} \) = change in soil organic carbon (SOC) stocks in mineral soils, tonnes C yr\(^{-1}\); \( SOC_0 \)
= SOC in the last year of an inventory time period, tC; \( SOC_{(0-T)} \) = SOC at the beginning of an
inventory time period, tC; \( T \) = number of years over a single inventory time period, yr; \( D \) = time
dependence of stock change factors (default value of 20 is used); \( SOC_{\text{REF}} \) = Reference Carbon
Stock, tC; \( F_{LU} \) = Stock change factor for land-use systems (how long the crop system has been in
place); \( F_{MG} \) = Stock change factor for a management regime (tillage practices); \( F_I \) = Stock
change factor for input of organic matter (use of manure, irrigation, and other inputs affecting
biomass growth); \( A \) = Land Area, ha

The GTA lies in a cool, temperate, moist climate zone as categorized by the IPCC (2006).
Table 3.5 lists all stock change factors used, along with some assumptions (IPCC 2006, Tables
4.2.3, 4.5.5). Agricultural Census data are used to provide changes in farming practices for the
GTA and are presented in Table 3.4. IPCC defaults are used for \( F_{UG}, F_{MG} \), and \( F_I \).

Using the soil association map of Southern Ontario (Agriculture and Agri-Food Canada, 1960)
and land use maps from Wright (2000), it is observed that the region’s agricultural land is
dominated by clay (Peel, Halton) and loam (York, Durham). From VandenBygaart et al., (2004),
it is shown that luvisol comprises the greatest proportion of Ontario soils; hence, it is assumed
that all cropland soils are high activity clay.
Table 3-5: Stock Change Factors and Key Assumption used in Cropland Carbon Sink Calculations (IPCC, 2006)

<table>
<thead>
<tr>
<th>Stock Change Factor</th>
<th>Quantity Used</th>
<th>Assumptions</th>
</tr>
</thead>
<tbody>
<tr>
<td>$\Delta$SOC</td>
<td>95</td>
<td>Cropland is High Activity Clay</td>
</tr>
<tr>
<td>$F_LU$</td>
<td>0.69</td>
<td>Long-term Cultivated</td>
</tr>
<tr>
<td>$F_{LU}$</td>
<td>1.00</td>
<td>Perennial / Tree Crop</td>
</tr>
<tr>
<td>$F_{MG}$</td>
<td>1.00</td>
<td>Full Tillage</td>
</tr>
<tr>
<td>$F_{MG}$</td>
<td>1.08</td>
<td>Reduced Tillage</td>
</tr>
<tr>
<td>$F_{MG}$</td>
<td>1.15</td>
<td>No Tillage</td>
</tr>
<tr>
<td>$F_{I}$</td>
<td>1.11</td>
<td>High Input</td>
</tr>
<tr>
<td>$F_{I}$</td>
<td>1.44</td>
<td>High Input with Manure</td>
</tr>
</tbody>
</table>

Agricultural land uses are assumed to be long-term cultivated. $F_{MG}$ and $F_{I}$ are weighted in proportion to reported farm practices. The proportion of farms using manure is assumed to be equivalent to fraction of total farm area using manure and hence is weighted as stated in Table 3.4 (48, 63% in 2006 and 2001, respectively). High input agriculture (including irrigation, fertilization, etc) will generally lead to a faster rate of soil carbon accumulation, though certain practices (such as the application of lime and nitrogen fertilizer) can result in a net positive GHG flux (IPCC, 2006). As the number of farms applying manure decreased in the GTA between the two census years, a reduction in the annual carbon storage potential of agricultural soils occurred.

For the calculation of total carbon sequestered in the GTA, the total cropland area in 2006 was used. It should be noted that total cropland declined 3% from 2001 to 2006; however, the effect of this on the carbon sink calculation is negligible. A 2006 total agricultural census area of 140,700 ha is applied to Equation 3.7.

The implementation of conservation agriculture practices generally results in a relatively rapid short-term increase in carbon storage in cropland soils. However, it is important to note that conversion of agricultural land to perennial, unharvested vegetation will lead to greater carbon storage rates and total carbon stocks, as emphasized in Post and Kwon (2000).
3.4 Embodied Sinks

3.4.1 Landfill Waste

Only a fraction of biogenic carbon that is stored in landfills is degraded to carbon dioxide and methane (a default IPCC estimate suggests this is 50%; IPCC, 2006). Using USEPA (2006) estimates for the quantity of non-degraded carbon under anaerobic conditions (provided in Appendix A, Table A.6) and waste audits for the City of Toronto, a spreadsheet calculation was made for the long-term carbon sequestration for waste deposited in 2005. This waste includes both residential (divided into single family dwellings (SFD) and multi-unit residential (MUR) dwellings) and non-residential waste, with an assumption that composition did not vary across the GTA for the dwelling types.

Residential waste composition is tabulated using 2005-2006 waste audits for the City of Toronto for MUR and SFD (Stewardship Ontario, 2006). Non-residential waste composition is tabulated using data from TCSA (2008) and City of Toronto (1991). Total waste deposited in landfills is taken from regional data for residential waste and TCSA (2008) for commercial waste, with 796,000, 361,000 and 3,000,000 tonnes of MSW from single family units, multi-unit residential and non-residential, respectively. Woody biomass is composted in the GTA, and is assumed to provide no net carbon storage.

The waste carbon sink provided by landfills is calculated using Equation 3.9.

\[
\text{Carbon Storage (tonnes C yr}^{-1}\text{)} = \sum_{WF} CSF_{WF} \cdot T_{WF}
\]

where WF = waste fraction (by type, see Appendix A, Table A.5), CSF_{WF} = net carbon storage factor for a given waste fraction (tonnes C t\(^{-1}\)), and T_{WF} = tonnage of a given waste fraction deposited in landfills (tonnes yr\(^{-1}\)). The net carbon storage assumes 75% of LFG emissions are captured and flared, 10% oxidation of all methane generated, and excludes emissions related to transportation.

3.4.2 Cement / Concrete

After concrete is poured within the urban environment, the subsequent carbonation process represents an additional embodied carbon sink. Using the assumption of Sahely et al., (2003) of
180 tonnes of concrete consumed on the municipal scale per housing unit construction started, and Statistics Canada (2010) data giving 42,000 housing starts in 2005, concrete consumption in the GTA is estimated to be 7,500,000 t. Pade and Guimaraes (2007) present estimations of life-cycle carbonation of concrete from four Scandinavian countries (Table 1), based on utilization and recycling practices of concrete. These are applied to GTA data to determine the potential for carbonation of concrete consumed, using weighted averages from each country. Calculations of anticipated carbon sequestration over a 100 year timeline of concrete consumed in the inventory year are made assuming the following:

1) Similar distribution of cement uses in the GTA as the weighted Nordic average, giving a carbon uptake $1.93 \times 10^{-2}$ tonnes $m^{-3}$ (see Table 2)

2) Density of concrete of 24 kN / $m^3$ (2.45 tonnes / $m^3$)

3) 70 year useful life and 30 year post-demolition life (recycling)

$$\text{Carbon storage (tonnes yr}^{-1} = \frac{T}{\rho_c} \cdot CU$$  \hspace{1cm} (3.11)

where $T =$ concrete consumption (tonnes yr$^{-1}$), $\rho_c =$ concrete density (tonnes $m^{-3}$), $CU =$ carbon uptake (tonnes / $m^3$).

### 3.4.3 Harvested Wood Products

The useful life of HWPs varies depending on application and this must be considered. For example, fuelwood or consumer products (such as books, paper or packaging) can be assumed to have a negligible residence time in urban environments (Hashimoto et al., 2004). The latter set, however, have the potential to contribute to the landfill waste sink. In order to quantify annual sequestration of carbon in cities due to forestry products accurately, related material flows must be well understood.

For an urban area (with negligible wood product harvest within its boundaries), the most important consideration for its HWP carbon sink would be the usage phase. A full accounting of year-over-year carbon fluxes would take into consideration upstream emissions (e.g. harvesting, processing and wood wastes emissions) and downstream emissions (from past disposal of wood products). Both of these are neglected in embodied carbon sink inventories.
As sawn wood is the one of the most significant HWP stocks in the urban environment, it is assessed here as an example. Sahely et al., (2003) provide an estimate of the total residential lumber added to the building stock in the City of Toronto for the year 1999 (186,000 tonnes; 31 m$^3$/house, 450 kg/m$^3$, based on the construction of 12,855 houses). Using Statistics Canada data from 2006 Census, a similar calculation can be made for 2005, assuming a constant rate of construction, and including all construction of single detached, semi-detached, row houses, and duplexes. This provides a figure of 14,600 homes, or 204,000 tonnes of sawnwood. The sawnwood carbon fraction values (IPCC 2003 Good Practices Guideline Tier 2 methodology) presented by Cláudia Dias et al., (2009) are used to provide the figure for this embodied sink, simplified using Equation 3.12.

$$\text{Carbon Storage (tonnes yr}^{-1}) = T \cdot CF \cdot DW$$  \hspace{6pt} (3.12)

where $T =$ tonnage of wood consumed (tonnes yr$^{-1}$), $CF =$ Carbon Fraction per unit of dry weight (0.5 for sawnwood), $DW =$ Dry weight conversion factor (average of 0.435 for coniferous sawnwood).

### 3.5 Results

The current carbon sinks for the GTA are small relative to the carbon sources, but there is potential to increase these sinks. A summary of the GTA carbon sinks quantified for 2005 is found in Table 3.6, based on one year of direct sink sequestration and one year’s consumption for embodied sinks sequestration. Biomass in regional forests provides the greatest sequestration for direct sinks; meanwhile, landfills were the largest embodied carbon stock created over a single year. As well, maximum and minimum sequestration values are also provided (see “Uncertainty” for further explanation). Kennedy et al., (2010) estimate that the GTA’s 2005 scope 1 and 2 (direct emissions and upstream electricity) GHG emissions were approximately 15.8 Mt C (58 Mt CO$_2$e). The estimated sequestration in direct sinks in the same year is 0.32 Mt C (1.2 Mt CO$_2$e), or 2% of emissions.
### Table 3-6: Summary of 2005 Direct and Embodied Carbon Sinks in the GTA

<table>
<thead>
<tr>
<th>Carbon Sink</th>
<th>Estimation Method</th>
<th>Land Area (ha)</th>
<th>Default Annual Sequestration (tonnes C yr(^{-1}))</th>
<th>Minimum Annual Sequestration (tonnes C yr(^{-1}))</th>
<th>Maximum Annual Sequestration (tonnes C yr(^{-1}))</th>
</tr>
</thead>
<tbody>
<tr>
<td>Urban Forests</td>
<td>IPCC 2006 – Vol 4, Section 8.2.1, Tier 2</td>
<td>40,000</td>
<td>116,000</td>
<td>58,000</td>
<td>175,000</td>
</tr>
<tr>
<td>Regional Forests</td>
<td>IPCC 2006 – Vol 4, Section 4.2.1, Tier 1</td>
<td>65,000</td>
<td>158,000</td>
<td>17,000</td>
<td>257,000</td>
</tr>
<tr>
<td>Perennial Crops</td>
<td>IPCC 2006 – Vol 4, Section 5.2.1, Tier 1</td>
<td>5,025</td>
<td>11,000</td>
<td>3,000</td>
<td>18,000</td>
</tr>
<tr>
<td>Agricultural Soil</td>
<td>IPCC 2006 – Vol 4, Section 5.2.3, Tier 1</td>
<td>141,000</td>
<td>32,000</td>
<td>0</td>
<td>90,000</td>
</tr>
<tr>
<td>Landfills</td>
<td>USEPA, 2006</td>
<td>N / A</td>
<td>131,000</td>
<td>(859,000)(^1)</td>
<td>481,000(^2)</td>
</tr>
<tr>
<td>Concrete</td>
<td>Pade and Guimaraes, 2007</td>
<td>N / A</td>
<td>59,000</td>
<td>44,000(^3)</td>
<td>73,000(^3)</td>
</tr>
<tr>
<td>HWP</td>
<td>Modified from IPCC 2003 Good Practices Guideline, Tier 2</td>
<td>N / A</td>
<td>44,000</td>
<td>43,000</td>
<td>46,000</td>
</tr>
</tbody>
</table>

NC = not calculated; \(^1\) – This suggests an emission, assuming a scenario where no landfill gas is captured; \(^2\) - This neglects all landfill gas emissions, and just quantifies the carbon stored; \(^3\) – Maximum and Minimum Nordic Values

Assuming landfill waste is not disturbed, waste from the GTA deposited in landfills in 2005 will provide a carbon sink of 131,000 tC. The resultant anticipated C uptake for concrete laid in 2005 is 59,000 tC. HWP calculations examined sawnwood exclusively, being the largest fraction of global durable HWP consumption (FAO, 2005), giving a sink of roughly 44,000 tonnes C in 2005 for the GTA. This provides some insight as to the scale of the sink HWPs provide, though further consideration of book/paper and wood-based panel stock changes would provide a wider, more complete expansion of scope.
In quantifying embodied sinks from HWPs and waste, it is important to note that a danger of double-counting exists. HWPs often end up in the waste stream and if their disposal occurs within the same year as they are imported to an urban area, the sink can be counted twice. Hence, HWP sinks should be based on wood imported in a year minus the fraction that ends up as construction waste. This was not included in the calculation above as data on HWPs from demolition and other uses (commercial building construction, apartments, and industrial applications) are incomplete.

3.5.1 Uncertainty

There is uncertainty associated with all methodologies and data when attempting to quantify sources and sinks; those applied to embodied sinks that should be highlighted. Given that IPCC guidelines used here (Tier 1 and 2) applied default values from a region or climate zone, error values associated with these means can be employed to determine the range of possible carbon storage potential for the sinks identified. Table 3.6 provides these ranges based on those stated in the IPCC 2006 guidelines.

With regards to the urban forests, measured area data are available for the City of Toronto through a UFORE study, with limited data available for the other municipalities within the GTA (UFORE, 2007). The data from the UFORE study suggests the figure calculated based on the City of Toronto’s urban canopy from the IPCC 2006 guidelines is a good approximation of that obtained with detailed tree surveys and field measurements (28,000 tonnes C from UFORE vs. 31,000 tonnes C from the IPCC methodology).

Carbon sequestered in the embodied sinks assessed here could have greater uncertainty than direct sinks, as these are projections based on future sink management. The permanence of these sinks is, however, uncertain. Landfills, as an example, could be mined in the future, which would subject waste to aerobic conditions. Depending on the duration of exposure to these conditions or the end use of mined materials (i.e. combustion), significant emissions of carbon (presumed to be stored indefinitely) could occur. As well, one could argue that by incorporating LFG emissions in the calculation, the comparison with other sinks isn’t possible. Therefore, the total carbon stored, neglecting of LFG emissions, would be roughly 481,000 tonnes.
HWP storage in structures is expected to have long-term stability (maintained in the built environment for nearly a century). If, by century’s end, present emission reduction targets are realized, the release from HWP decay will be within the context of a low carbon urban system. However, disturbances (such as fire or decay), retrofits, and demolition could shorten the actual carbon release horizon. Additionally, uncertainty in the actual emissions and climate change scenarios at the time of release may result in HWP exacerbating future challenges.

Finally, concrete carbonization projections are based on Nordic averages of cement usage. The amount of carbonization in the 100-year is dependent on the end use of cement, which dictate key absorption factors such as thickness of structural elements and exposure conditions (Pade and Guimaraes, 2007). Actual carbonization rate may vary based on factors such as floor area density (i.e. proportion of concrete exposed to indoor conditions), concrete formulation (use of supplementary cementitious materials) or climate (relative humidity). Additionally, a large proportion of carbonization suggested by the Pade and Guimaraes (2007) comes from the creation of rubble of demolished concrete structures for use as an aggregate replacement. The proportion of concrete recycled will be dependent local practices.

### 3.5.2 Unquantified Sources and Sinks

For each type of sink identified, a number of emissions sources can be identified which have not been quantified yet have the potential to greatly diminish the net carbon sinks (Table 3.7). As the objective was purely gross sink quantification to identify the scale of sinks and their potential expansion, emissions were not quantified. It is important to note then that if sink expansion were to be pursued, consideration must be made in policy decisions to ensure that net emissions are not positive.

<table>
<thead>
<tr>
<th>Sink</th>
<th>Associated Emissions</th>
</tr>
</thead>
<tbody>
<tr>
<td>Forest</td>
<td>Decaying Dead Organic Matter (Litter, Infestations, Fire, Disease)</td>
</tr>
<tr>
<td>Agriculture</td>
<td>( \text{N}_2\text{O} ) from Application of Manure / Synthetic Nitrogen Fertilizer, Crop Residue Decomposition, Cultivation, Summerfallowing &amp; Irrigation</td>
</tr>
<tr>
<td>Harvested Wood Products</td>
<td>Harvesting, Processing, Storage, Transportation, Usage</td>
</tr>
<tr>
<td>Concrete</td>
<td>Cement Production, Transportation &amp; Usage</td>
</tr>
<tr>
<td>Landfills</td>
<td>Biogenic Carbon Harvesting, Processing, Storage, Usage &amp; Transportation, Landfill Operations</td>
</tr>
</tbody>
</table>
3.5.2.1 Forestry

Annual carbon losses can be assessed by tabulating all wood removals (for fuel wood and HWPs) and trees affected by disturbances (IPCC, 2006). Wood removals and fuelwood losses are most significant in regions with commercial logging operations. Assuming that no commercial removal operations (with the exception of Christmas tree plantations and nurseries) are present in the GTA, these losses are assumed to be zero. Discussion of tree removal is found in the perennial crop discussion in Section 3.1.1. It is assumed that other disturbances are negligible in the GTA for the year 2005. Tier 1 methodologies assume that carbon releases from litter are balanced by annual additions, creating a balanced carbon stock. The magnitude of carbon release from litter has been measured to be on the order of 1-2 tC / ha; however, inputs of new litter are generally in the same range and these were also not tabulated (Jonard et al., 2007; Ngao et al., 2005; Sulzman et al., 2005; Edwards & Harris, 1977).

Disturbances, such as fire, pests or disease, can result in further carbon emissions and are an ongoing concern in the GTA. The City of Toronto, as an example, is currently contending with an infestation of Emerald Ash Borer (EAB) and Asian Long-Horned Beetle (ALHB), to which at least 6% (up to 10% in surrounding areas; Toronto Star, 2009) and 50% of public street trees are susceptible to attack, respectively (City of Toronto, 2009). The ALHB infestation resulted in the removal of over 23,000 trees as of 2004. This is less than 1% of the 3 million trees on public property. Annual statistics on the extent of disturbances will be valuable in the quantification process, especially in relation to DOM. Given the scale of infestation in 2005, these are assumed to be negligible for the GTA.

3.5.2.2 Soils

While carbon stocks may be increasing in cropland soils due to conservation agriculture techniques, emissions from these soils are not insignificant. The 2008 Canadian National Inventory Report suggests that agricultural soils provided 18 Mt of CO\(_2\)e due to the sources listed in Table 3.7. Using the 2006 Census figure of roughly 36 million hectares, this represents an average of 0.5 t CO\(_2\)e per ha (Statistics Canada, 2008). Additionally, Rochette et al., (2008) suggest that in heavy clay soils, N\(_2\)O emissions from denitrification caused by organic matter decomposition may exceed the benefits of soil carbon storage.
Gains in carbon stocks in urban areas are difficult to quantify due to uncertainty in management and land cover. Since urban forest area correlates with canopy cover, overlap can occur with turfgrass and forestry area. If one were to make the assumption that all pervious land area in the GTA turfgrass, and one neglects aboveground and belowground inputs from trees, and applies gross rates of storage from literature (0.32-0.78 tC ha\(^{-1}\) yr\(^{-1}\); Qian et al., 2010), it is seen that urban soils have the potential to sequester between 38,000 and 85,000 tC annually (assuming that GTA urban areas are 50% pervious). However, it should be mentioned that turfgrass maintenance generally has numerous carbon inputs (mowing, fertilization & irrigation) that result in direct and indirect emissions, reducing the net carbon storage significantly.

Changes in SOC in urban forest soils were also not quantified. Berg et al., (2009) suggest that a mean sequestration in Swedish forest soils of 251 kg C ha\(^{-1}\) yr\(^{-1}\), with the potential to add nearly 20,000 tC yr\(^{-1}\) in the GTA. Using the CBM-CFS3 (Natural Resources Canada, 2009) forest carbon tool (and assuming the average age of forests in the GTA is 60 years), both DOM and soil carbon could potentially amount to 46,000 tC yr\(^{-1}\).

### 3.5.2.3 Harvested Wood Products (HWP)

Emissions from the harvest & processing of HWPs may be the closest in balancing with the resultant sink when compared with the other embodied sinks quantified; however, this is uncertain based on the forest stand management, harvesting, processing methods & end use. White et al. (2005) studied roundwood production in Wisconsin, and suggest that the net forest carbon budget, taking a full life cycle approach, ranges from -897 to 348 g C m\(^{-2}\) yr\(^{-1}\), i.e. it can be a net source or a net sink prior to use. Côté et al. (2002) found that by the time HWPs are ready for delivery to the consumer, 1.8 times the amount of carbon emissions are released as are stored in the end product. However, net carbon storage was positive when including carbon stocks retained in the harvest forest and in landfill sites for ultimate disposal. This suggests that the full HWP cycle from forest management to disposal can be a system for carbon storage.

### 3.5.2.4 Concrete

Absorption of carbon during concrete carbonation can indeed result in carbon storage. If one were only to consider the carbon balance associated with concrete stocks for a city which did not manufacture cement, the carbon sequestered may be surprisingly large. However, if upstream
carbon emissions from cement production (calcination and fossil fuel combustion, each contributing 50% of production-related emissions) are included, net emissions would be around 200 ktC using the estimation method of Pade & Guimaraes (2007). Even this provides a conservative figure, as it neglects emissions associated with transportation, pouring and demolition. One further point of interest is that two cement plant exists within the regional boundaries of the GTA (though the destination of this cement is uncertain), emitting roughly 380 ktC in 2005 (Environment Canada, 2008).

3.5.2.5 Landfills

Landfills have represented the largest MSW treatment option in the US and Canada historically and do provide storage capacity of carbon, but justifying this option as a sole alternative for waste disposal based on GHG-reduction benefits would be disingenuous. Significant emissions occur upstream from the manufacture of the products containing biogenic carbon that provide the waste sink. This emphasizes the need for considering additional factors other than carbon storage alone when deciding on waste treatment options (such as life cycle energy consumption or forestry resource preservation). For the sake of comparison, USEPA (2006) coefficients suggest that recycling of the same materials that provide this sink (biogenic carbon, not including food waste) would reduce upstream emissions by up to 2,000,000 tC (in place of using virgin materials).

3.6 Discussion – Potential for Carbon Sink Enhancement

Options exist to create or expand carbon sinks associated with urban regions. Many of these are commercially feasible at present, though others are speculative, with research ongoing. The sinks discussed below are described within the GTA context, yet their relevance is broader, as there is the potential for several of these to be applied elsewhere.

A variety of large point source industrial emitters exist within the GTA, even though the region is undergoing a transition from an industrial-based to a service-based economy. In 2005, over 14.5 Mt of CO$_2$e were released from facilities emitting 100 kt or more of CO$_2$e per annum in the GTA (Environment Canada, 2009). It is likely that cost-effectiveness (i.e. the return on capital invested, assuming a price on carbon is in place) of projects intended to utilize waste heat and CO$_2$ streams will increase for larger point-source emitters (rather than smaller emissions sources,
such as private residences); as a result, it may be possible to create further carbon sinks within the urban environment that utilize these flue gases.

With the potential for the installation of district energy systems within the GTA, it is possible to utilize low grade heat and CO$_2$ from flue gases. Some uses for CO$_2$ that have received attention recently include algae fertilization, which is then used as a feedstock for biofuels, with coproducts being used as a soil fertilizer (Packer, 2009). Additionally, CO$_2$ has the potential for use as an industrial feedstock, including polymers or synthetic fuels, though significant economic barriers exist to wider use of these applications (Aresta and Dibenedetto, 2007).

Another potential use for CO$_2$ is to increase its concentration in greenhouse environments. Studies have shown that crops grown at elevated temperatures, with greater nutrient availability and at higher CO$_2$ concentrations (between 600-900 ppm is generally the optimal concentration) have an increased yield and decreased time to flowering (Ontario Ministry of Agriculture and Rural Affairs, 2005; Mortensen, 1987). Huang and Bi (2006) propose that by integrating biogas production from agricultural wastes with greenhouse operations, the offset of natural gas combustion would amount to 3200 tonnes CO$_2$ e ha$^{-1}$ yr$^{-1}$. Depending on how wastes from yields are disposed of through the supply chain (i.e. landfill, composted/digested into soil conditioner), carbon stocks could potentially increase as a result.

Point-source emitters also present an opportunity for carbon separation and storage (or carbon capture and storage, CCS); however, this option is still in the development phase. CCS opportunities in southern Ontario are available in saline aquifers whose capacities are estimated at more than 700 Mt CO$_2$, though the injection point would be roughly 250 km from the GTA (Shafeen et al., 2004a). Shafeen et al., (2004b) estimate that the sequestration infrastructure costs for 5 Mt CO$_2$ yr$^{-1}$ would be between 27 and 50 USD/tC; however, the cost of separation is likely to cause this to increase to 275 USD/tC. Additionally, McKinsey & Company (2009) have suggested that CCS is on the high cost end of the spectrum of GHG mitigation solutions. Singh et al., (2011) also suggest that emissions (other than GHGs) from CCS are considerable. CCS is also being explored through capture of ambient CO$_2$ by means of chemical scrubbing technologies that could be applied at remote sites, eliminating costs for transportation (Lackner and Brennan, 2009; Keith, 2009). These ambient air CO$_2$ removal technologies would not likely
be applied within a dense urban environment, though may fall within a city’s domain of influence.

A number of additional carbon storage options which use biomass feedstocks, hence removing atmospheric carbon, have been described in literature. Some examples include biochar production (Dover, 2007) with soil application and biomass electricity generation with CCS (Möllerston et al., 2003). There is potential for these options to provide carbon sinks in instances where emissions from biogenic sources would have resulted (such as in regions where biomass is being affected by severe pest disturbances). However, these options are likely beyond the sphere of municipal governmental influence (agriculture and electricity). Additionally, increased use of HWPs could enhance this sink, with a provincial-level example in British Columbia, where the building code was recently altered to increase the maximum allowed height of wood-frame construction from four to six storeys (Province of British Columbia, 2009). This limit on wood-frame building height may be conservative; plans exist for the construction of a 30-storey wooden building in Austria (Inhabitat, 2010).

Green roofs represent another potential future carbon sink within the GTA, as they can be classified as grasslands (i.e. areas used for livestock grazing that would not regrow as forest under natural conditions). Indirect emission reduction benefits can be attributed to green roofs, such as reducing cooling energy needs through evaporative cooling (Saiz et al., 2006). No studies have been conducted on the potential area for extensive green roof installations for the wider GTA, though Banting et al., (2005) have conducted an analysis based on GIS data for the City of Toronto.

Banting et al., (2005) assessed potential greenroof capacity in the City of Toronto on all flat roofs greater than 350 m$^2$, with 75% roof coverage and estimate that 5,000 ha of compatible roof area exist. Using carbon storage values in above/below ground biomass and substrate carbon provided by Getter et al., (2009; total of 375 g C m$^{-2}$ yr$^{-1}$), approximate values for extensive greenroofs carbon sequestration potential for the City of Toronto are calculated to be 20,000 tonnes C yr$^{-1}$. It should be noted that annual storage potential has an underlying assumption of continued biomass growth and unsaturated substrate conditions, which will be dependant on greenroof and waste management activities.
3.6.1 Comparison with the Producer/Consumer Emissions Concept

The distinction between consumer and producer GHG emissions has received attention in recent years due to perceived inequity in allocation in current inventorying practices (Davis and Caldeira, 2010; Peters, 2008; Weber et al., 2007, 2008); essentially, when taking a producer approach to emissions inventorying, few upstream emissions are attributed to the consumer of goods and services. This leads to an “outsourcing” of GHG emissions as developed nations transition away from manufacturing economies and rely on developing countries for an increasing proportion of goods and services. The degree of exclusion of these emissions is exacerbated on the urban scale by narrowing the spatial boundaries further. Consumer-based methodologies allow for the accounting of upstream emissions occurring beyond the jurisdictional boundary of the final consumer, with many urban examples currently being employed (Kennedy et al., 2010; Hillman 2010; Ramaswami et al., 2008; Lenzen and Peters, 2010; Minx et al., 2009; Schulz, 2010).

It is of interest to compare the direct versus embodied carbon sink concept to producer/consumer emissions responsibility allocation literature, as allocation is relevant to both. In this article, the issue of allocation for sinks is simplified; both direct and embodied sinks are allocated within the spatial boundary of the inventory to the benefit of that jurisdiction.

Conventional emission quantification approaches have generally been producer-allocation, relieving consumers of their due share of responsibility. Consumer emissions allocation (perhaps equally unfairly) shifts all responsibility to the end-user. In a sense, embodied sinks do the opposite; embodied sinks shift all of the responsibility of emissions to the producer, whereas the consumer reaps the benefit of a negative figure on their carbon balance sheet from the creation of a carbon stock. There is the potential to use a more sophisticated allocation method for embodied sinks in theory, such as the value-added method suggested by Lenzen et al., (2007). However, the challenges in identifying components of the supply chain, especially when examining a sink as complex as landfilled waste, puts this approach beyond the scope of this article.
3.7 Conclusions

Embodied and direct sinks require differentiation as they tell different stories of carbon storage. Direct sinks store carbon due to land use management activities within an urban region, promoting the production of a biogenic carbon stock. An embodied sink may or may not occur within the inventoring boundary, but a carbon stock is created through consumption activities occurring inside the urban region’s jurisdictional limits.

Gross carbon sink quantification for the GTA shows that the assessed direct sinks (296 ktC) are greater than the assessed embodied sinks (234 ktC), though inclusion of upstream and downstream emissions will likely convert the latter to net GHG sources. The magnitude of direct sinks may also be reduced when considering a broader scope of activities associated with their management.

Pouyat et al., (2009) argue for the concept of “Urban Ecosystem Convergence”, where urban systems transition towards providing the same ecological services as would be provided under natural conditions. The different options presented above provide engineered solutions to attaining and potentially surpassing what would ordinarily be possible from a carbon sequestration standpoint. Some policy initiatives that would increase/maintain the capacity of existing urban carbon sinks are:

1) Conversion of marginal agricultural land to grassland, forestland or bio-energy crops
2) Halt development on agricultural, wetlands and forest land
3) Promote the planting of street trees and other urban greenery (including low maintenance urban forests, greenroofs and urban agriculture with waste heat and CO₂ utilization)
4) Encouragement of conservation agriculture, such as tillage reduction (where appropriate)
5) Promotion of wood-framed construction over concrete, where possible

It is important to note that the intention of this work was not to quantify sinks for the purpose of carbon credit allocation, rather to assess their magnitude, differentiate between direct and embodied sinks and gain greater insight into the discrepancy between GHG sources and sinks. A long-term target for sustainable cities should be to address this disparity and close the carbon loop.
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4 Greenhouse Gas Emission Scenario Modeling for Cities using the PURGE Model

Technological solutions are routinely suggested as a means for reducing the emissions of greenhouse gases (GHGs) contributing to climate change (Pacala and Socolow, 2004). These technological measures include fuel switching (i.e. coal to natural gas), adoption of renewable energy sources, the pursuit of carbon sequestration and the transition to higher efficiency conversion technologies. The gaps that these technologies must bridge are expansive; (Meinshausen et al., 2009) suggest that a peak CO$_2$e concentration of 450 ppm would likely limit warming to below 2°C, avoiding some of the most severe consequences of climate change. The IPCC suggests that to achieve this concentration, Annex I nations (including Canada) need to reduce their GHG emissions by 80-90% from 1990 levels by 2050 (Box 13.7; IPCC, 2007). In 2008, Canada’s GHG emissions were 24% above the 1990 baseline. Hence, there is value in modeling the impact of technological changes from now until 2050.

From a planning perspective, the potential exists to incorrectly perceive any action to mitigate climate change as a single, instantaneous, mitigating event with readily quantifiable reductions based on the change of emissions intensity. However, one must consider analysis through the temporal dimension to gain a better understanding of how emission reductions are realized and identify the pathways to a true low-carbon economy. A more complete understanding of how emission reductions are achieved is gained when considering the four important ways the temporal dimension influences GHG emissions and mitigation efforts: stock effects, weather effects, potency effects and process effects.

Stock effects are a function of how the technological stock can affect the rate of emission reductions. While the adoption of carbon reduction policies can result in a lower-carbon technological stock in the long term, the rate of replacement can limit short and medium term mitigation. As long as a technological service is being provided economically by an existing carbon-intensive technology (neglecting sunk costs), it will remain a component of the technology stock amongst lower carbon alternatives. This results in a lag in the reduction of emissions intensity of technology stock. For example, building codes and practices may improve over time; however, buildings are inherently long-lived. A significant lag will therefore be observed before the entire stock reflects changes to codes or practices.
Second, *weather effects* can have both positive and negative impacts on GHG emission reductions. Due to annual weather variability, building energy consumption and transport mode selection can differ (Saneinejad et al., 2010). As an example, warmer weather due to climatic change could result in lower heating energy demand but higher cooling energy demand. A change such as this could affect the proportion of fuel consumption to provide energy services year-over-year. When examining historical emissions, it is prudent to normalize energy consumption with respect to a base temperature (i.e. heating / cooling degree days). The relationship between weather and energy use (and, hence, emissions) has the potential to be broken as efficiency is improved (Mohareb et al., 2011).

Third, *potency effects* are associated with the absorbtivity and lifespan of GHGs in the atmosphere. Greenhouse gases vary with respect to the amount of infrared radiation they absorb and the period of time they persist in the atmosphere. Hence, certain processes and their resultant emissions may have shorter term impacts which are severe or perhaps persist for a greater length of time but have a lesser effect on radiative forcing. These effects are generally captured by normalizing with respect to CO$_2$ through global warming potentials over a fixed time frame (i.e. GWP$_{100}$ gives the relative contribution of a GHG to radiative forcing over a one hundred year timeframe). However, this ignores the actual immediate climatic impacts (Kendall et al., 2009).

Finally, *process effects* vary due to the processes/pathways through which GHGs are produced or sequestered, which are themselves time dependent. These can involve discrete actions (such as planting a tract of forest or depositing waste in a landfill) that have impacts that are continuous and non-linear (biomass growth or landfill gas production). One example is in the waste sector, where observed landfill gas emissions in a given year are dependent on actions taken in previous years (Mohareb et al., 2010). Another illustration is the removal and combustion of biomass from forests, which immediately results in a release (and potentially an offset) of carbon. However, this is followed by forest stock regeneration and can eventually provide a net carbon sink (Manomet Center, 2010). A final technological consideration of this nature is the degradation of performance in energy technologies (such as batteries; Ning and Popov, 2004; Dubarry et al., 2011) which have the potential to increase emissions intensity in providing their respective energy services.
This chapter describes an urban-scale model (developed here) where temporal considerations of GHG emission reductions associated with transitions to low-carbon technologies can be examined, specifically process and stock effects. This model, the *Pathways to Urban Reductions in Greenhouse gas Emissions* (or PURGE) model, focuses on the principal sources of GHG emissions attributable to cities: buildings, transportation, and waste.

The rates of building stock change, as well as the adoption of and decisions for retrofitting, are applied to the model for different eras of construction. The adoption of vehicles powered through alternate propulsion systems (i.e. those with electric motors) is also incorporated. Total annual electricity demand is likely to increase due to the anticipated reliance on battery-powered vehicles; however, the net impact on GHG emissions is quantified. The increase in diversion of organic waste and its treatment with incinerators or anaerobic digesters is also examined. The changing emissions intensity of the electrical grid is also factored into the emissions calculations for technologies that utilize this energy source (predominantly in buildings and transportation). Finally, the model captures the temporal effects on the principal source of carbon sequestration in urban areas (urban and regional forests). The PURGE model is applied to the current strategies suggested by government policy that are intended to impact GHG emissions in the Greater Toronto Area (GTA).

Table 4.1 provides an idea of the relative scale of these emissions for the City of Toronto. It should be noted that personal transportation contributes to 75% of transportation-related emissions and natural gas applications result in 60-80% of building energy consumption (with the rest predominantly attributable to electricity; City of Toronto, 2007a; OEE 2009).

**Table 4-1: Proportions of Toronto GHG associated with the Four Major Sectors to be Assessed**

<table>
<thead>
<tr>
<th>Sector</th>
<th>% of Total Emissions</th>
</tr>
</thead>
<tbody>
<tr>
<td>Natural Gas (predominantly heating)</td>
<td>37%</td>
</tr>
<tr>
<td>Transportation¹</td>
<td>27%</td>
</tr>
<tr>
<td>Electricity Generation</td>
<td>26%</td>
</tr>
<tr>
<td>Waste</td>
<td>4%</td>
</tr>
</tbody>
</table>

¹Only ground-based emissions are considered

Considering the contribution that the selected sectors make to urban GHG emissions, modelling of the transition of these to low carbon alternatives would provide a representative estimate of the timeline and scale of emission reductions. Using historical data and logistic curve modeling,
the methodologies for estimating these reductions are presented. Scenarios suggested by a broader study of literature are used to project the profile of emission reductions that can be expected from mitigation policies present will be further explored in Chapter 5.

4.1 Background on Technological Change

Technological change is generally the most economically and socially palatable option with regard to GHG emissions reduction. The IPAT equation (Holdren and Erlich, 1971) has been modified for greenhouse gas emissions using the Kaya identity (Kaya, 1990; from Raupach et al., 2007),

\[
C = P \times \frac{GDP}{P} \times \frac{E}{GDP} \times C_i \times \frac{E}{E}
\]

where \(C\) is carbon emissions in a given year, \(P\) is population, \(GDP\) is gross domestic product, \(C_i\) is the average carbon content of energy consumed, \(E\) is energy consumption. Looking at the terms, carbon emissions are a product of population, per capita income, energy intensity of the economy, and carbon intensity of energy conversion devices. Global population reduction is likely only achievable in the long term (UN, 2008) and socio-political resistance to policies aimed at reducing per capita GDP would present a barrier in a free market. This leaves technological solutions, i.e. energy conversion efficiency and demand reduction (represented by the 3\(^{rd}\) term), and alternative fuel sources or carbon sequestration technologies (characterized by the 4\(^{th}\) term) as the most saleable options in combating climate change. Given the likely focus placed on widespread technological change to reduce anthropogenic GHG emissions, and that this change is subject to a temporal influence, it is important that the expected shift to low-carbon alternative technologies be assessed for its expediency.

The transition to low-carbon technologies is a complex matter, with political, technological, economic and social factors influencing the rate and pathway of change. Political influence comes in the form of subsidies and financing schemes, which may favour certain technologies, in turn dictating the transitional pathway. Technological influences include technological lock-in of an energy-intense technology, preventing or restraining the adoption of one that is more efficient (Unruh, 2000). In the absence of carbon pricing, many transitional and disruptive low-carbon energy technologies will not be able to compete with entrenched technologies. Finally,
social influences affect adoption in that technological selection often does not solely favour low-carbon technologies; (Turrentine and Kurani, 2007) explain that fuel consumption is but one factor in the decision of which vehicle to purchase, and the impacts of fuel consumption are often not accurately estimated by the consumer. While considering these points, we set out to examine some commonly held notions of technological change.

Cumulative technological change has frequently followed a sigmoidal diffusion curve over time (Rogers, 2003; Banks, 1994; Grubler, 1997; see Figure 4.1). Initially, adoption is slow, with a period where niche applications dominate new usage (point #1). As barriers to adoption are overcome (such as economic or informational barriers), rate of adoption begins to increase, with a period of exponential growth (point #2; the “take-off” stage). Eventually the rate of adoption reaches a maximum (an inflection point, #3) and begins to decline to a point where adoption essentially plateaus (point #4).

Figure 4-1: Typical Technological Diffusion Curve
The first derivative of this function (signifying the rate of adoption) is examined by Rogers (2003; p. 281, Figure 7-3) to characterize different adoption groups using mean (\( \bar{x} \)) and standard deviation (\( sd \)) statistics. Assuming a symmetrical diffusion curve, the inflection point will occur at the mean. The innovators (those before \( \bar{x} - 2sd \)) would include the niche applications for the technology, giving way to the early adopters (those before \( \bar{x} - sd \)) along the exponential growth portion of the curve. The laggards (those after \( \bar{x} + sd \)) conclude the adoption phase up until the plateau.

In the transformation of the urban energy systems, Negro (2007) suggests using the term “transition” to describe the long-term infrastructural shift due to the gradual, continuous nature of the change, affected by a variety of factors and simultaneous developments. Rotmans et al., (2000) describe four distinct phases of transition:

1. Predevelopment/Exploration – A phase where a recognition and support of an alternative begins (i.e. innovators for a niche application)
2. Take-off – Substitution of the existing technology starts to occur (i.e. early adopters)
3. Breakthrough – Mass replacement to the point that large-scale infrastructure are irreversible altered to allow utilization of the new technology (early/late majority)
4. Stabilisation – The new technology is entrenched within a system, reaching a dynamic equilibrium (i.e. laggards)

One model for sigmoidal diffusion (described in Figure 4.1) is the logistic equation, where the “population” of a given technology (\( Y \)) at time \( t \) is given by

\[
Y(t) = \frac{K}{1 + \frac{K - Y_o}{Y_o} e^{-at}}
\]  

(4.2)

where \( K \) is the carrying capacity (or ultimate population) of the technology, \( Y_o \) is the initial population and \( a \) is the growth coefficient.

Grubler et al., (1999) examine technological change in the context of energy technologies. They note that in the predevelopment/exploration phase (using the terminology above), technologies are typically not economical, but once a niche market is discovered, a shift into the take-off stage begins. After a certain amount of technological learning, further progression into the take-off
phase occurs; this is due partly to production processes becoming more efficient and, hence, less costly, with greater experience in manufacturing the emerging technology. This has been observed with energy technologies as well, with solar photovoltaic (PV) and wind turbines experiencing exponential declines in production costs as cumulative production increases. A ~20% decline in cost for PV and wind has been observed after each doubling of production of these technologies (or ~80% “progress ratio”; IEA, 2000).

It is important to state that progress ratios are not uniform across all technologies, and have incorrectly been treated in this manner by many practical researches (Dutton et al., 1984). Progress ratios may display regional and perhaps temporal variability. Recent data U.S. data on PV and wind turbine production and costs suggest that progress ratios have declined relative to those observed by the IEA (2000); using capacity weighted costs and installed U.S. capacity, PV and wind have demonstrated progress ratios of 93% and 90%, respectively (Wiser & Bolinger, 2011; Barbose et al., 2011). In an effort to increase energy efficiency (in light of price volatility) and reduce carbon emission intensity (in anticipation of the internalization of the costs associated with carbon emissions), the transition to low-carbon emitting technologies has begun. Pacala and Socolow (2004) suggest that the technologies required to prevent doubling of pre-industrial emissions already exist, many at a commercial scale. It follows that a large number of the technologies that require mass deployment in the sectors of interest in this research (electricity generation, space / hot water heating & transportation) have already reached the “take-off” stage. Many alternatives to the current energy provision technologies are commercially available and many are economically competitive with the status-quo.

4.2 Components of the PURGE Model

Quantification of the release of GHG emissions from the four main sectors of the model (electricity generation, private transportation, buildings and waste) is described below. Generally speaking, the model requires knowledge of the current stock of technologies contributing to a given sector’s GHG emissions. The means of achieving the necessary emission reductions from technological change can follow any number of paths, requiring a model which utilizes broadly applicable parameters.

Past rates of stock change are explored in order to quantify how quickly new technologies can be adopted. These can be estimated using historical data, accompanied by contextual information
associated with the timeframes and geographic area analysed. As well, the energy requirements of the new technology and the expected carbon intensity of the fuel source utilised (not applicable for waste) are quantified based on analysis from early peer-reviewed sources. The sources and methodologies used for obtaining this information varies for each sector and are described in detail below.

It is important to clarify the boundaries associated with urban emissions quantified using the PURGE model. All direct emissions associated with personal transportation, electricity generation (used by buildings and vehicles), building thermal energy, and waste treatment (e.g. incineration of fossil carbon, methane emissions from landfills, methane/nitrous oxide from composting and anaerobic digesters). Embodied emissions in fuels (natural gas, coal and gasoline), material flows (e.g. food, consumer goods, construction materials, etc.), indirect emissions (such as from the transportation of waste) and CO$_2$ emissions from biogenic sources (incinerated wood/paper, landfill gas combustion) are not quantified. The only sources of biogenic carbon storage that are quantified are those associated for forests (biomass growth and dead organic matter) and urban street trees (biomass growth only).

### 4.2.1 Electricity Generation

When examining electricity generation over a period of time for a given set of generating stations and end users, emissions intensity ($E_{\text{EI}}$) can generally be calculated as

$$E_{\text{EI}}(t) = \frac{\sum_{FF_{\text{Type}}}(EG_{FF_{\text{Type}}}(t) \cdot EF_{FF_{\text{Type}}})}{EG_{\text{Total}}(t)}$$

(4.3)

where $EG_{FF_{\text{Type}}}$ is the total electricity generated using a given fossil fuel type (coal, natural gas) in MWh, $EF_{FF_{\text{Type}}}$ is the emissions factor of a fossil fuel generating station type (t CO$_2$e / MWh) and $EG_{\text{Total}}$ is the total electricity generated. Total GHG emissions from electricity generated are simply given as

$$GHG_{\text{Electricity}}(t) = \frac{E_{\text{EI}}(t)}{1 - LL(t)} \cdot \sum_{EndUse} EC_{\text{EndUse}}(t)$$

(4.4)
where $EC_{EndUse}$ is the total electrical energy usage (MWh) per end-use sector and $LL$ is the fraction of electricity lost in transmission and distribution.

### 4.2.2 Transportation

Urban transportation GHG emissions are a function of total fuel consumption and the emissions intensity of the required secondary energy source used. Direct annual aggregate private transportation GHG emissions is modeled as a product of the vehicle stock, average vehicle fuel consumption and vehicle kilometres traveled (VKT) for each vehicle type in operation.

\[
GHG_{Transport}(t) = \sum_{Model} \sum_{Year} \sum_{Vehicle\ Type} U(t) \cdot D(t) \cdot C(t) \cdot EI(t) \quad (4.5)
\]

where $U$ is the number of units of a given vehicle and technology type from a given model year that is currently in operation, $D$ is annual vehicle distance travelled (or Vehicle Kilometres Travelled, VKT) of the vehicle type, $C$ is the fuel consumption of vehicle type (GJ/km), and $EI$ is emissions intensity of the fuel (tCO$_2$/GJ). Emissions intensity of each fuel type can be assumed constant, given that there are no provincial/federal programs currently in place that would reduce the carbon intensity. The efficiency must be disaggregated for vehicle type in order to allow for sigmoidal diffusion equations of alternative technologies.

Greene (2006) presents a model for estimating vehicle stock change based on life-expectancy, GDP, vehicle price, fuel price and lagged vehicle sales. This model was produced for and applied to the NRC publication *Effectiveness and Impact of Corporate Average Fuel Economy (CAFÉ) standards* (NRC, 2002). It was then adapted for a Pollution Probe study on reducing GHG emissions from light-duty vehicles 25% by 2020 (Greene, 2006). The total number of vehicles purchased annually ($n$) is given as

\[
n_t = \left( \frac{GDP_t}{GDP_{t-1}} \right)^{\beta_1} \left( \frac{P_t}{P_{t-1}} \right)^{\beta_2} \left( \frac{p_t}{p_{t-1}} \right)^{\beta_3} \left( \frac{n_{t-1}}{n_{t-2}} \right)^{\lambda} \quad (4.6)
\]

where in year $t$, $GDP_t$ is the regional gross domestic product, $P_t$ is the average vehicle price, and $p_t$ is the average price of fuel. Elasticities of demand with respect to $GDP$, $P$ and $p$ are given by $\beta_1$, $\beta_2$ and $\beta_3$, respectively. Effect of lagged vehicles sales is represented by $\lambda$. 
The values of $\beta_1, \beta_2, \beta_3$ and $\lambda$ are determined using the Microsoft Excel® Solver tool\(^1\), using US data on average vehicle price, Toronto data on fuel price, Ontario GDP data and vehicle sales data (see Appendix A, Table A.7 for results); the coefficient of determination ($R^2$) between actual and modeled data is optimized (made as close to one as possible) by changing the values of the elasticities. The results suggest that for an increase in year over year for all parameters leads to an increase in vehicles sales, with the exception for light trucks. The positive elasticity from vehicle price increase was counterintuitive; this suggests that increases in vehicle price would increase sales. This counter-intuitive result could be attributed to region-specific correlations where finer scale data were not available (e.g. regional vs. provincial GDP) that were not captured when using national-scale cost data, or other factors that have overwhelmed the correlation of price and have pushed sales upwards. As a result, vehicle price is held constant for the application of this model.

Validation of this model (by comparing the modeled estimates to actual Ontario sales data) suggest that a reasonable agreement, with the model agreeing within an average error of 3%. The greatest outlier was in the year 2009, where error rose to 7 and 13% for cars and light trucks, respectively; modeled vehicle sales were lower than actual data. This could be attributed to incentives for vehicle sales that were implemented to mitigate declines due to the economic recession observed that year.

To determine the number of vehicles sold in a given year that rely solely on an internal combustion engine (ICE) powertrain, the following simplified approach is taken

$$f_{ICE}(t) = 1 - f_{ALT}(t) \quad (4.7)$$

where $f_{ICE}$ is the fraction of new vehicles sold using ICEs and $f_{ALT}$ is the fraction of vehicles sold employing alternative powertrains. The value of $n_t$ is then multiplied by either $f_{ICE}$ or $f_{ALT}$ to give the number of vehicles of each powertrain type sold in a given year.

\(^1\) MS Excel Solver tool allows the user to optimize a value or set of values to achieve a stated goal (maximize, minimize, equate to a specified target).
Greene (2006) also provides a three-parameter logistic model for predicting vehicle scrappage rate \( \sigma \) based on vehicle age \( a \).

\[
\sigma_a = 1 - \frac{1}{B_o + e^{B_1 + B_2 a}}
\]  
\((4.8)\)

The parameters \( B_0 \) and \( B_1 \) are given as 4 and 5.28, respectively, while \( B_2 \) is given as -0.33 for cars and -0.27 for light trucks for the USA (Greene, 2006) by fitting this model to historical US scrappage data from cars and light trucks from Davis and Diegel (2004). Scrappage data are unchanged in the most recent edition of this publication (Davis et al., 2010), and it is assumed that data for Canada would be similar.

Population of a given model year \( (P_i) \) for either cars or light trucks is given by

\[
P_i = P_{i-1} - \sigma_a
\]  
\((4.9)\)

Annual vehicle use is presumed by Greene to exponentially decay; as vehicles age, usage declines. Annual vehicle usage \( (U) \) is given as

\[
U_a = U_o e^{-\delta i}
\]  
\((4.10)\)

where \( \delta \) is the annual exponential rate of decline in VKT for a vehicle of age \( a \) and \( U_o \) is the VKT by a vehicle in its first year of operation. The PURGE model does not take the rebound effect into account (which Greene applies in his model), where energy savings are offset somewhat by increases in use due to cost savings. By excluding the rebound effect, a positive influence on GHG emissions is neglected; as absolute energy costs decrease, energy consumption has historically increased. In transportation, the magnitude of the rebound effect has generally been observed to lie between 10-30\% (i.e. for a reduction in fuel consumption through vehicle efficiency, 10-30\% of that reduction is lost due to increased vehicle usage). \( U_a \) is assumed to be constant across each class of car.

Sigmoidal diffusion in vehicle technologies has been observed previously (see Figure 4.3; USEPA, 2010); front wheel drive, port metering and variable valve timing all follow S-shaped paths in their adoption in new vehicles. The diffusion of alternative vehicles into the vehicle
stock is also assumed to follow sigmoidal diffusion, specifically a logistic diffusion curve is applied (i.e. a symmetrical sigmoidal curve).

![Figure 4-2: Sigmoidal Adoption of Various Technologies in Cars (Source: USEPA, 2010)](image)

In calculating the $f_{ALT}$ (see equation 4.11), the following is applied

$$f_{ALT}(t) = \frac{K}{K - \frac{f_o}{f_o}} e^{-at}$$  \hspace{1cm} (4.11)

where $K$ is the ultimate fraction of new vehicle sales for alternate powertrain vehicles, $f_o$ is the fraction of new vehicles sold that utilize an alternative powertrain at $t = 0$, and $a$ is rate of adoption.
4.2.3 Buildings

In many cities, buildings generally directly emit GHG emissions through the provision of heating services (i.e. water or space). However, upstream GHG emissions associated with building operations include electricity generation and material consumption / disposal. The focus of this module will be on GHG emissions associated with building heating and electrical energy demand, as operating energy is the largest component of building energy use (Sartori and Hestnes, 2007).

Based on the complexity related to the factors associated with energy demand, GHGs from buildings are modeled as the sum of emissions from each type of building (single family dwelling detached, multi-unit residential, retail trade, etc.) and both electrical and fossil fuel energy usage using

\[
\text{GHG}_{\text{Buildings}}(t) = \sum_{\text{Era}} U(t) \cdot \bar{A} \cdot \left[ \sum_{\text{Fuel}} C(t) \cdot EI(t) \right]
\]

(4.12)

where, for a given era, \( U \) is the total number of units of a given building type, \( \bar{A} \) is the average floor space of the building type (m\(^2\)), \( C \) is the energy consumption of the given building type (either electrical or fossil energy; GJ/m\(^2\)) and \( EI \) is emissions factor of the energy source used (CO\(_2\)/m\(^2\)).

Retrofits to existing buildings are assumed to follow a sigmoidal adoption curve (see explanation in Section 4.3.3). This applies equation 4.11 with some modifications to the definition of the variables; in the case of buildings, \( K \) is the ultimate fraction of buildings to adopt the retrofit, \( f_0 \) is the fraction of properties that are retrofitted in the year \( t = 0 \), and \( \alpha \) is the rate of adoption.

4.2.3.1 Residential Buildings

Given the total stock of buildings demanded (which can be forecasted based on population data and average inhabitants per unit type), estimations of building-type mix, and declines in housing stock, the building stock replacement rate and composition can be calculated. One can estimate the rate of energy efficiency improvement using this change in building stock composition. The cumulative number of new houses required from \( t = 0 \) can be modeled using
where for a given housing type (single family or apartment) constructed in a given year, $U_{Type, New}$ is the number of new units, $U_{Type, Era}$ is the number of old units from a specific era, $O$ is the number of occupants per unit of a particular dwelling type and $f_{Type}$ is the fraction of the urban population living in a dwelling type.

$$U_{Type, New}(t) = \frac{P(t)}{O_{Type}} \cdot f_{Type} - \sum_{Era} U_{Type, Era}(t) \tag{4.13}$$

where $D$ is the cumulative number of units demolished. Cumulative units demolished is a function of building age and is assumed to be linear in the PURGE model. This is given as

$$D_{type, era}(t) = A \cdot \phi \tag{4.15}$$

where $A$ is the building age in years and $\phi$ is the rate of demolition (units / yr).

### 4.2.3.2 Commercial and Institutional Buildings

While the residential building stock is suggested to be correlated with population growth, commercial and institutional buildings are assumed to be reliant on GDP growth. GHG emissions are calculated using Equation 4.12 and resultant emissions are calculated for each sector. Estimates in growth in GDP provide a means to calculate increases in floor area, assuming a fixed GDP/m$^2$ of floor area. Commercial floor area growth for an individual sector in a given year is calculated using

$$\Delta A_{Sector} = \Delta GDP \cdot \left( \frac{f}{(GDP/A)} \right)_{Sector} \tag{4.16}$$

where $f$ is the fraction of GDP growth attributed to a given sector.

### 4.2.4 Waste

Landfill disposal is currently the dominant source of GHG emissions from waste in the US and Canada, contributing 79% and 95% of all national waste sector emissions, respectively (USEPA, 2010b; Environment Canada, 2010). Even at the urban scale, landfill gas (LFG) is the most
prevalent source of direct emissions from waste treatment at 78% of all waste emissions in the GTA in 2005 (Mohareb et al., 2011). IPCC (2006) provides methodologies for quantifying waste from anaerobic digestion, landfill, incineration, and composting; a model for future emissions from waste treatment can be developed using their quantification methods. Projections can be made based on future estimates of diversion from landfills and diversion policies by applying models provided by the IPCC (2006) and other literature sources.

Residential waste production in the PURGE model takes into account population and residential building type, employing data used in the “Buildings” model module. The relationship used is described as

\[
\text{Annual Waste Generation} = \sum_i W_i(t) \cdot P_i(t)
\]  

where \( W_i \) is the waste generated per capita for occupants of a building type \( i \) (single family or apartment) and \( P_i \) is the total urban population dwelling in building type \( i \). Waste is then divided into distinct categories, consistent with those used in IPCC (2006) emissions quantification methodologies: metals, glass, food waste, garden waste, textiles, nappies, rubber/leather, plastics, sludge and other inert materials. This allows for the identification of diversion strategy for each waste material category, shaping future emissions.

Landfill gas production has been modeled using a first-order decay model with IPCC (2006) methodology. Focusing on methane (CH\(_4\)) production, landfill GHG emissions (t CO\(_2\)e) are calculated using

\[
\text{GHG}_{\text{Landfill}}(t) = ((\text{DDOC decompos} \,(t) \cdot 16/12 \cdot F) - R(t)) \cdot (1 - OX(t)) \cdot \text{GWP}_{100}
\]  

where, in the year \( t \), \( \text{DDOC decompos} \) is the mass (in tonnes) of decomposable degradable organic carbon decomposed, \( F \) is the fraction, by volume, of CH\(_4\) in LFG, \( R \) is the amount of LFG that is collected (t CH\(_4\)), OX is oxidation factor of the emitted LFG and \( \text{GWP}_{100} = \text{Global Warming Potential based on a 100-year timeframe (25 for CH}_4; \, 298 \text{ for N}_2\text{O)} \). Information on waste composition is required in order to determine \( \text{DDOC decompos} \) in a given year and detail on this calculation is found in IPCC (2006), with further illustration in Mohareb et al., (2011).
The IPCC guidelines provide quantification methodology for incineration, bioreactor anaerobic digestion (AD) and large-scale composting. Emissions from these sources (t CO$_2$e) are, respectively:

$$GHG_{\text{Incineration}} (t) = MSW (t) \cdot \sum_j (WF_j(t) \cdot dm_j \cdot CF_j \cdot FCF_j \cdot OF_j) \cdot \frac{44}{12}$$ \hspace{1cm} (4.19)

$$GHG_{\text{AD/Compost}} (t) = [(MSW (t) \cdot EF_{GHG}) \cdot 10^{-3} (1-R(t))] \cdot GWP_{100}$$ \hspace{1cm} (4.20)

where $MSW$ is the mass of wet waste treated for the given treatment type, Gg/yr; $WF_j$ is the fraction of component $j$ in the $MSW$, $dm_j$ is the fraction of dry matter in component $j$, $CF_j$ is the fraction of carbon in dry matter of component $j$, $FCF_j$ is the fossil carbon fraction in of component $j$, $OF_j$ is the oxidation factor, 44/12 is the conversion factor from C to CO$_2$, $EF$ is the emissions factor (kg / t waste treated; 4 for CH$_4$ compost, 0.3 for N$_2$O compost, and 1 for CH$_4$AD) and $R$ is the gas recovered (t; 0 for composting, 95% for AD).

Amlinger et al., (2008) provide empirical data on GHG emissions from the degradation of waste in residential-scale composting units. This is applied to the PURGE model using the equation

$$GHG_{\text{Backyard}} (t) = \sum_{GHG} MSW (t) \cdot EF_{GHG} \cdot GWP_{100}$$ \hspace{1cm} (4.21)

where the emissions factor for small scale composting is 2.2x$10^{-3}$ and 0.45x$10^{-3}$ (tonnes / tonne of wet waste composted) for CH$_4$ and N$_2$O, respectively.

Of the waste treatment options described above, incineration, AD and landfilling have the capability to generate electricity through CH$_4$ capture, when the necessary infrastructure is in place. For landfill gas and AD, electricity generation is calculated using

$$EG (t) = MR(t) \cdot LHV \cdot \eta \cdot \frac{1 \text{GWh}}{3.6 \text{TJ}}$$ \hspace{1cm} (4.22)

where $MR$ is the mass of methane recovered (in tonnes; can be calculated using equations 4.18 and 4.20), LHV is the lower heating value of methane (50 x $10^{-3}$ TJ / t) and $\eta$ is the efficiency of the conversion device. For incineration, a conversion factor per tonne of waste treated of 0.480 x $10^{-3}$ GWh / t ((using Denison, (1996))).
4.2.5 Forestry

Biomass, soil and dead organic matter (DOM) represent some of the principle carbon sinks for most nations. In the 2008, forestry provided 83 & 57% (or 704 & 18 Mt CO2e, respectively, excluding credits from harvested wood products) of net carbon sinks in the US and Canada, respectively (USEPA, 2010b; Environment Canada, 2010). Scaling these down to the urban level will provide some insight on the potential impact of municipal policy decisions in directly offsetting their carbon sources.

It is generally held that biomass growth on aggregate follows a sigmoidal curve with respect to its volume/mass, and hence, carbon sequestered (Botkin, 1993). Initially, growth follows an exponential curve, but slows as time progresses. The slowing of biomass growth in forests as they mature is due to competition with other species for limited access to solar radiation at and through the canopy and the annual loss of litter from limbs and branches. In essence, this implies that at its climax, solar radiation utilized by a tree will be used to maintain net biomass and any gains in carbon storage will come from soil and DOM.

The CBM-CFS3 forest carbon budget modelling tool, based on an IPCC Tier 3 approach to carbon dynamics in forests, allows the development of carbon storage curves based on yield data for given forest species. Biomass yield curves (carbon stock growth in even-aged forests) can generally be characterized as sigmoidal, while carbon stock changes in DOM follows an exponential decay (Payandeh, 1991, Kurz et al., 2009). Using Plonksi’s yield curve data, carbon stock change curves were developed for tolerant hardwoods and white pines (to represent softwoods) in the case study described below (Plonski, 1974). It should be noted that these curves are species and climate dependent and would require further development for application of this model elsewhere. These curves are applied to regional forest stands (i.e. forested areas outside of settlements), for both afforestation and existing forest tracts. The annual carbon sink provided by all regional forest stands in tCO2e is given as

\[ \text{GHG}_{RF}(t) = \sum_{\text{stands}} [C_{DOM}(t) + C_{Biomass}(t)] \cdot A_i \cdot \frac{44}{12} \]  

(4.23)

where \(RF\) is regional forest, \(C_{DOM}\) and \(C_{Biomass}\) are the carbon sequestered in dead organic matter and living biomass (t C), respectively, and \(A_i\) is the area of forest stand of age \(i\) (ha).
For urban carbon storage in forests within settlements, the IPCC (2006) approach is applied to quantify t CO$_2$e, using the equation

$$GHG_{UF}(t) = \sum A_{i,j} \cdot CRW_{i,j} \cdot 44/12$$  \hspace{1cm} (4.24)

where $UF$ is urban forest, $A_{i,j}$ is the total crown cover area of class $i$ woody perennial type $j$ (ha), $CRW_{i,j}$ is crown cover area-based growth rate of class $i$ in woody perennial type $j$, (t C / ha).

4.3 Projected Emissions from a Business-as-Usual Scenario

The GTA has a number of explicitly-stated GHG emission reductions strategies in place, allowing for the application of the PURGE model. These have been used to develop a business-as-usual (BAU) scenario for each sector and to quantify total emissions projected to 2050. Detailed information on assumptions made in the BAU scenario is provided in Table A.7 in Appendix A. Two key underlying assumptions are a 2.0% annual GDP growth (below the 1998-2008 average of 4.5%; Statistics Canada, 2011d) and a 1.4% annual increase in population.

4.3.1 Electricity Generation

The Ontario Government has set an aggressive strategy to reduce the emissions intensity of its power generation, with plans to close all coal-fired electricity generating stations by 2014 (Ministry of the Environment, 2009). Grid GHG emissions intensity (gCO$_2$e / MJ) up until the final decommissioning of these plants is provided by the OPA (2010, personal communication). It is assumed that after 2015, this GHG emissions intensity is held constant, as estimates on future changes to the electricity grid are not available. Given annual electricity generation of 146 TWh and 165 TWh by 2015 and 2030, respectively (Government of Ontario, 2010), this would cap natural gas electricity production at 3.3% of the total grid, with no other sources of direct GHG emissions (assuming an electricity generation efficiency of 35% and natural gas emissions factor of 64.2 t/TJ; Harvey, 2010; IPCC, 2006). In 2010, the fossil contribution to the electricity mix was 21.9% (IESO, 2011). The proposed decrease in emissions intensity can be attributed to the elimination of coal generation, increases in renewable, nuclear and natural gas electricity generation, and conservation programs.
4.3.2 Transportation

The province of Ontario plans for 1 in 20 vehicles to be electric vehicles (EVs) by 2020 (Government of Ontario, 2009a). Interpreting this, 5% of all vehicles on the road (as opposed to annual sales) will be electric-based (Battery Electric Vehicles (BEV) or Plug-in Hybrid Electric Vehicles (PHEV)). In order to accomplish this, the government currently provides rebates of up to $10,000 toward the purchase of an EV (depending on vehicle battery capacity). Applying the Greene model along with the parameters outlined in Table A.7, the entire vehicle stock in the GTA in 2020 will be 2.97 million; this would require 150,000 electric vehicles to be operating on GTA roads at that time if this target is to be met.

All alternate vehicles assessed here have an electric component (HEVs, PHEVs, & BEVs); hence, they are categorized into a single replacement technological category (i.e. all alternative replacing ICES). This assumes that gains in battery technology (storage capacity gains, economies of scale, progress ratios in general) benefit all electric vehicle types. It is assumed that HEVs are a transitional technology towards PHEVs and BEVs (Suppes, 2006).

The path of technology alternative vehicle adoption is examined within both the context of the diffusion trends observed in US sales data and from the goal stipulated by the Ontario
government. Historical US vehicle sales data (2000-2010) provide the first part of a substitution curve for alternative vehicle diffusion (HEVs; USEPA, 2010a). Applying this alternative vehicle substitution data to fit a curve and using the Excel Solver tool to provide a rate of diffusion for that results in the 150,000 vehicle figure (yielding values of $b$ of 0.4908 and 0.4926 for PHEVs and BEVs, respectively; see Equation 4.11 and Table A.7 for value of HEV), a technological replacement curve is developed. The percentage of vehicle sales that are alternate vehicles over time is seen in Figure 4.5, with both the current diffusion rate (based on regression of current sales) and the diffusion rate required to achieve the “5% by 2020” goal. In both cases, all new vehicle sales by 2030 will be battery-based vehicles. For more detail on shares of annual vehicle sales inherent in the “Current” scenario in Figure 4.5, see Figure 5.1a (scenario T1) in Chapter 5.

A metric used to determine the relative speed of diffusion of a given technology is the profile of diffusion rate (or $\Delta t$), the time elapsed between 10% and 90% market penetration (Grubler, 1997). The current rates of diffusion, using historical US sales data and Ontario government targets, demonstrate $\Delta t$’s of 11 and 8 years respectively.

Whether or not these seem to be realistic diffusion rates can be determined by examining the diffusion of previous automotive technologies. For passenger cars, port metering, variable valve timing and front-wheel drive, as examples, demonstrate $\Delta t$’s of 10, 15 and 19 years, respectively (USEPA, 2010a). Given market subsidies for their adoption, as well as the economic incentives from cost savings over the vehicle operation phase, it may be possible that diffusion rates for battery-powered vehicles fall at the lower end of this range.
### Buildings

With a fixed or growing population, building energy consumption reductions can only occur either through building retrofits / equipment upgrades or through demolition and rebuilding of existing structures. The PURGE model is able to model both of these approaches in the transition to a more efficient building stock.

A number of programs currently exist that would impact both of the approaches mentioned above. The first is the national Eco-Energy Retrofit program which provides financial incentives to residential and ICI sectors to retrofit existing structures to reduce energy use. The second is the 2012 update to the provincial building code, which will impact the energy efficiency of new construction in industrial, commercial and institutional (ICI) and residential sectors after 2011. The final program is the Tower Renewal Strategy, which aims to encourage the retrofitting of existing multi-unit residential (MUR) buildings (City of Toronto, 2010).

The Eco-Energy retrofit for homes program accepted applications in 2007-2009, and was resumed in 2011. The structure of this program required pre- and post-retrofit energy audits. Detailed data from these audits were collected by the Office of Energy Efficiency, allowing for assessment of energy savings and energy-use intensity (EUI) per unit of area by era of construction. Applying these data for the City of Toronto, and extrapolating for the GTA, an

**Figure 4-4**: Projections of Market Diffusion of Alternative Vehicles Using Current and Government Rates of Adoption
approximation has been made on energy savings by dwelling age, along with the rate of adoption. As the adoption of retrofits displays exponential growth (see Figure 4.6 for National Data), a sigmoidal diffusion curve is applied as described in Section 4.2.3, assuming complete adoption of retrofits ($K = 100\%$). While these data do seem to suggest the rate of retrofits is simply exponential during the subsidy program, the sharp increase in number of retrofits occurs during the final months of the program (during which deadlines required completion of retrofit projects or else funding would be lost). It is assumed that if the program was simply held in place indefinitely, an $S$-shaped adoption curve would result.

Details of parameters used in the sigmoidal adoption of retrofits are described in Appendix A, Table A.7. For residential buildings the initial adoption parameter is obtained from residential retrofit rates for the GTA during the ecoENERGY retrofit program and rate of adoption is selected so that it provides a complete retrofit by 2030.

![Figure 4-5: Number of Retrofit Exit Audits Registered Nationally Over Time (Source: OEE, 2011)](image)

The Tower Renewal strategy aims to retrofit 1000 multi-unit residential buildings by 2030. Tzekova et al., (2011) have found that pilot studies of building retrofits under this program have reduced electricity and natural gas consumption by 40 and 18 $\%$, respectively for gas heated buildings, which represent 72$\%$ of all apartments buildings in Ontario (OEE, 2009). Electrically
heated buildings (15% of apartments), realised reduction of 18% and 42% for natural gas and electricity, respectively. These energy savings are applied to the retrofit model, assuming they are representative of the energy savings possible.

The 2012 building code update states that the improvements will result in 35% energy savings relative to the existing code (Ministry of Municipal Affairs and Housing, 2010). For application to the PURGE model’s BAU scenario, the 25% savings is applied to thermal energy requirements for all new construction occurring in 2012 and beyond.

The contribution of commercial and institutional sectors to GDP in the GTA is taken from provincial and regional data (Statistics Canada, 2006; 2011c). GDP and energy intensity of industries (per m$^2$) is tabulated using the data from OEE (2009). Sectors examined are seen in Appendix A, as well as the contribution of each sector to GDP growth (assumed constant). Retrofits are assumed to follow a sigmoidal path, to the point where they achieve a 10% reduction in energy use by 2014 (Greening Greater Toronto, 2011). For the retrofit model for commercial and institutional buildings, the initial adoption parameter is 40% of that applied for residential buildings (since total floor area of is 40% that of residential buildings; OEE, 2009) and rate of adoption is selected so that it provides a complete retrofit by 2015. New commercial buildings are assumed to follow a sigmoidal path to reduction of energy use intensity as more efficient methods diffuse into new construction practices This is suggested by the exponential increase seen in LEED-registered non-residential developments in the GTA (shown in Figure 4.7) and nationally in the US since 2004 (Yudelson, 2010). High-performance building attributes from LEED certified buildings with similar end uses are used as the ultimate energy intensities of these buildings, using data from LEED certified buildings (USGBC, 2011). Fuel oil in new buildings is assumed to be negligible, as its usage has declined in recent years due to rising prices (OEE, 2009; Stat Can, 2011a).
4.3.4 Waste

All municipalities in the GTA have issued long-term waste management strategies for achieving greater diversion from landfills. Generally speaking, municipalities have targeted a 70% diversion rate by 2015.

The BAU scenario projects beyond the time horizon suggested by these strategies; hence, the diversion rate is presumed to have exceeded the 70% target by 2050 and reaching 88% (and 100% of organics). The diversion targets for each material are reached using a linear interpolation approach (see Appendix A, Table A.7 for details). Assuming that the ultimate goal is to minimize landfill as a treatment option while reducing GHG emissions and increasing electricity generation, the diversion strategy summarized is applied to individual materials of interest. It should be noted that these strategies may not in fact be optimal for achieving greenhouse gas emission reductions; they are an estimate based on current trends.
4.3.5 Forestry

The potential expansion of forestry for direct (and indirect) carbon benefits can be categorized into three main groups; urban forestry growth, regional forestry maintenance, and regional afforestation. Urban forestry targets have been identified for a number of cities, while regional forestry maintenance and afforestation assume no change in business as usual. That is to say, it is assumed that no afforestation will take place in the region and existing forests will not be cleared.

The City of Toronto has set a target to increase its urban canopy from its current 17% to 34% (City of Toronto, 2007b). Additionally, the Town of Oakville, with its canopy currently at ~29%, aims to increase its canopy to 40% by 2040 (Town of Oakville, 2006). Since most municipalities in the GTA have yet to complete a UFORE study, an assumption of 17% canopy coverage is applied. Changes in annual carbon sequestration figures are projected to 2050 using the 17% figure and a GTA target of 35% urban canopy. An IPCC (2006) default value of 2.9 t C/ha is applied in this study for CRW (see Equation 4.21).

4.4 Results

The City of Toronto states that its medium- and long-term climate goals are to reduce emissions by 30% and 80% by 2020 and 2050, respectively, applying a 1990 baseline (City of Toronto, 2007b). If one were to apply this same target to the GTA, scaling Toronto’s per capita emissions to the GTA (using a 1990 inventory completed by Harvey (1993) and 1990 GTA population data) would produce 2020 and 2050 targets of 24.5 and 7 Mt CO$_2$e, respectively. Applying the strategies described above using the PURGE model provides results that suggest that these targets are infeasible, primarily due to growth in GDP and population (Figure 4.8). Total emissions in 2050 are roughly 20 Mt CO$_2$e. It is important to note that this scenario does not include future policy that may result in further efficiency gains of the technology stock, but highlights the need for such policy. More aggressive technology stock improvements in are explored in Chapter 5.
The greatest source of emission reductions comes from transportation, primarily due to the assumed rapid transition to EVs and the assumption of low electricity sector emissions. Using the diffusion parameters that match government projections in the PURGE model, GHG emissions from GTA private vehicles will reach 3.8 Mt, a 74% reduction from a modeled 2005 baseline. In Figure 4.9, the change in vehicle stock, total annual VKT and average energy intensity of each vehicle type are shown to 2050.
Residential building sector emissions see a reduction in its total emissions relative to 2010 emissions, dropping 23%. This does not reach the depth of reductions, however, and continues to climb after initial declines from electric grid emissions carbon intensity and retrofits of existing buildings. Due to the improved efficiency of new construction, the pre-2010 building stock dominates emissions in 2050 (Figure 4.10).

**Figure 4-8:** Changes in Transportation Sector Over Time: a) Vehicle Stock Composition; b) VKT travelled annually; c) Fuel Efficiency of Vehicle Types
The C/I building stock follows a similar trend as the residential building stock, with emission reductions coming from a 10% reduction observed by 2014 (with no future reductions occurring in existing buildings) and improvements to new construction (completed by 2020). These improvements temporarily constrain the growth in emissions from new C/I buildings but are quickly overwhelmed by the increases in floor area due to economic growth (Figure 4.11).

**Figure 4-9:** Contributions of Residential Energy Consumption from Existing and Future Building Stock

**Figure 4-10:** Commercial Building Emissions by Fuel Type and Era of Construction
Emissions from waste in 2050 (78 kt CO\textsubscript{2}e) were reduced to a level that is roughly an 85% reduction from the 2008 estimate. Most of this is attributable to the declines in landfill gas emissions (Figure 4.12). The 2050 total emissions from waste is roughly one twentieth the amount of estimated CO\textsubscript{2}e stored in urban forests in that year.

![Graph showing GHG emissions and Electricity generation](image)

**Figure 4-11:** Emissions from the Management of Residential Solid Waste, plotted with Electricity Generation from Incineration, Landfill Gas Collection and Anaerobic Digesters (AD). MC=Municipal Composting and BC = Backyard Composting

### 4.4.1 Validation

It is worthwhile to provide a comparison of modeled GHG emissions from the PURGE model with inventories from other sources. The PURGE model is a bottom-up quantification tool, which examines energy end use data and scales up based on the size of the stock. Bottom-up quantification is generally not done for inventoring purposes, as these usually look at energy consumption within the urban boundary from utility data. This approach is taken by CivicAction (2011) in their quantification of GTA emissions in 2008 and 2009. These are compared to results obtained from the PURGE model (Table 4.2).
### Table 4-2: Comparison between Actual Inventory Data and PURGE Model Results

<table>
<thead>
<tr>
<th>Civic Action, 2011</th>
<th>PURGE Model</th>
</tr>
</thead>
<tbody>
<tr>
<td>Elect</td>
<td>NG</td>
</tr>
<tr>
<td>2008</td>
<td>7.27</td>
</tr>
<tr>
<td>2009</td>
<td>4.96</td>
</tr>
</tbody>
</table>

Natural gas and electricity emissions results from the PURGE model are scaled according to the proportion of residential & commercial/institutional energy demand in Ontario, in order to exclude emissions from the industrial sector. Given that the approaches to quantification are different, there is reasonable agreement between the results and measured data. Transportation emissions from the PURGE model in 2009 are lower than the Civic Action estimate; however, in this particular year, the PURGE model underestimates the number of vehicles purchased (see discussion in section 4.2.2 on transport model validation).

### 4.5 Discussion

Growth in demand for residential and commercial/institutional buildings will be difficult to overcome, in addition to the slow replacement of existing buildings. The modeled scenario above uses a relatively rapid rate of stock demolition and replacement, using data on stock change from given eras suggested by the OEE (2009). However, by examining building permit data, these estimates may be optimistic; within the City of Toronto, on average, 0.25% of its single family dwelling stock has been demolished per year between 2000-2010 (Personal Communication, City of Toronto, June 2011). This suggests a complete stock renewal in 400 years. In addition, average energy savings from subsidized home retrofits have achieved up to 24% and 5% savings for natural gas and electricity, respectively (dependent on the era of construction; personal communication, Natural Resources Canada, June 2011). For comparison, new housing will result in a 15% and 94% increase to the 2008 total of natural gas and electricity, respectively, for single family dwellings. To achieve the emissions reductions goals necessary, deeper savings in retrofits or a greater rate of replacement are required. As well, future reductions in the building energy code beyond the 35% target are necessary to prevent increases from this segment of the building stock.
This suggests that if the government is successful in reaching this goal through its subsidy programs, dramatic emission reductions can be achieved; however, it should also be noted that the PURGE model predicts that 20% of new vehicles sold in 2011 will be based on alternative propulsion systems. This is a very aggressive target, when considering that 2010 HEV sales in the US were roughly 6% of all vehicles (USEPA 2010a). Given this, the “1 in 20 by 2020” adoption target suggested by the Ontario government goal is likely unrealistic; it appears that even in transportation, the current strategies for emission reductions may not be sufficient to achieve an 80% reduction from the 1990 baseline (roughly 1.7 MtCO$_2$e). This is examined further through the transportation scenarios presented in Chapter 5. The scale of adoption of vehicles with an electric drivetrain has been constrained to 70%, with the remaining 30% HEV; if an alternative system that provides the energy storage capacity of fossil fuels becomes widely available (compressed hydrogen, second generation biofuels, ammonia, advanced battery technology, etc.), this could resolve the issue of vehicle range that prevents wider adoption.

4.5.1 Costs Associated with Current Strategies

As is suggested by the scale of the technological change required to meet the emissions targets, there are significant costs relative to the status quo in reducing GHG emissions. Over time, economies of scale and technological learning in the production of new technologies provide quick cost reductions; just as has been observed in PV and wind turbines, costs will decline as expertise and scale of production increase. Through the inclusion of externalized costs (e.g. a price on carbon), many of these technologies will become more competitive with existing technologies. It is still of interest to examine the expectations of the cost implications of current strategies.

The strategy for transition towards battery-based vehicles in the transportation vehicle stock has begun and a primary limitation for adoption is battery cost. Current battery technologies being employed (predominantly Li-Ion) are relatively expensive and it is likely that a cost premium relative to ICE vehicles will remain in the near future (Thiel et al., 2010, Offer et al., 2010, Van Mierlo et al., 2006). Offer et al., (2010) suggest that current premiums for BEV powertrains (25 kWh capacity) over ICEs approach $25,000 USD, while Thiel et al., (2010) suggests that the total purchase price premium between PHEVs/BEVs and 1.3 GDI turbo ICE is approximately €11,000 and €15,000, respectively. Both authors also suggest a reduction in this premium by
2030; Thiel et al., (2010) suggests a reduction of the PHEVs and BEVs premium to roughly €2,000, while Offer et al., (2010) suggests lifecycle costs for BEVs could be below ICEs (assuming, amongst other factors, that gasoline prices at or above $4.50/gallon). As well, the application of each of these types of cars (BEV, PHEV, HEV) could result in lower cost options being selected. (Pearre et al., 2011) found that 9% of drivers in their sample drove never exceeded 100 miles of travel in a given day. As well, Offer et al., (2010) suggest that BEVs with a range of 50 miles would have the lowest lifecycle cost of powertrain option under their projections. These suggest that changes in behaviour (greater use of car-sharing, rentals and/or public transportation for longer trips), could mitigate some of these costs if vehicles with lesser performance are widely adopted. Additionally, a future revenue stream for private vehicle owners exists if vehicle-to-grid energy becomes a reality for battery-based vehicles.

There are significant costs associated with retrofit subsidies in buildings. The EcoENERGY retrofit program in 2009 and 2010 provided federal grants totalling $585 million CAD (Government of Canada, 2011). The Ontario Government had budgeted over $250 million CAD since 2007 for a similar program (Government of Ontario, 2009b). The fraction of the Canadian single-family dwelling building stock retrofitted under the federal program is approximately 7% (Personal Communication, Natural Resources Canada, 2011; Statistics Canada, 2006). Additionally, costs associated with resultant savings from these programs will be higher than these figures might suggest, given that free-riders (those who would have adopted the energy saving technologies without incentives) and marketing likely contribute a potentially significant fraction of the expenditure (Joskow & Marron, 1991).

While costly, it is expected that most of these upgrades will result in a relatively short payback period. One Colorado study observed a payback period averaging 2.3 year for retrofits similar to those supported by the federal and provincial governments (Wierzba et al., 2011). However, retrofits on the scale that would be necessary to reduce emissions on the order of 80-90% would likely be much greater. The Now House project was able to reduce emissions associated with a post-second world war single family home by 55% at a cost of $85,000; however, using an electricity price of 0.08 CAD, the simple payback would be approximately 30 years (Now House, 2011; Ontario Hydro, 2011). Improving energy efficiency of new construction can provide a relatively short payback, with Gray et al. (2005) suggesting that an R2000 home had a simple payback period of 11 years. Finally, more efficient commercial buildings could result in
cost savings during construction, with Harvey (2009) suggesting that a high-performance commercial building in Vancouver, Canada was 9% less costly to build, while at the same time consuming roughly 50% less energy. It is unclear whether cost savings would be found in more extreme climates.

4.5.2 Co-Benefits of the Reduction in Greenhouse Gas Emissions

Transitioning away from fossil fuel consumption will have numerous other co-benefits, as suggested by (Chae, 2010) in an analysis of GHG mitigation options for the Seoul Metropolitan Area. One that was previously mentioned would be the decentralized storage capacity in the private vehicle stock; given the variability of energy generation from renewable technologies, this provides a means of storage and reduces the need for public sector investment in centralized technologies. As well, with the cleaner electricity generation, improved vehicle efficiency and the decline of the ICE, resultant criteria air contaminants (sulphur oxides, nitrogen oxides, particulate matter, volatile organic compounds, etc.) emissions will also decline. As a result, impacts from acid rain will be reduced and local air quality will improve, with fewer health effects from motive energy services. A study by the National Hydrogen Association (2009) suggests that in a scenario where all ICEs are replaced by BEVs alone, a 30% reduction in urban air pollution would result by 2100 relative to 2010. Finally, if distributed power generation is pursued with a variety of renewable technologies, coupled with diffuse storage media, the overall energy system resilience is improved; disturbances to system components will have smaller impacts on the entire system, when compared to the implications of the loss of a centralized generating station in the current electricity grid.

4.6 Conclusions

The PURGE model is a tool for emissions scenario development, and the results provided must be viewed within the context of the uncertainty of the projections presented here. Generally speaking, the assumption of continued population and economic growth greatly influences the upward GHG emissions trends in the absence of technological change. However, limits on the scale of energy reductions on retrofits that are currently undertaken in the buildings sector will prove to be a major obstacle towards the emissions levels prescribed by the IPCC. Deep retrofits and more stringent building energy code requirements for new construction are necessary, as
well as a renewables-based electricity grid to provide a low-carbon alternative to heating from fossil fuels.

In addition, greater reductions from private vehicle stock change will require infrastructural support (i.e. charging stations) and increased vehicle range. Manufacturing capacity of vehicles incorporating electric propulsion technologies must evolve rapidly to match current government projections, however the feasibility of industry to do so (as well as the required increase in market demand) is in question.

Waste emissions can achieve the reductions on the scale necessary but only if there is greater diversion of organics and harvested wood products, the benefits of which would provide indirect emissions reductions elsewhere. Forest biomass expansion could provide some sequestration to offset regional emissions and should be encouraged, especially in urban areas where other benefits (such as building shading and the reduction of urban heat island effect) can reduce energy demand. Current policy initiatives are a reasonable starting place, but more aggressive policy must be adopted to meet the targets required for global equity in efforts to reduce GHG emissions. It is clear that a significant challenge lies ahead to meet the 7 Mt emissions target; while attempts must be made to meet this challenge, it appears prudent to consider climate change adaptation measures within the urban context as well.
References


Saneinejad, S., Kennedy, C., and Roorda, M.J., (2010) “Analysis and modeling the impact of weather conditions on active transportation travel behaviour”, *CD Proceedings, 12th World Conference on Transportation Research, Lisbon*


Statistics Canada, 2011d. Table 384-0001 - Gross domestic product (GDP), income-based, provincial economic accounts, annually.


5 Scenarios for Technology Adoption Towards Low-Carbon Cities

The long-term reduction of greenhouse gas (GHG) emissions from the technology stock will require the adoption of a number of new technologies that are more efficient and less carbon intensive. The PURGE model (described in Chapter 4) is a tool to estimate diffusion of these new technologies and the impact on future GHG emissions. The examination of a number of different scenarios must be conducted to provide a clearer picture of the sensitivity of urban emissions to alternative technology futures to those suggested by current government policy.

This chapter aims to present scenarios for the transition of the three major sectors currently contributing to GHG emissions in the GTA; private transportation, residential buildings and commercial/institutional buildings. Further analysis of the impacts of specific technology transition pathways is provided by varying the underlying drivers of the dynamics of emissions of this sector (population and economic fluxes), as well as the electricity grid emissions that result in upstream emissions. Scenario analysis of the technological options will also provide insight into the scope of technological change that is required and the limitations of technological change in achieving deep reductions in emissions.

The City of Toronto has set a target of reducing GHG emissions 80% below a 1990 baseline by 2050. If one were to impose this target on the Greater Toronto Area (GTA), this would require an emissions reduction to 7 Mt – calculated using population data from Statistics Canada (2011) and per capita emissions from a GHG inventory from Harvey (1993). This is used as the benchmark to which the emissions reduction scenarios will be compared.

5.1 Adoption Scenarios

As seen in Chapter 4, future GHG emissions from the GTA will greatly depend on three primary variables (factors); population growth, economic growth and technology stock. Population growth stimulates demand for energy services provided within the three main sectors (transportation, residential buildings and commercial/institutional buildings) of urban GHG emissions; while current projections estimate 1.4 % growth per annum (Ontario Ministry of
Finance, 2009), it is worthwhile to examine slightly higher and lower growth estimates. For this purpose, 1% and 2% scenarios have been selected.

Economic growth is an input into the institutional/commercial buildings model, as well as the vehicle purchase model. Within the past decade, annual economic growth has averaged 1.7% (Statistics Canada, 2011). However, there is currently uncertainty in growth projections in mature economies; as a result, estimates of 1% and 3% growth are applied as well. Details on scenarios developed for the three major sectors are described below and summarized in Appendix A, Table A.8.

5.1.1 Transportation

Numerous technological options are currently being developed to replace internal combustion engines (ICEs), generally requiring electricity as a source of secondary energy. Delivery of tertiary energy (motive services) is commonly proposed to be through electric motors fuelled directly from battery storage from the grid, through biofuel combustion or through the conversion of hydrogen to electricity using fuel cell technology (Chan, 2002; Lave et al., 2003; Romm, 2006). A selection of studies currently exist that provide projections on the future of composition of the private vehicle stock, which allow for scenario development for the PURGE model. Summaries of the scenarios applied to the PURGE model are illustrated in Figure 5.1.

The baseline scenario (T1) uses the assumption that battery-based vehicles dominate the market and issues with hydrogen fuel cell vehicles (HFCVs; see Romm, 2004) prove cost prohibitive. Range limitations and cost for BEVs and PHEVs, respectively, prevent complete market domination, resulting in the persistence of HEVs for the long term (Pearre et al., 2011; LFEE, 2008). PHEVs are given a slight edge as they provide both electricity-based (presumably low carbon) and high efficiency fuel-based transportation (addressing range anxiety).

MIT’s Laboratory for Energy and Environment (LFEE) has produced a report that presents scenarios for vehicle technology stock to 2035 (LFEE, 2008). One such scenario includes a “no clear winner” future in which the market is comprised of a mixture of vehicles, including turbo-charged gasoline ICEs, hybrid electric (HEVs), diesel ICE, plug-in hybrid electric (PHEVs), with ICE-based vehicles still dominating annual vehicle sales by 2050 (extrapolating from the figures presented). This scenario (T2) is useful to estimate the impacts of maintaining ICEs in the
vehicle stock in the long term. This scenario would be realistic in the context of continued reluctance to pay for battery-based vehicles, or if pessimistic projected lags in the supply-side adoption hold true (15-20 years to achieve 30% of annual sales, estimated in the MIT study). This projection could hold true, given the suggestion by (Nakicenovic, 1986) that vehicle technologies generally have taken 10-30 before they are adopted in 50% of new additions to the vehicle stock; however, relative cost differences between ICE vehicles and alternatives could be reduced through increases in the price of oil, accelerating the adoption of non-ICE options. The MIT study assumes an oil price and availability that would not negatively impact demand.

A final scenario (T3) is applied to the GTA using the International Energy Association (IEA) and Organization for Economic Cooperation and Development (OECD) projection of global vehicle sales to 2050 under their BLUE Map scenario (IEA, 2009). This scenario suggests that, by 2050, vehicle sales of conventional gasoline vehicles will be negligible. While HEVs serve as a transitional technology, HFCV and BEVs would provide the majority of vehicle sales. This scenario assumes that costs of HFCV technology will decline dramatically in long term, and that challenges in providing the necessary infrastructure (such fuelling stations, hydrogen distribution networks, storage, and low-carbon hydrogen production) can be overcome.
Figure 5-1: a) Scenario T1, b) Scenario T2 and c) Scenario T3 for Vehicle Technology Adoption Applied to the PURGE Model; ICE - C = Internal Combustion Engine - Conventional; ICE – T = Internal Combustion Engine – Turbo; HEV = Hybrid Electric Vehicle; PHEV = Plug-in Hybrid Electric Vehicle; BEV = Battery Electric Vehicle

It should be noted that none of the scenarios directly follows the Corporate Average Fuel Economy (CAFE) standards to reduce the average fuel economy of new vehicles to 34.5 MPG by 2016 or 54.5 MPG by 2025. None of the scenarios reach the 34.5 target by 2016, however the T1 scenario does meet the 54.5 MPG target by 2023. The T2 scenario never meets the 2025 CAFE standard, and reaches the 2016 target in 2033. The T3 scenario attains the 34.5 MPG and 54.5 MPG targets by 2031 and 2038, respectively. It should be noted that the approach these standards take (focusing on MPG of fuel used) will like be altered once EVs and HFCV begin to play a more important role.
5.1.2 Buildings

Buildings, as seen in the previous chapter, present a challenge to reducing greenhouse gas emissions. Most newly constructed residential and commercial properties, once erected, are maintained for the long term (demolition permits in the City of Toronto amounted to roughly one quarter of one percent of the residential building stock between 2000-2010). Moreover, there are limitations to efficiency gains that can be made through conventional retrofits (Dong et al., 2005). As a result, building energy consumption can become “locked-in”, requiring changes in either behaviour or carbon intensity of the energy source in order to achieve deeper reductions in GHG emissions; this points to the need to reduce energy demand in new buildings in the short term. However, it is of interest to examine the effects of different technological scenarios to reduce GHG emissions.

5.1.2.1 Residential

Residential buildings (whose numbers are driven by population growth in the PURGE model as occupancy rates are held constant), are projected to increase in number in the coming years (Ministry of Finance, 2009). As a result, two principal strategies are available to directly reduce emissions; retrofits and improvements to the building energy code. Additionally, indirect reduction of emissions through lowering the carbon intensity of energy used is also an option, such as through the increased use of nuclear and renewables in electricity generation or the introduction of upgraded biogas to the natural gas distribution network. The focus here will be on the reduction of energy demand onsite through technological improvements.

Retrofit data is available for the GTA from Natural Resource Canada’s (NRCan) ecoENERGY Retrofit program (NRCan, Personal Communication, 2011). This data is provided from energy audits of residences in the GTA, both prior to and after retrofits. The energy reductions from retrofits are estimated using energy modeling software (i.e. HOT 2000), and available for both Single Family Units (SFU) and apartments.

Data on energy savings from retrofits are categorized by building type (apartment, SFU-attached, SFU-detached), vintage and energy source (natural gas or electricity). Three scenarios are developed using this data by vintage and fuel type; average savings (Scenario BR1), average of data points above the median (BR2) and average below the median (BR 3). These provide the
types of retrofits that homeowners have pursued given financial incentives and are assumed to be representative of what can be accomplished in the future. All of these scenarios are assumed to diffuse logistically, reaching 100% adoption by 2050 (Figure 5.2).

**Figure 5-2:** a) Diffusion of Home Retrofits; b) Building Code Changes into the GTA Housing Stock

Additionally, building code changes are assessed for their impact on GHG emissions. In a “current” approach, a 2011 35% reduction in building energy use is modeled. An aggressive scenario assessed will apply a 35% reduction followed by two successive 25% reductions in 2016 and 2021 (Figure 5.2).
5.1.2.2 Non-Residential

Commercial and institutional (C/I) sector emissions are assumed to be related to GDP growth, with further assumptions on the contribution and space intensity ($m^2$/GDP) of sub-sectors being held constant in the long term. As a result, rate of emission increases is proportional to GDP growth for each sector. This is moderated by improvements to the building practices (decreasing energy intensity) and retrofits to existing buildings.

The non-profit group CivicAction has suggested targeted a reduction in commercial sector energy use of 10% by 2014 (CivicAction, 2011). This is applied as a base scenario. A more aggressive scenario is developed using McKinsey & Co (2009); estimates are presented on energy reductions possible by applying strategies that have a positive net present value under their analysis. These energy use reductions are applied to the 10 sub-categories used in the PURGE model, with diffusion completed by 2025. An illustration of the reduction of energy intensity through retrofitting using office buildings as an example is seen in Figure 5.3.

A base case (CIB1) for new construction is assumed to apply the average LEED new construction energy use for relevant building types, taken from the USGBC database (2011). An aggressive new construction scenario is also applied (CIB2) using the best case from each sector from the USGBC database. These practices are assumed to diffuse logistically into building practices, though with a slower rate of adoption than is used in the base case above.
5.1.2.3 Electricity

The Ontario government is currently in the process of removing coal-fired generation from the provincial electricity grid. They have currently set a target for achieving this by 2014. The resultant grid intensity reductions represent the baseline scenario, where the 2014 grid intensity is maintained to 2050 (E1).

Uncertainty exists regarding the political will to maintain a course of the replacement of coal-fired electricity generation with less carbon-intensive options. An alternative scenario is presented where 2010 grid intensity is maintained in the long term (E2).

Finally, a scenario which leads to a zero-carbon electricity grid by 2050 is applied (E3). The means to achieving this reduction could follow a number of different approaches, including increasing the contribution of nuclear generation, coupling renewables with energy storage options and greater adoption of (net-zero) biomass electricity generation (Pacala and Socolow, 2004; Krajačić et al., 2011; Möllersten et al., 2003). The adoption of battery-based vehicles could facilitate the transition to low-carbon energy, with private vehicles providing storage capacity of intermittent renewable energy generation. While adoption of large, centralized
generating technologies is likely to follow a reduction pathway that resembles a step function (such as the replacement of coal-fired generation with natural gas), smaller-scale decentralized technologies (battery-based vehicles, renewables and fuel switching) may better be approximated by sigmoidal diffusion. The latter, more conservative adoption rate is applied.

**Table 5-1**: Parameters applied to the PURGE model under various scenarios

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Modeled Values</th>
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<td><strong>Universal Parameters</strong></td>
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<tr>
<td>2006 GTA Population</td>
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<td>Electrical Grid Line Loss Factor</td>
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<td>Annual Population Growth</td>
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<tr>
<td>Gross Domestic Product Growth (Ontario) – Applied for GTA</td>
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<td><strong>Transportation</strong></td>
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<td># of VKT in First Year of Operation</td>
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<td>Price of Gasoline, 2008</td>
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<td>Average Vehicle Purchase Price (assumed constant)</td>
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<td>Annual Fuel Efficiency Increase (all Technologies)</td>
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<td>Hydrogen Production Efficiency from Electrolysis (HHV)</td>
<td>0.85†</td>
</tr>
<tr>
<td>Hydrogen Compression Efficiency</td>
<td>0.798†</td>
</tr>
<tr>
<td>Hydrogen Distribution Efficiency</td>
<td>0.961†</td>
</tr>
<tr>
<td>Ratio of CS to CD mode for PHEV</td>
<td>1:3‡</td>
</tr>
<tr>
<td>Proportion of Ontario Sales Completed in the GTA</td>
<td>44%</td>
</tr>
<tr>
<td>$\beta_1, \beta_2, \beta_3, \lambda$ (Equation 4.6) – Cars – Obtained using historical data for each parameter and the Excel Solver tool</td>
<td>0.227, 3.606, 0.268, 0.679</td>
</tr>
<tr>
<td>$\beta_1, \beta_2, \beta_3, \lambda$ (Equation 4.6) – Trucks – Obtained using historical data for each parameter and the Excel Solver tool</td>
<td>0.316, 0.730, -0.228, 0.704</td>
</tr>
<tr>
<td>$B_0, B_1, B_2$ (Equation 4.8) (cars/light trucks)</td>
<td>4, 5.2781, -0.3306/-0.2682*</td>
</tr>
<tr>
<td>$K, a, b$ (Equation 4.11) – Obtained using HEV data and the Excel Solver tool</td>
<td>1, 0.008, 0.4128*</td>
</tr>
<tr>
<td><strong>Buildings</strong></td>
<td></td>
</tr>
<tr>
<td>Proportion of Single Family Dwelling– Attached (SFDA)</td>
<td>20.1%‡</td>
</tr>
<tr>
<td>Proportion of SFD – Detached (SFDD)</td>
<td>52.2%*</td>
</tr>
<tr>
<td>Proportion of Multi-Unit Residential (MUR)</td>
<td>27.6%‡</td>
</tr>
<tr>
<td>Occupants per SFD-Attached</td>
<td>2.94*</td>
</tr>
<tr>
<td>Occupants per SFD-Detached</td>
<td>3.27*</td>
</tr>
</tbody>
</table>
### Occupants per MUR

<table>
<thead>
<tr>
<th>Occupants per MUR</th>
<th>2.14^4</th>
</tr>
</thead>
<tbody>
<tr>
<td>$\varphi_{SFDD}$</td>
<td>-14.9</td>
</tr>
<tr>
<td>$\varphi_{SFDA}$</td>
<td>-3.6</td>
</tr>
<tr>
<td>$\varphi_{MUR}$</td>
<td>-12.6</td>
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</table>

### Building Energy Use Intensity Properties

<table>
<thead>
<tr>
<th>Single Family Detached Properties</th>
<th>Heat (GJ/m^2)</th>
<th>Electricity (kWh/m^2)</th>
<th>Avg Floor Area (m^2)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Before 1946</td>
<td>1.01</td>
<td>40.9</td>
<td>148.24</td>
</tr>
<tr>
<td>1946–1960</td>
<td>0.74</td>
<td>40.9</td>
<td>126.69</td>
</tr>
<tr>
<td>1961–1977</td>
<td>0.60</td>
<td>39.5</td>
<td>143.24</td>
</tr>
<tr>
<td>1978–1983</td>
<td>0.51</td>
<td>39.5</td>
<td>169.53</td>
</tr>
<tr>
<td>1984–1995</td>
<td>0.44</td>
<td>37.7</td>
<td>189.56</td>
</tr>
<tr>
<td>1996–2000</td>
<td>0.37</td>
<td>37.7</td>
<td>187.77</td>
</tr>
<tr>
<td>2001–2005</td>
<td>0.36</td>
<td>37.7</td>
<td>195.66</td>
</tr>
<tr>
<td>2006–2012</td>
<td>0.29</td>
<td>37.7</td>
<td>195.66</td>
</tr>
<tr>
<td>2012–2050 (w/o building code improvements)</td>
<td>0.24</td>
<td>37.7</td>
<td>195.66</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Single Family Attached Properties</th>
<th>Heat (GJ/m^2)</th>
<th>Electricity (kWh/m^2)</th>
<th>Floor Area (m^2)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Before 1946</td>
<td>0.91</td>
<td>40.9</td>
<td>125.05</td>
</tr>
<tr>
<td>1946–1960</td>
<td>0.67</td>
<td>40.9</td>
<td>126.68</td>
</tr>
<tr>
<td>1961–1977</td>
<td>0.54</td>
<td>39.5</td>
<td>114.36</td>
</tr>
<tr>
<td>1978–1983</td>
<td>0.46</td>
<td>39.5</td>
<td>122.31</td>
</tr>
<tr>
<td>1984–1995</td>
<td>0.39</td>
<td>37.7</td>
<td>119.55</td>
</tr>
<tr>
<td>1996–2000</td>
<td>0.33</td>
<td>37.7</td>
<td>133.39</td>
</tr>
<tr>
<td>2001–2005</td>
<td>0.32</td>
<td>37.7</td>
<td>147.29</td>
</tr>
<tr>
<td>2006–2012</td>
<td>0.29</td>
<td>37.7</td>
<td>147.29</td>
</tr>
<tr>
<td>2012–2050 (w/o building code improvements)</td>
<td>0.24</td>
<td>37.7</td>
<td>147.29</td>
</tr>
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</table>

<table>
<thead>
<tr>
<th>Multi-Unit Residential Properties</th>
<th>Heat (GJ/m^2)</th>
<th>Electricity (kWh/m^2)</th>
<th>Floor Area (m^2)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Before 1946</td>
<td>0.7</td>
<td>40.91</td>
<td>84.1</td>
</tr>
<tr>
<td>1946–1960</td>
<td>0.5</td>
<td>40.91</td>
<td>83.7</td>
</tr>
<tr>
<td>1961–1977</td>
<td>0.4</td>
<td>40.91</td>
<td>85.2</td>
</tr>
<tr>
<td>1978–1983</td>
<td>0.4</td>
<td>39.46</td>
<td>96.3</td>
</tr>
<tr>
<td>1984–1995</td>
<td>0.3</td>
<td>39.46</td>
<td>90.2</td>
</tr>
<tr>
<td>1996–2000</td>
<td>0.3</td>
<td>37.70</td>
<td>86.5</td>
</tr>
<tr>
<td>2001–2005</td>
<td>0.3</td>
<td>37.70</td>
<td>96.5</td>
</tr>
<tr>
<td>2006–2012</td>
<td>0.2</td>
<td>37.70</td>
<td>97.1</td>
</tr>
<tr>
<td>2012–2050 (w/o building code improvements)</td>
<td>0.2</td>
<td>37.70</td>
<td>97.1</td>
</tr>
</tbody>
</table>

### Single Family Unit – Detached Retrofit Improvements (OEE, 2011)

<table>
<thead>
<tr>
<th>Natural Gas</th>
<th>Electricity</th>
</tr>
</thead>
<tbody>
<tr>
<td>BR1</td>
<td>BR2</td>
</tr>
<tr>
<td>Before 1946</td>
<td>-26.6%</td>
</tr>
<tr>
<td>1946–1960</td>
<td>-25.3%</td>
</tr>
<tr>
<td>1961–1970</td>
<td>-22.4%</td>
</tr>
<tr>
<td>1971–1980</td>
<td>-18.8%</td>
</tr>
<tr>
<td>1981–1990</td>
<td>-16.4%</td>
</tr>
<tr>
<td>1991–2000</td>
<td>-17.0%</td>
</tr>
<tr>
<td>2001–2010</td>
<td>-15.9%</td>
</tr>
</tbody>
</table>

### Single Family Unit – Detached Retrofit Improvements (OEE, 2011)

<table>
<thead>
<tr>
<th>Natural Gas</th>
<th>Electricity</th>
</tr>
</thead>
<tbody>
<tr>
<td>BR1</td>
<td>BR2</td>
</tr>
<tr>
<td>Before 1946</td>
<td>-26.6%</td>
</tr>
<tr>
<td>1946–1960</td>
<td>-25.5%</td>
</tr>
<tr>
<td>1961–1970</td>
<td>-22.8%</td>
</tr>
<tr>
<td>Year</td>
<td>Natural Gas</td>
</tr>
<tr>
<td>--------------</td>
<td>-------------</td>
</tr>
<tr>
<td>Before 1946</td>
<td>-24.3%</td>
</tr>
<tr>
<td>1946–1960</td>
<td>-26.5%</td>
</tr>
<tr>
<td>1961–1970</td>
<td>-12.0%</td>
</tr>
<tr>
<td>1971–1980</td>
<td>-26.5%</td>
</tr>
<tr>
<td>1981-1990</td>
<td>-22.3%</td>
</tr>
<tr>
<td>1991-2000</td>
<td>-22.3%</td>
</tr>
<tr>
<td>2001–2010</td>
<td>-22.3%</td>
</tr>
</tbody>
</table>

**Commercial and Institutional Buildings – 2008 Intensity (OEE, 2010)**

<table>
<thead>
<tr>
<th>Sector</th>
<th>Electricity (MJ/m²)</th>
<th>Natural Gas (MJ/m²)</th>
<th>Fuel Oil (MJ/m²)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wholesale Trade</td>
<td>914.7</td>
<td>840.1</td>
<td>23.5</td>
</tr>
<tr>
<td>Retail Trade</td>
<td>932.8</td>
<td>854.5</td>
<td>23.9</td>
</tr>
<tr>
<td>Transportation and Warehousing</td>
<td>675.0</td>
<td>728.1</td>
<td>72.3</td>
</tr>
<tr>
<td>Information and Cultural Industries</td>
<td>894.4</td>
<td>681.3</td>
<td>131.1</td>
</tr>
<tr>
<td>Offices</td>
<td>752.2</td>
<td>691.8</td>
<td>61.3</td>
</tr>
<tr>
<td>Educational Services</td>
<td>894.4</td>
<td>770.0</td>
<td>63.5</td>
</tr>
<tr>
<td>Health Care and Social Assistance</td>
<td>1,352.2</td>
<td>1,145.4</td>
<td>147.3</td>
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<tr>
<td>Arts, Entertainment and Recreation</td>
<td>999.3</td>
<td>767.2</td>
<td>147.8</td>
</tr>
<tr>
<td>Accommodation and Food Services</td>
<td>1,242.2</td>
<td>1,235.0</td>
<td>73.8</td>
</tr>
<tr>
<td>Other Services</td>
<td>855.5</td>
<td>718.9</td>
<td>91.1</td>
</tr>
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</table>


<table>
<thead>
<tr>
<th>Sector</th>
<th>Electricity (MJ/m²)</th>
<th>Natural Gas (MJ/m²)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wholesale Trade</td>
<td>434.6</td>
<td>-22.9</td>
</tr>
<tr>
<td>Retail Trade</td>
<td>559.5</td>
<td>353.2</td>
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<tr>
<td>Transportation and Warehousing</td>
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<td>232.8</td>
</tr>
<tr>
<td>Information and Cultural Industries</td>
<td>496.2</td>
<td>-22.9</td>
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<tr>
<td>Offices</td>
<td>380.9</td>
<td>-39.3</td>
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<tr>
<td>Educational Services</td>
<td>305.0</td>
<td>9.2</td>
</tr>
<tr>
<td>Health Care and Social Assistance</td>
<td>400.1</td>
<td>253.3</td>
</tr>
<tr>
<td>Arts, Entertainment and Recreation</td>
<td>485.3</td>
<td>-22.9</td>
</tr>
<tr>
<td>Accommodation and Food Services</td>
<td>495.1</td>
<td>84.8</td>
</tr>
<tr>
<td>Other Services</td>
<td>423.1</td>
<td>-39.3</td>
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</table>

**Commercial and Institutional Buildings – Energy Savings From Retrofits**

<table>
<thead>
<tr>
<th>Sector</th>
<th>BAU%</th>
<th>Aggressive%</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wholesale Trade</td>
<td>10%</td>
<td>31%</td>
</tr>
<tr>
<td>Retail Trade</td>
<td>10%</td>
<td>28%</td>
</tr>
<tr>
<td>Transportation and Warehousing</td>
<td>10%</td>
<td>31%</td>
</tr>
<tr>
<td>Information and Cultural Industries</td>
<td>10%</td>
<td>30%</td>
</tr>
<tr>
<td>Offices</td>
<td>10%</td>
<td>30%</td>
</tr>
<tr>
<td>Educational Services</td>
<td>10%</td>
<td>30%</td>
</tr>
<tr>
<td>Health Care and Social Assistance</td>
<td>10%</td>
<td>24%</td>
</tr>
<tr>
<td>Arts, Entertainment and Recreation</td>
<td>10%</td>
<td>33%</td>
</tr>
<tr>
<td>Accommodation and Food Services</td>
<td>10%</td>
<td>24%</td>
</tr>
<tr>
<td>Other Services</td>
<td>10%</td>
<td>27%</td>
</tr>
<tr>
<td>Wholesale Trade</td>
<td>GDP Growth</td>
<td>Space (m²)</td>
</tr>
<tr>
<td>-----------------</td>
<td>------------</td>
<td>------------</td>
</tr>
<tr>
<td>10.4%</td>
<td>$1,993.04</td>
<td>9,486,293</td>
</tr>
<tr>
<td>Retail Trade</td>
<td>6.9%</td>
<td>$668.20</td>
</tr>
<tr>
<td>Transportation and Warehousing</td>
<td>5.2%</td>
<td>$1,632.15</td>
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<tr>
<td>Information and Cultural Industries</td>
<td>6.9%</td>
<td>$3,454.68</td>
</tr>
<tr>
<td>Offices</td>
<td>51.9%</td>
<td>$1,424.22</td>
</tr>
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<td>Educational Services</td>
<td>5.6%</td>
<td>$700.72</td>
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<tr>
<td>Health Care and Social Assistance</td>
<td>6.6%</td>
<td>$1,696.24</td>
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<tr>
<td>Arts, Entertainment and Recreation</td>
<td>1.2%</td>
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<tr>
<td>Accommodation and Food Services</td>
<td>2.3%</td>
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<td>Other Services</td>
<td>3.1%</td>
<td>$2,202.57</td>
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</table>

<table>
<thead>
<tr>
<th>Fossil Fuel Emissions Factors</th>
</tr>
</thead>
<tbody>
<tr>
<td>Gasoline Emissions Factor (t / TJ)</td>
</tr>
<tr>
<td>Natural Gas Emissions Factor (t / TJ)</td>
</tr>
<tr>
<td>Fuel Oil Emissions Factor (t / TJ)</td>
</tr>
</tbody>
</table>

5.2 Results & Discussion
From the scenarios presented above, high and low scenarios for direct GTA GHG emissions (i.e. neglecting embodied emissions from materials, food and energy) can be derived and are presented in Figure 5.4. These incorporate the best- and worst-case scenarios that were modeled in each sector (residential buildings, non-residential buildings and private transportation).

Examining the high scenario, the greatest source of emissions (50% of 2050 emissions) was from the commercial/institutional building sector, stemming from continued growth in GDP. The next greatest contribution came from the transportation sector, due as well to increased GDP and the application of the scenario which incorporated significant ICE vehicles (30% of 2050 emissions). Finally, residential buildings contributed roughly 20% of 2050 emissions in the “high” scenario, owing to higher carbon electricity, modest building code improvements and high-range annual population growth (2%). This case results in a 66% increase from the 2010 GHG emissions estimate.

The low case is a 60% reduction from 2010 emissions; as well, this change is also 60% below an estimate of 1990 GTA emissions. The equivalence between 2010 and 1990 emissions can be attributed to the reduction in carbon intensity of the electricity grid, but also does not include emissions from waste management. The greatest source of emissions reductions relative to the high case is the commercial/institutional sector. These can be attributable to the aggressive cuts in energy intensity in new buildings, as well as the deeper efficiency gains from retrofits. No
scenario assessed above was able to reach the regional emission reduction target of 7Mt CO₂ (80% below the 1990 baseline).

Table 5-2: High and low emissions scenarios for the GTA from the PURGE model

Results from all scenarios from the three sectors assessed are seen in Figure 5.5. A few strategies can be identified as important from each plot. From transportation, it is clear that any future that includes conventional ICEs (even those that are more efficient) as an important component of the vehicle fleet will not result in emission reductions on the scale suggested by the IPCC (2007). From Figure 5a, all T-2 scenarios result in future emissions greater than 15 Mt, while focusing on electric-based vehicles tend to result in 2050 emissions lower than 5 Mt. The lowest emissions result in 2050 from transportation when applying the MIT “No Clear Winner” scenario was 15.9 Mt (assuming low GDP growth and a low carbon intensity electricity grid). Alternatively, biofuels that approach net carbon neutrality over their life cycle could also achieve significantly lower emissions. This suggests that transitioning to a fleet based on electric propulsion or a low-carbon fuel is vital to reducing emissions.

The case in which hydrogen vehicles become a dominant technology mostly resulted in emissions less than the PHEV / BEV / HEV mix, due to the elimination of HEVs; comparing T1 & T3 in the most effective emissions reduction case, using “Electricity-2” and an average GDP
growth of 1%, the latter scenario resulted in 2.8 MT CO$_2$, or 10% below the former (Figure 5.5a).

Since there is also potential for reductions in vehicle usage due to a projected expansion of regional transit, it is of interest to examine the impact of mode shifting on 2050 GHG emissions. The PURGE model uses 23,000 km as the vehicle kilometres traveled (VKT) in the first year of ownership (which then declines exponentially over the life of the vehicle). Metrolinx (2009) has targeted a reduction in personal vehicle work trips by approximately 35% by 2035, leading to the development of a scenario modeled here where annual vehicle use will decline by the same margin. With this lower annual VKT, emissions are reduced to 1.8 Mt, slightly greater than the 2050 target of 1.7 Mt (an 80% reduction in 1990 transportation emissions; applying “T3”, the low population and economic growth projections, and “Electricity-2”; see Figure 5.6). It should be noted that this does not consider emissions from public transportation, which Metrolinx projects to be roughly 1/3 of all work trips in 2035, nor does it consider non-work trips, making the 35% reduction in annual VKT an optimistic projection (**need a reference stating share of non-work VKT made by personal vehicles**).
Figure 5-4: GTA GHG Emissions in 2050 from a) Transportation; b) Residential Buildings (Aggressive Building Code); c) Commercial/Institutional Buildings (Aggressive Retrofits)
Residential buildings demonstrate the need for the aggressive building code updating, but neither of the schedules for code renewal applied above were sufficient to curb emissions from the growing population expected in the GTA. The lowest achievable residential building stock emission scenario from the options applied was a 47% reduction below 2010 levels. If one were to apply a net zero standard by 2026 (utility connected, with no offset credits for electricity generated; Christiansen, 2007) and a Nordic passive house standard by 2031 (a 60% intensity reduction followed by a 40% cut, respectively; this would leave building thermal energy use at roughly 10% of new construction in 2010) to the scenario that achieve the greatest reductions, residential building emissions could be reduced to 51% below 2010 levels. To further reduce this total, total potential reductions for towers modeled by Kesik and Saleff (2009) were applied to all MUR in the GTA, regardless of vintage (85.46% and 16.56% of natural gas and electricity use, respectively), emissions were only reduced to 56% below 2010 levels. This emphasizes the need for deep retrofits to the existing single-family building stock or faster demolition and reconstruction with lower energy buildings (which would be more capital intensive, but also

Figure 5-5: Transportation GHG Emissions using the Metrolinx Demand Reduction Scenario
more effective; Dong et al., 2005), as these contribute more than 80% of natural gas energy demand in 2050.

Commercial and institutional buildings appear to present the biggest challenge in reducing GHG emissions. Certain assumptions may tie commercial growth too strongly to increases in emissions, such as proportionality of GDP growth with floor area increases (spread evenly across all sectors) or that no loss in C/I floor area occurs other than in recession years (which were not modeled). As well, GDP intensity ($\text{GDP}/m^2$) is also held constant. To address this, a model run was completed which applied a recent patterns in $\text{GDP}/m^2$ change. With the changes in GDP intensity of the commercial sector, floor area of existing buildings are reduced and a 55% reduction is achieved over 2010 GHG emissions (though this scenario also applies the aggressive new building construction, low GDP growth and carbon-free electricity grid scenarios). The comparable scenario with GDP intensity held constant yields an emissions decrease of 47%.

What becomes evident is that incentives to renovate the existing C/I building stock are required; the existing system where the disconnect between property ownership and utility costs exists will not address this on a wide scale. As well, aggressive changes to the building code are also required, though the potential exists for the rental market to drive change in this regard given the rapid growth in LEED-registered construction in the non-residential sector; 20% of all new construction in this sector in 2008 was LEED-registered (Yudelson, 2010). Finally, the scenarios for high performance buildings in certain sectors are likely unrealistic; as an example, the model assumes all new construction in the “Offices” sector will be zero-energy by 2050. This is unlikely if multi-storey commercial spaces are still constructed, as there are limitations on the height of net-zero buildings (Phillips et al., 2009). This implies that the emissions reductions must come through the elimination of physical locations in providing commercial/institutional services, deeper retrofits or demolition and reconstruction of existing service spaces.

Rising affluence may also put upwards pressure on GHG emissions, given the market desire for increasing residential floor area and decreasing occupancy rates (see Table 5-1; Statistics Canada, 2011). These are not assessed in the scenarios above and could diminish GHG emissions savings even further.
5.3 Conclusions

The PURGE model provides a means by which to examine emissions scenarios on the urban scale using data that is generally accessible. By applying scenarios that have been drawn from literature, the projections made here suggest that it will be difficult to attain emission reductions on the order of 80-95% below 1990 levels even with aggressive changes to technology. Fundamental changes in the means by which energy services are delivered must be considered in order to reach these difficult goals, especially in the face of increasing populations and economic growth (which will likely further drive consumption).

Transportation sector analysis demonstrates that any significant ICE-based presence in the vehicle stock will overwhelm adoption of lower-carbon vehicle technologies. In the building sectors, a significant focus must be made on the deep retrofitting or replacement of existing buildings as these represent the largest barrier to achieving an 80% emissions reduction. Finally, besides from sink enhancement (which is generally limited in its scope due to cost or relatively small carbon storage capacity; Mohareb & Kennedy, 2011), pursuing long-term demand reduction strategies must be coupled with technological change in order to lower emissions to levels that will mitigate the risk of climatic change.

Concepts such as dematerialization of the economy (when considering a far greater proportion of developed world emissions occur outside of urban boundaries; Kennedy et al., 2011) and aggressive behavioural change (private transportation and building conditioning energy demand reduction) will also likely be important. It is possible that technological change in information systems could facilitate the acceptance of such strategies, such as through improved telecommuting, greater emphasis being placed on digital consumer goods, or “smart” buildings. There is no clear or simple strategy that can be applied in the pathway to a low-carbon future, though it is clear that action must be swift and aggressive in its approach. Additionally, the importance of adaptation measures to meet the severe implications of failing to reduce GHG emissions is emphasized by the improvements.
References


6 Conclusions

Cities have been amongst the earliest leaders towards a low carbon future. Through their initiatives, municipal governments have spread awareness and begun to direct policy to consider their energy and GHG emissions implications. This has been shown by Kennedy et al., (2011), with a number of global cities already demonstrating that emissions can be cut through decisive action; in recent years, Berlin, NYC and the GTA have reduced absolute emissions 5%, 2.7% and 2.4%, respectively. While the latter two achieved their reductions over 4 years, Berlin was able to achieve its reduction in 3 years. This suggests that rapid, significant GHG reductions can be achieved in the short term. However, there is uncertainty whether deeper, long-term emissions cuts are within reach for cities and what strategies can be employed to achieve mitigation on this scale.

It is clear that political will exists for these reductions at the urban scale, with the participation of over 200 partners in FCM’s Partners for Climate Protection, and 1700 cities globally who have joined the World Mayor’s Council on Climate Change (FCM, 2010; World Mayor’s Council, 2011). The technological lock-in observed in society as a whole is the major barrier to overcome. With certain energy service provision technologies being long-lived, especially buildings and transportation, lag times are likely to be observed in reducing the emissions intensity of certain sectors. This will be especially true in the developed world, where the degree of lock-in is greater due to well-established infrastructure systems that facilitate the energy-intensive status quo.

The focus of this thesis is to examine the scope for GHG emission reductions in cities, as well as the means to which a low-carbon future can be achieved. Specifically, the goal was to answer the following questions.

- What impact do boundary and/or methodological selection have on the quantification of GHG emissions from waste?
- How significant are carbon sinks within the urban environment?
- How can carbon sinks be classified, based on the temporal and spatial boundaries applied?
• What are the current options to increase the magnitude of carbon sinks within the urban boundary?

• How can future emissions from cities be quantified by examining existing/future technology stocks?

• What magnitude of emissions reductions can be expected based on current technological adoption trends and policies/targets suggested by various levels of government with jurisdiction in the GTA?

• Based a literature review that suggests possible technological change pathways, how do different scenarios in the GTA impact GHG emissions to the year 2050?

The answers that were discovered to these questions over the course of this research are summarized below.

6.1 Summary of Chapters

**What impact do boundary and/or methodological selection have on the quantification of GHG emissions from waste?**

The ambiguous answer to this question is that the impact is dependent on the comparison being conducted and the relevance of the underlying model assumptions to the purpose of the quantification. Firstly, looking at the most simplistic methodology (FCM-PCP), simplistic methodologies that apply an emissions factor per unit of waste deposited should not be applied for inventorying purposes other than simply as an early approximation prior to a more rigorous approach. As well, methane commitment (MC) methodologies, whose temporal boundaries differ from the waste-in-place (WIP) approach, are useful for planning purposes. MC approaches do not provide an estimate of emissions within the inventory year and hence are not suitable for emissions inventorying for reporting purposes.

In instances where the quantity of waste has been declining, an underestimate of the present year’s emissions will occur when the MC approach is used. This is due to the future projection used as the temporal boundary, which neglects emissions occurring in the inventory year from previously landfilled waste. Conversely, when a MC-based method is used in the context of an
increasing trend towards using landfill for waste disposal, an overestimate will occur in the
inventory year, with previous years contributing fewer emissions in the inventory year than
would be observed during the residence time of the present year’s waste.

**How significant are carbon sinks within the urban environment?**

The two sink types analyzed (direct and embodied) proved to be relatively minor in the case of
the GTA, less than 5% of total scope 1 and 2 emissions in the year 2005. The greatest
contribution came from landfilled waste and regional forests. The same is likely to be observed
in most global cities where non-renewable energy meets demands for energy services.

**How can carbon sinks be classified, based on the temporal and spatial boundaries applied?**

The quantification of carbon sinks in the urban environment demonstrated a process by which
carbon sinks are categorized. Direct sinks, those through which the sequestration process occurs
within the urban boundary (such as trees and soil), have a spatial boundary that must match that
of the jurisdiction. As well, the temporal boundary of the storage process lies within the
inventory year. Contrarily, embodied sinks, driven by activities/decision occurring within the
spatial boundary of the jurisdiction conducting the inventory, are those where the processes that
store carbon (or create the conditions for which carbon storage can occur, such as is the case with
cement) are independent of the spatial boundary; however, demand for these sinks occurs within
this boundary. Embodied sinks use temporal bounds that project forward and are based on
assumptions given conditions at the time of inventorying.

**What are the current options to increase the magnitude of carbon sinks within the urban
boundary?**

As stated previously, regional forests and landfills represent the largest sinks within the GTA.
While sink expansion presents an opportunity for moving towards a more balanced carbon cycle,
it is unlikely that either of these two sinks would be part of this solution. The sink provided
through the deposition of degradable organic carbon (through paper, food scraps, and harvested
wood products) in landfills represents a narrow accounting of the carbon cycle. The lifecycle
energy and emissions savings from alternative treatment options (i.e. incineration, bioreactor
digesters, or recycling) have the potential to be much greater than the carbon sink potential for removing them from the waste stream (Kaplan et al., 2009, Christensen et al., 2009).

Regional forest expansion is unlikely due to existing land use pressure for development, as well as for the maintenance of existing agriculture. Urban forests (trees grown within settlements) do have the potential to expand and regional municipalities have targeted their enhancement over time. As well, there is the potential to enhance new forms of carbon sinks, including artificial trees, flue-gas sequestration (either using conventional carbon capture and storage or as a feedstock in polymers), biochar or using green roofs. This could increase carbon stocks in the GTA in coming years, though the permanence of these sinks must be considered so as not to release stored carbon through shifting sink management practices.

**How can future emissions from cities be quantified by examining existing/future technology stocks?**

The PURGE model presented in Chapter 4 provides a means by which carbon emissions of the existing technology stock is estimated using historic data, taking a bottom-up perspective. Energy requirements from vehicle stocks are developed using Ontario sales data, coupled with average vehicle fuel consumption by year and US annual production of cars and light trucks. The residential building sector energy consumption is tabulated using housing stock data from Statistics Canada, in addition to energy consumption data by era of construction from the Office of Energy Efficiency’s (OEE) End Use Database. Commercial sector energy consumption in the GTA uses OEE data on Ontario sector-specific energy consumption, scaled using with the proportion of the GTA’s GDP coming from each sector, and energy intensity per unit of floor area.

All sectors are subject to growth in energy demand relating to increases in population/GDP. Private transportation sector GHG emissions can be projected into the future using literature available on fuel consumption of emerging vehicle technologies and estimating their rate of adoption based on the US data on hybrid vehicle sales (applying a logistic diffusion curve). Residential retrofits use average retrofit data from the Eco-energy retrofit data, where new construction (Dong et al., 2005) are based on current and projected changes to building codes. Commercial buildings energy demand changes are based on energy savings from projected retrofits and the influence of LEED buildings on new construction.
Waste sector emissions are estimated based on current diversion practices and future strategies based on infrastructure currently being planned in the GTA (such as incineration and anaerobic digestion). Diversion projections apply current diversion rates and approaches that would achieve the greatest reduction in emissions from various waste stream components.

Finally, regional forest sector carbon storage in biomass and dead organic matter are calculated using region specific growth models from the Carbon Budget Model of the Canadian Forest Service, based on existing forest cover and projections for enhancement. Urban forest carbon storage is estimated using the IPCC 2006 guidelines for biomass within settlements and current policy for increasing canopy cover.

**What magnitude of emissions reductions can be expected based on current technological adoption trends and policies/targets suggested by various levels of government with jurisdiction in the GTA?**

Based on current strategies being explored in the GTA, changes in technologies will likely be insufficient to achieve the long term emissions reductions that are suggested by the IPCC. Greatest shortfalls in reductions are attributable to the building stock, as there are limitations to what retrofits can achieve and the structures themselves are long-lived. The IPCC targets are used as the aspirational goals of the City of Toronto, as well the federal government (though it is a 60-70% reduction using a different baseline). However, that the current policies do not reach this goal is not meant to suggest their failure, rather to emphasize the need for strengthening long-term planning. Though it is difficult to predict the exact technology set that is required in the GTA to reduce GHG emissions to levels that are 80% below the 1990 baseline, Chapter 4 suggests a means to which the types of energy demands or emissions intensities that we are currently target are just a starting point. The PURGE model can be used to emphasize the depth of mitigation activities that will achieve stated goals.

**Based a literature review that suggests possible technological change pathways, how do different scenarios in the GTA impact GHG emissions to the year 2050?**

Different scenarios seem to suggest that, despite aggressive targets for new construction, a low-carbon electricity grid and a vehicle stock that is not based on fossil fuels to provide motive services, existing buildings present a challenge due to space heating demands. Low-carbon heat
sources (e.g. a mixture of biomass, electricity, biogas, ground-source heat, solar thermal) must be employed, or else the IPCC emissions reduction target will remain beyond reach.

6.2 Significant Contributions

The contributions of this thesis can be summarized in 3 main points:

1. **Rigorous Examination of Waste Quantification** – The assessment of methodologies to quantify waste sector emissions was required due to the variety of approaches currently employed in inventorying GHG emissions from cities. This work provides rationale for which quantification methods to use for specific applications and provides a detailed case study to illustrate the differences.

2. **Classification and Quantification of Urban Carbon Sink** – Urban carbon sink quantification is generally limited to the publication of Urban FORestry Effects (UFORE) modelling reports and is currently not included in standard inventory reports. This work demonstrates how gross sinks can be quantified, categorizes sinks into “Direct” or “Embodied” and suggests ways that sinks could be enhanced.

3. **Development of the PURGE Model** – GHG Inventorying in cities has generally focused on retrospective emissions and have been generally top-down in their approach (quantifying fuel use and scaling based on sector data at the finest available granularity). The PURGE model provides a bottom-up approach which facilitates a dynamic quantification of emissions based on technology change. Future emission estimates are possible with the PURGE model as end-use demand data are applied and can be altered based on the adoption rates of alternatives. The model also applies parameters based on publicly available data, facilitating is application to other jurisdictions.

6.3 Future Research

This work provides means to quantify current and future carbon flows from cities, and is a starting point in examining ways to transition to the low carbon settlements necessary to mitigate climate change. A number of opportunities for further study were uncovered during the completion of this research. First, analysis is possible on the potential for deeper retrofits to existing building. Second, upstream emissions associated with consumption of resources within
cities, and projections for their future dynamics, warrant further study. Finally, an expansion of the case studies to other cities within Canada would be enlightening, as it would demonstrate the types of challenges facing different geographical regions.

6.3.1 Potential for Retrofits

During the course of this doctoral research, it was found that the dominant limiting factor to sizeable GHG emissions reductions was the slow rate of change observed in building stocks; as buildings are long lived and changes to energy codes slow in their approach, the legacy of the current and near-term building stock will inhibit the ability of cities to realize reductions in energy demand and GHG emissions. It would be useful to investigate the potential to reduce GHG emission from buildings through deep retrofits and new construction by creating a dynamic simulation tool to assess the life-cycle energy, economic and emission implications.

This would require the examination of costs and energy intensities of best practices in building technologies for retrofits and new construction, building on existing research (Dong et al., 2005, Kikuchi et al., 2009). As well, by creating generic building archetypes (within residential, commercial and institutional sectors), retrofitting activities could be assessed to determine the impacts of scalable strategies for the reduction of energy consumption in a range of climates. The strategies would employ many current and near-term fossil energy conservation options, both onsite (i.e. active solar thermal heating, insulation upgrades, and HVAC system efficiency improvements) and offsite (i.e. district energy). Options for onsite energy generation, such as building-integrated photo-voltaics and microturbines, can also be explored where suitable. Selection of retrofit strategies for each archetype will be done using economic and energy analyses over their life-cycle. Retrofitted buildings and new construction could then be modeled at the urban scale for a variety of global cities (in both developed and developing regions), looking at local data on existing building stock and population/economic growth trends.

After developing a set of options for retrofits, the creation of a systems dynamics model to estimate the adoption of these technologies. By incorporating consideration of capital and operating costs compared with current systems, as well as payback time, the rate of adoption of retrofits can be estimated using this bottom-up approach. By conducting this analysis across regions with varying climatic and economic conditions, insight will be gained on the location-specific policy mechanisms that could be used to leverage public and private investment in the
long-term reduction of building GHG emissions. This could include quantification subsidies or interest rates that could lead to wider adoption of these technologies.

### 6.3.2 Upstream Emissions

The focus of this research has generally focused on direct emissions (scope 1), with some consideration of upstream (scope 2 & 3, specifically electricity generation and landfills, respectively). However, there are many materials upon which cities are dependent that could be assessed for embodied energy. Cement, food, water and energy all have significant upstream energy components, and their inclusion in urban inventories could increase emissions by a non-trivial amount (Hillman and Ramaswami, 2010). It would be of interest to capture these upstream emissions based on projected urban consumption patterns related to transition to a low-carbon economy.

As stated previously, the expectation of technological change will bring about building retrofits, electric vehicle technologies and new methods of construction. Retrofitting often includes the increased use of insulating materials, some of which can result in significant life cycle emissions. (Harvey, 2007). Electric and hydrogen fuel cell vehicles, through the addition of batteries and fuel cells, also result in an increased embodied energy contribution to life cycle energy use, relative to conventional internal combustion engine vehicles (Lave et al., 2000). Finally, the impact of increased use of wood-based construction materials (such as cross-laminated timber) might have on urban scale upstream emissions relative to concrete-based construction.

### 6.3.3 Broader Application of PURGE

This research has focused on emissions inventories and future projections within the GTA. While this region provides an interesting case study that has the potential to inform other municipalities, there is value in assessing the challenges associated with cities located in different historical and geographic contexts. Examining the effects of cold environments (such as Winnipeg), as well as developments whose infrastructure is relatively new (such as Calgary) or older (Quebec City, as an example). Cities experiencing rapid development at present could use a tool such as PURGE to better understand the long-term climatic and energy impacts of current planning decisions to their 2050 GHG emissions.
6.3.4 Adaptation

Given that the BAU and the various additional scenarios developed for PURGE model emphasize that there is great difficulty in meeting the reductions targets in the context of a developed world urban context, the exploration of adaptation measures in the urban context is prudent. While these measures will vary between different cities and need careful consideration by local governments, it would be of interest to examine approaches that can be taken in a municipal policy case study. Implications of heavy rainfall events, increasing food prices, and higher seasonal temperatures should be examined, along with mitigation strategies to address these.

6.4 Summary

Cities appear to be cognisant of the need to address energy demand and GHG emission issues. The challenges that they face in balancing the direct carbon cycle within their boarders are stark, given constraints from investments made in current infrastructure and uncertainty in the financial capability to transition to alternatives. While this research has emphasized that options for increasing carbon sink capacity and reducing the carbon intensity of urban activities exist, their rate of adoption will dictate whether or not humanity can reach the GHG emission reduction goals of 2050. For cities to achieve low-carbon future will require cooperation with senior levels of government, as well as international cooperation, in addressing the current externalization of the costs of climate change. Until market forces provide short and long term incentives to reducing energy demand and carbon intensity, the ability or desire of urban residents to select low-carbon options will remain limited. The true values of the enhancement of carbon sinks, the reduction of building energy intensity and the shift to more sustainable transportation options will not be apparent until the cost of GHG emissions is plain in each planning or technological
References


Appendices
Appendix A: Summary Tables

Table A.1: Values Applied to the USEPA WARM Model for Residential Waste

<table>
<thead>
<tr>
<th>Material</th>
<th>Tons Generated</th>
<th>Tons Recycled</th>
<th>Tons Landfilled</th>
<th>Tons Combusted</th>
<th>Tons Composted</th>
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</thead>
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<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
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<td>Glass</td>
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<td>-</td>
</tr>
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<td>-</td>
<td>-</td>
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<td>-</td>
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<td>Food Scraps</td>
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<td>Yard Trimmings</td>
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<td>Grass</td>
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<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Leaves</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Branches</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
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<td>Mixed Paper (general)</td>
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<td>-</td>
<td>-</td>
<td>-</td>
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<td>-</td>
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<td>31,622</td>
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</tr>
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<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Tires</td>
<td>3,434</td>
<td>-</td>
<td>3,434</td>
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</table>
Table A.2: Emissions Factors for Various Landfill Options as Applied to the U.S. EPA WARM Model, as well as Net Emissions from Landfilling (Source: USEPA, 2006)

<table>
<thead>
<tr>
<th>Material</th>
<th>(a) Net GHG Emissions from CH₄ Generation (MTCE/Wet Ton)</th>
<th>(c) Net Carbon Storage (MTCE/Wet Ton)</th>
<th>(d) GHG Emissions From Transportation (MTCE/Wet Ton)</th>
<th>(e) (f = b + c + d) Net GHG Emissions from Landfilling (MTCE/Wet Ton)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Aluminum Cans</td>
<td>0.00</td>
<td>0.00</td>
<td>0.00</td>
<td>0.00</td>
</tr>
<tr>
<td>Steel Cans</td>
<td>0.00</td>
<td>0.00</td>
<td>0.00</td>
<td>0.00</td>
</tr>
<tr>
<td>Copper Wire</td>
<td>0.00</td>
<td>0.00</td>
<td>0.00</td>
<td>0.00</td>
</tr>
<tr>
<td>Glass</td>
<td>0.00</td>
<td>0.00</td>
<td>0.00</td>
<td>0.00</td>
</tr>
<tr>
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<td>0.00</td>
<td>0.00</td>
<td>0.00</td>
</tr>
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<td>0.00</td>
<td>0.00</td>
</tr>
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<td>PET</td>
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<td>0.03</td>
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<td>Textbooks</td>
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<td>0.15</td>
<td>0.56</td>
</tr>
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<td>Dimensional Lumber</td>
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<td>0.08</td>
<td>0.05</td>
<td>0.17</td>
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<td>0.05</td>
<td>0.17</td>
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Note: MTCE = Metric Tonnes of Carbon Equivalent; Wet Tons are in Short Tons