THE EFFECT OF CLIMATE POLICIES ON LOCAL AIR POLLUTION:

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APPROVAL

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ABSTRACT

In 2002 Canada ratified the Kyoto Protocol, committing to reduce greenhouse gas emissions (GHGs) to combat climate change. Leading up to ratification, and continuing today is a heated debate surrounding the cost of climate policy. Evaluating the costs of reducing GHGs is complex, and estimates vary depending on how costs are defined, how uncertainty is treated, and whether or not ancillary costs and benefits are considered. Ancillary benefits or costs result in addition to the effects of the climate policy on its stated target. An important ancillary effect is the potential for climate policies to impact local air pollution. Caused by criteria air contaminants (CACs), local air pollution holds serious consequences for regional environments and human health.

A modelling tool was developed to simulate, through an integrated representation of the Canadian economy and energy system, the GHG-reducing actions induced by climate policy and the associated changes in CAC emissions. Criteria were established characterizing the ideal energy-economy ancillary effects estimation tool, including: technological explicitness, preference incorporation, disaggregated calculation of CAC emissions, and spatial resolution. The CIMS model served as the base modelling tool, and was enhanced with technology specific CAC emission factors. Incorporating CACs into CIMS represents the first attempt at estimating CAC emission changes in Canada with a behaviourally realistic, technologically detailed model.

The CAC pollutants added to CIMS include fuel-based, process–based, and fugitive sources of nitrogen oxides (NO_x), sulphur oxides (SO_x), volatile organic compounds (VOCs), carbon monoxide (CO), and particulate matter (PM). The

Dedication

For my parents,

who always made me feel

like I could do anything.

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Table of Contents

List of Tables

List of Figures

Introduction

For the past 15 years there has been a growing focus in the international arena on the threat of climate change and the role of increasing anthropogenic emissions of greenhouse gases (GHGs). The government of Canada has stated that climate change is 'the ultimate sustainable development issue' and in 2002 ratified the Kyoto Protocol, an international agreement that established GHG emission reduction targets for Canada of 6% below 1990 levels by 2008-2012 (Government of Canada, 2002). Concurrently, concerns about local air pollution and the serious consequences for regional environments and human health have increased.

In 1998 the federal and provincial governments initiated the National Climate Change Process (NCCP) to evaluate the cost of different options for achieving Kyoto emission reductions. A central focus of the process is the cost effectiveness of policies as well as the distribution of costs across the provinces and territories (Government of Canada, 2002). However, the evaluation of the costs of GHG reduction policies is not straight-forward. Cost estimates vary considerably with differing definitions of costs, treatment of uncertainty, and the consideration of ancillary costs and benefits.

A key component in evaluating the costs of reducing GHGs, is accounting for the ancillary effects that may occur and to what extent they offset, or further inflate, these costs. Ancillary effects are the benefits or costs that result in addition to the effects of the climate policy on its stated target (Pearce, 2000; Burtraw and Toman, 2001). One ancillary local air pollution decision makers are forced to craft policies based on incomplete information (Davis et al., 2000). By failing to take the full costs and benefits into account, the resulting climate policies may miss the opportunity to minimize the costs to society.

1.1 Background

Greenhouse gases and CACs, the groups of pollutants that contribute to climate change and worsening local air quality, are suited for simultaneous evaluation in climate policy analyses. Both GHGs and CACs are produced as a byproduct of fossil fuel combustion (Ayres and Walters, 1991). Furthermore, each of these pollutant groups causes significant economic, environmental and social impacts. Understanding the nature of these impacts and the differences and similarities between them further underlines the importance of considering both pollutant groups when evaluating environmental policy.

1.1.1 Greenhouse Gases and Climate Change

Greenhouse gases (GHGs) accumulate in the atmosphere and absorb infrared radiation from the earth that would otherwise be released to space, disrupting the cooling and heating cycles of the ecosphere (IPCC, 1996b). Some GHGs occur naturally, however, the increasing atmospheric concentrations of GHGs that are implicated in climate change are due to human activities such as deforestation and fossil fuel combustion. Furthermore, GHGs tend to mix evenly in the atmosphere, meaning that one unit of GHG emitted in Canada is one unit emitted globally in terms of its effect on climate. While the impacts of climate change are typically slow and long-term, they may be quite severe. Some of these impacts include: increased flooding in some areas and droughts in others, the migration of ecosystem boundaries, displacement of people, and increased pestilence and disease (IPCC, 1996a). The long-term impact of climate change on ecosystems and human welfare may be severe and is quite uncertain.

In order to understand the key factors that affect GHG emission production and to help identify ways to target GHG emission reductions, decomposition equations are often used. One such equation, called the “Kaya Identity”, is displayed in equation 1. The Kaya Identity asserts that changes in GHG emissions will result from changes in the GHG-intensity of energy use in the economy (*GHG/E*), the energy intensity of economic production (*E/Q*), the economic output per-capita, and the population size (*P*). The two final terms (economic output per-capita and population size changes) are considered much more difficult for governments to influence for mainly political, social, and economic reasons (Jaffe et al., 1999; Jaccard et al., 2002). Hence, policies hoping to stimulate a reduction in GHG emissions tend to focus more on the energy intensity of economic production (*E/Q*), and the GHG-intensity of energy use in the economy (*GHG/E*).

Equation 1. The Kaya Identity

SO_x, NO_x, VOCs and the smaller fraction of particulate matter ($PM_{2.5}$)[1] as these are known to have serious health impacts and contribute to the formation of acid rain and photochemical smog (Burtraw and Toman, 2001). Carbon monoxide is more important as an indoor air pollutant.

In comparison to GHGs, CACs differ in terms of the nature of their production, but also the nature of their impact on the environment. The production of CACs is more complicated than GHGs because the amount of pollutant emitted is less directly related to the quantity of fuel combusted. For example, the quantity of NO_x emitted per unit of fuel combusted will vary for different sizes of industrial boilers running at different operating temperatures (U.S. EPA, 2000). Thus, variations in process characteristics (like operating temperature) have a greater influence on the magnitude of CAC emissions than GHG emissions.

Another important difference between GHGs and CACs is the environmental effect they have. Where GHGs mix uniformly in the atmosphere, CACs behave in a more localized manner. Notably, CACs contribute to the formation of acid rain and

GHGs must carefully consider the effect of resulting actions on CAC production and the economic and environmental ramifications. Researchers have emphasized that an evaluation of the ancillary effects of climate policies on CAC emissions should not assume that these effects will necessarily be ‘benefits’ (Davis et al., 2000; Burtraw and Toman, 2001), as is commonly the case in the literature. Rather, careful attention should be paid to understanding which GHG-reducing actions will reduce CACs, and which actions will exacerbate them.

1.2 Why Ancillary Effects Matter to Climate Policy Design

Understanding the ancillary effects of climate policy is important for many reasons, including the potential to affect the speed at which climate policies are implemented, affect the planning of policy incidence, shift the relative desirability of policy options that target trading versus domestic reductions, and alter the ‘no regrets’ level of abatement (Davis et al., 2000; Krupnick et al., 2000; Pearce, 2000; Burtraw and Toman, 2001).

The importance of understanding the effect of GHG policy on local air pollutants is enhanced by the difference between the impacts of the two pollutant categories. Fighting climate change is a key aspect of federal environmental policy, yet is an uncertain goal with diffuse, intangible benefits that will be felt over the long term. Local air pollution is more tangible and equally severe (Ekins, 1996), only on a different spatial and temporal scale. Hence, if concentrated local benefits related to air quality can be realized from implementing GHG measures, climate policy implementation will likely occur in a more timely fashion (Pearce, 2000).

Furthermore, because of the local impact of CACs, the planning of policy incidence becomes more complicated than if GHGs are considered alone. First, if densely populated areas are targeted with more GHG reductions, the potential ancillary benefits could be much greater. Moreover, consideration must be given to existing international transboundary agreements, such as the Canada-United States Air Quality Agreement, to ensure that GHG policies do not result in increased emissions of pollutants targeted in these agreements (Heintz and Tol, 1996; Pearce, 2000). In the Canada-United States Air Quality Agreement both countries committed to reductions of NO_x and SO_x emissions (International Joint Commission, 2002); therefore, if a climate policy caused increased emissions of NO_x or SO_x this could place Canada in contravention of the agreement.

Considering the ancillary effects of climate policies also calls into question the relative desirability of targetting domestic versus international GHG measures. For example, a flexibility mechanism incorporated into the Kyoto Protocol is an international system of tradable permits (Government of Canada, 2002). When ancillary effects are considered, nations that would be ‘permit-buyers’ may re-evaluate their choice to invest in reductions in other countries when they could reap the additional benefits of improved local air quality associated with more domestic GHG reduction measures (Pearce, 2000; Lutter and Shogren, 2002).

Finally, and perhaps most importantly, considering ancillary effects can alter the level of ‘no regrets’ GHG abatement. ‘No regrets’ refers to the level of abatement that can be achieved if all GHG measures with no net cost to society are implemented (Dessus and O’Connor, 1999). When monetized ancillary effects are included in the calculation of net costs or benefits they may alter the no regrets level of abatement and thus change the number of measures that could be taken with no net loss to social welfare.

1.3 Ancillary Effects Estimation

A commonly followed approach to estimating the ancillary effects of climate policies was first suggested by Ayres and Walter (1991), and then further modified by Ekins (1996). This generalized analytical approach to ancillary effects estimation is illustrated in figure 1. The first step is to use CO_2 abatement models (e.g. energy-economy models) to evaluate the CO_2 emission changes and abatement costs, and the underlying changes in fossil fuel demand associated with a climate policy. Energy-economy models describe the relationship between the energy system and the economy and are often used to estimate the cost and CO_2 emission reductions associated with climate policies. Next, emission factors that relate the CAC emissions associated with the different fuels are used to estimate the resulting changes in CAC emissions.

Once the associated change in CAC emissions is calculated there are two alternative ways to estimate the resulting impact and associated ancillary costs and benefits of a climate policy. The simple approach indicated by step 3a in figure 1 involves multiplying the estimated change in CACs by aggregate unit values that describe the benefits per tonne of pollutant reduced ($/tonne) (Ayres and Walters, 1991; Ekins, 1996). These aggregate values indicate the ancillary cost or benefit associated with each tonne of GHG reduced by the policy. Alternatively, a more disaggregated, damage-function approach may be followed, as outlined in Burtraw and Toman (1997). In this latter approach (beginning with step 3b), CAC emission changes are translated into changes in the ambient air concentration of the different pollutants, followed by estimation of the effect on human and natural systems. Finally, the impact on human health and the environment is monetized to reflect the final ancillary cost or benefit of the GHG policy ($ / tonne of GHGs abated). The previously described aggregate approach (step 3a) is less time consuming and involves more simplifying assumptions than the latter, more rigorous damage function approach (Burtraw et al., 1999).

Figure 1. Analytical pathway for evaluating the ancillary effects of climate policy

1.4 Energy Economy Modelling and CAC Estimation

As discussed, anthropogenic GHG and CAC emissions are primarily a result of fossil-fuel based energy production and consumption. Therefore, climate policy analysts tend to focus on how policies can change the GHG-intensity of energy (*GHG/E*) and the intensity of energy use in the economy (*E/Q*). The objective of policymakers is to design technologies that are more efficient and rely increasingly on renewable or clean energy sources. Correspondingly, policymakers rely on tools to simplify the energy-economy system, and help them understand how policies will affect the choices of actors, and induce technological change (Jaccard et al., 2002).

Energy-economy models are one such type of tool used extensively in the past to evaluate climate policies, and as the first step in ancillary effects evaluation. These models represent the link between the economy and the environment by modelling how technology decisions affect *GHG/E* and *E/Q*, and how policies can alter these decisions, thereby changing the amount of emissions produced in the economy (Edmonds et al., 2000).

1.4.1 Bottom-up, Top-down and Hybrid Modelling

Energy-economy models are typically classified as 'top-down- or 'bottom-up' in their approach. Each category of model produces very different estimates of the cost and effectiveness of climate policies. Three assumptions that play a large role in creating the differences between top-down and bottom up model estimates include: a) how costs are defined and subsequently how actors in the economy respond to changing costs, b) how the direction and rate of technology change is represented, and c) how the baseline is defined (Azar and Dowlatabadi, 1999; Edmonds et al., 2000; Jaccard et al., 2003). The following paragraphs review the 'top down' and 'bottom up' energy-economy modelling approaches with two goals in mind: illustrating how the different treatment of the aforementioned assumptions affect the change in emissions and costs estimated by the these models, and developing a list of criteria that can help evaluate the usefulness of energy-economy models as tools to help evaluate the ancillary effects of climate policy.

Bottom-up

Bottom-up analysis, most frequently applied by engineers and systems analysts, focuses on the alternative technologies that are available to provide energy services, and how increasing diffusion of these technologies can result in changes in energy use and emissions. Correspondingly, a detailed account of current and future technologies is included in the model, including cost (financial) and performance (efficiencies) characteristics (Jaccard et al., 1996).

The speed and direction of technology change in bottom-up models is driven by the differences in cost and efficiency of competing technologies. It is assumed that consumers will choose the option with the lowest *ex-ante* (anticipated) estimate of financial costs, causing technologies that are more energy efficient to penetrate relatively quickly because their energy-costs are lower than a similar, less efficient alternative (Edmonds et al., 2000; Jaccard et al., 2003). However, the bottom-up approach is criticized for assuming that the full social cost of switching between technologies can be represented by a simple *ex-ante* estimate of the financial cost differences between these technologies. Technologies are not always perfect substitutes in the eyes of consumers, and may differ in ways that are not captured by a single financial estimate (Jaffe and Stavins, 1994; Jaccard et al., 2003).

There are three main ways technologies may differ which are not captured by financial estimates. First, some technologies are perceived as being ‘risky’, with a greater potential for premature failure and long payback periods (as a result of high upfront costs). The value of not investing in a technology that is perceived as risky is termed ‘option value’: The consumer perceives a gain in value while postponing investment and waiting for additional information to inform their decision. Second, the service provided by two alternative technologies may not be identical in the eyes of a consumer. Jaccard et al. (2003) use the example of traditional incandescent versus more energy efficient compact fluorescent light bulbs. Some people consider the compact fluorescent a less different locations will face varying acquisition, installation and operating costs.

The failure to account for option value, consumers' surplus and market heterogeneity in bottom-up models when estimating the cost of technology alternatives results in an overestimated willingness of consumers to switch to GHG-reducing technologies. The result is that the social cost of climate policies is underestimated and a prematurely quick and inexpensive improvement in energy efficiency and energy intensity over time is predicted.

Assumptions regarding the baseline scenario (the characterization of the energy economy without a climate policy) also affect the results of bottom-up analyses. Bottom-up models typically assume that the baseline is relatively inefficient due to the presence of market barriers and market failures that hinder the adoption of energy-efficient technologies. For example, there are high transaction costs associated with learning about alternative, energy-efficient technologies as well as in acquiring and operating them which are not captured in the financial cost of a technology - meaning that the market will tend to under-supply them (Jaffe and Stavins, 1994; Jaccard et al., 1996). Bottom-up analysts generally assume that the policies to correct for these barriers and failures will have no net costs to society (as they are restoring economic efficiency by increasing the supply of more efficient technologies), and that other costs associated with these policies are minor (Edmonds et al., 2000).

Finally, because bottom-up models incorporate considerable technological detail they are less able to incorporate elements of economic feedback which is required to evaluate the macro-economic effect of policies. Instead, these models usually provide only a partial equilibrium (equilibrium is reached in one or a sub-set of economic sectors), in response to GHG policies (Jaccard et al., 2003). Thus, the full macroeconomic effects of a GHG policy targeted on a single sector may not be adequately portrayed by a bottom-up analysis.

When the characteristics of bottom-up models are considered together, the corresponding effect on estimates of emission reductions and total costs of climate policies can be deduced. Because of the combined effect of assumptions regarding the baseline, the lack of macro-economic feedback, the characterization of costs and the subsequent representation of the rate of technological change, bottom-up models typically result in low estimates of the total cost and high estimates of emission reductions from climate policy.

economically efficient (i.e. consumers have made welfare-maximizing decisions), any change induced by policy entails a loss of welfare, or a cost to society (Edmonds et al., 2000). Second, the use of price-consumption relationships to calculate the full costs to consumers of achieving a given emission target inherently includes lost consumers' surplus. Hence, top-down models produce higher cost estimates for emission reductions than bottom-up models.

However, the top-down approach is criticized for over-estimating the cost of emission reductions because the historical price-consumption relationship cannot accurately indicate the likely consumer preference for new technologies in the future. Emerging government policies induce development and commercialization of new, more efficient technologies, and associated economies of learning and economies of scale drive down the financial costs of these technologies over time. Correspondingly, the increased market penetration and falling costs of these technologies infer higher AEEI and ESUB values and the ability for GHG emission reductions to be achieved at a lower cost.

conventional bottom-up models incorporate considerable technological detail, but do not adequately incorporate consumers' preferences or equilibrium economic feedbacks, placing them in the top-left-front quadrant of the cube. Top-down models fall in the bottom-right-back quadrant of figure 2 because they are strong in equilibrium feedback and preference incorporation, but lack explicit representation of technologies.

Figure 2. Characterization of energy-economy models

(from Jaccard et al., 2003)

Hybrid

Energy-economy models that are strong in all three characteristics are most useful to policymakers, and fall in the upper- right-back quadrant of the cube pictured in figure 2. Hybrid models attempt to fill this role by addressing the criticisms of top-down and bottom-up models by acknowledging the importance of, and incorporating, technological detail, consumer preferences, and economic feedback. Analyses using hybrid modelling approaches typically produce estimates of costs and GHG emission reductions that fall in between bottom-up and top-down analyses of the same problem (Jaccard et al., 2003).

Hybridization has been approached from both the top-down and bottom-up directions. For example, top down models can gain more technological detail by further disaggregating sectors and using more detailed elasticity values. The Second Generation Model (Edmonds et al., 1991) is an example where production sectors were further disaggregated and more disaggregated ESUB values were used, thus gaining greater technology resolution. The level of technological explicitness of the SGM and other top-down hybrid models are still second to that of bottom-up models.

Bottom-up models begin with the benefit of considerable technological detail and can be enhanced with both greater economic and equilibrium feedbacks, and a representation of consumer preferences. The MARKAL model, a bottom-up linear programming model, has been enhanced with economic drivers (e.g. population growth, own price demand elasticities) to improve the economic feedback in the model. However, MARKAL is based on a least-cost approach, which assumes that consumers choose technologies with the lowest financial cost – ignoring consumers' surplus and option value. MARKAL would then fall in the upper-back quadrant, but towards the left reflecting the lack of realistic preference incorporation.

Bottom-up models may also incorporate consumers' preferences with the use of information from marketing research and discrete choice modelling studies. CIMS, a bottom-up hybrid model of the Canadian energy-economy, has incorporated economic feedback with the use of energy service elasticities and integrated supply and demand between energy and production sectors. CIMS has also incorporated parameters describing consumers' preferences informed with the use of discrete choice surveys, as well as revealed and stated preference surveys. Because CIMS includes technological detail, economic feedback as well as a realistic representation of consumers' preferences it falls farthest to the right in the upper-back quadrant of figure 2.

While this section has focused on the characteristics that make a useful energy-economy model, further characteristics are required in order to produce a model that is useful in evaluating the effect of climate policies on CAC emissions. These characteristics are discussed in following sections and include the aggregation level of coefficients used to calculate CAC emissions (section 1.5), and the level of geographical detail that the estimated changes are reported with (section 1.7).

1.5 Calculating CAC Emissions and the Aggregation Level of Emissions Factors

Following the chain of analysis outlined in section 1.3, the next step in ancillary effects estimation is to use the estimated changes in fuel demand from an energy-economy model to calculate changes in CAC emissions. Approaches to estimating CAC emission changes vary in terms of how well they include the process parameters that determine CAC emission intensity and the level of detail used to determine these changes. The following paragraphs discuss past approaches to estimating the ancillary effects of climate policy with a focus on the level of detail used in calculating and representing CAC emissions.

The level of aggregation in emission factors applied in different studies is a function of both the nature of the energy-economy model used in the first step of the analysis, as well as the focus and scope of the study in question. When relying on the outputs from a top-down energy-economy model, analysts have little choice but to apply aggregate emission factors, as the output from the model is limited to estimated aggregate changes in fuel demand. Burtraw and Toman (1997) summarize the modeling approaches taken in past studies, with the vast majority being top-down, national scale economic models relying on aggregate fuel based or sector based emission factors. Complainville and Martins (1994) is an exception to this case as they employed a top-down, multi-sector, multi-country dynamic applied general equilibrium model (GREEN) and combined this with emission factors that began as disaggregated factors that were then rolled-up into more aggregated, cross-sector emission factors.

More disaggregated models and hence more specific emission factors have been used in the past, but generally when the scale of the study is smaller, and often focused on regional electricity sectors. For example, Burtraw and Toman (1999) incorporated emission factors specific to the facility level, and also summarize the different models that have been applied to the U.S. electricity sector for the purpose of ancillary effects estimation.

In contrast, all Canadian, national-scale evaluations of the ancillary effects of climate policy published to date have begun with outputs from technology specific, hybrid energy-economy models (EHI, 2000), and then applied aggregate fuel-based emission factors. This approach was also taken by Syri et al. (2001) who used PRIMES, a hybrid, technologically detailed energy model for the European Union and incorporated aggregate, fuel based emissions factors.

Understandably, the reliance on aggregate emission factors is one way to maintain simplicity in a model, and prevent creating an overly complex representation of the system that would make understanding the underlying mechanisms more difficult (Ayres, factors precludes the use of the model to address a number of policy questions and eliminates a richness of detail that could better help decision makers understand why decreases and increases in CAC emissions can result from climate policies. As mentioned earlier (section 1.4.1), policymakers may prefer to use instruments that target specific technologies (such as regulations). If the emissions cannot be traced through the model to the associated technology, designing and evaluating these targeted policies will be a more difficult and less valuable exercise. This further emphasizes the value of energy-economy models that have a high degree of technological detail (i.e. bottom-up hybrids) in ancillary effects research. When these models are enhanced with equally detailed CAC coefficients the types of policies that can be addressed are more numerous and the richness of the analysis is improved.

1.6 Estimating Impact and Valuation of Costs and Benefits

The final steps in the ancillary effects analysis pathway include estimating changes in ambient air quality, determining the potential environmental and health impacts of theses changes, and ultimately monetizing these changes into costs and benefits (steps 4-6 in figure 1). As indicated, each of these steps involves considerable expertise and

The final two steps in the analysis are controversial, both in terms of the great uncertainty involved in estimating the dose-response to pollutants, and the valuation of human health effects and environmental damages (Davis et al., 2000). A detailed description of the literature surrounding these steps in beyond the scope of this report, but for a comprehensive review of the issues involved see Davis et al. (2000), Burtraw and Toman (2001), and Cifuentes et al. (2001).

1.7 Geographic (Spatial) Disaggregation

As indicated, in order to accurately estimate the air quality changes and the impact on the environment and humans, the geographic location of emission changes must be known in considerable detail. Davis et al. (2000) and Burtraw and Toman (1998) assert that the estimated benefits or costs associated with changes in CAC emissions will vary greatly depending on the geographic location and proximity to human populations. Thus the estimated change in CAC emissions predicted by energy-economy models will result in more accurate estimates of associated costs and benefits if they are spatially precise.

An example of geographically detailed estimation of the ancillary effects of climate policy is presented by Burtraw and Toman (1999), who use a location-specific, economic model of the electricity sector (named HAIKU). The model produces region-specific emission changes for the five, eastern North America Electricity Reliability Council (NERC) regions in the United States (each NERC region includes a number of states). The emission changes estimated by HAIKU were then fed into an integrated assessment model that determined the change in air quality, environmental and human impacts. A number of other similar studies, specific to the regional scale are outlined in Burtraw et al. (1999).

Canadian attempts at estimating ancillary effects of climate policy, as described in section 1.6, have relied on technologically detailed hybrid models (CIMS, MARKAL) which can produce estimates of GHG emissions that are specific to the sector-region scale (e.g. Ontario electricity sector). However, the sector activities and the related emission changes may be scattered across the province making it difficult to translate sector/region emission changes into changes in the air quality of a particular airshed.

The ideal level of spatial disaggregation in a model depends on the characteristics of the affected airshed (how big is it, is it split across two regions), and the location of affected populations in the airshed. But analysts must also consider the complexity of the modelling tool. Incorporating better spatial resolution into energy-economy models may greatly increase the data needs and the time it takes the model to calculate results. One compromise is to use a modelling tool that takes the emission changes estimated by an energy-economy model and disaggregates them to a level of finer geographic detail. With this approach the emission changes can be translated to a finer level of spatial resolution without adding cumbersome details to the energy-economy model itself.

1.8 Summary of Evaluative Criteria

Elements of the preceding discussion can be tied together to form a list of evaluative criteria that describe the characteristics of an energy-economy model that would be most useful in evaluating the ancillary effects of climate policy. These criteria include:

technological detail in the model through to the estimation of CACs. Finally, the level of geographic detail should be sufficient to associate the changes in CAC emissions to the appropriate airshed, and allow a more accurate determination of air quality changes, the subsequent impact on humans and the environment, and the resulting costs and benefits.

1.9 Uncertainty in Energy-Economy Modelling

Each step in the analytical chain to evaluate the ancillary effects of climate policy involves a degree of uncertainty. As asserted by Morgan and Henrion (1990), responsible policy analysts should always strive to characterize the limitations (uncertainties) associated with the ‘answers’ they provide. There are numerous relevant sources of this uncertainty in policy analysis, including: the type of model used to represent the complex relationships involved, the natural variability in the system being described, systematic errors such as bias and imprecision in estimating the parameters in the model, and a lack of information regarding future conditions and changes in parameter values (Morgan and Henrion, 1990). Thus in order to understand the total uncertainty involved in estimating technological change.

Characterizing the uncertainty in the ancillary effects estimation of climate policy is even less common a practice, yet is equally important in terms of understanding the overall effect of uncertainty on the ancillary costs and benefits of related policy (Davis et al., 2000; Burtraw and Toman, 2001). The uncertainties associated with these latter steps in the analytical chain (namely atmospheric concentration, impact estimation and valuation) are believed to be large but are rarely quantified. Whether the characterization of uncertainty is quantitative or qualitative, some indication of the effect of this uncertainty on the ultimate estimation of costs and emission reductions should be noted (Davis et al., 2000).

1.10 Research Objectives

The preceding paragraphs have established the need for climate policy analyses to consider the ancillary effects on local air pollution. Correspondingly, decision-makers need a way to keep track of how policies crafted to reduce GHGs can also affect the emission of CACs. The most common assumption in the literature is that measures to reduce GHGs will result in CAC reductions. In order to properly test this type of assumption, and to get a clearer idea of the magnitude of CAC emission changes, a tool is required that can track the actions stimulated by GHG policy and the corresponding changes in CAC emissions.

Hence, the objectives of this research project are:

1) To develop a Canadian energy-economy model capable of estimating GHG emissions and CAC emissions over time.
2) To use this model to evaluate the CAC emission changes associated with policies aimed at reducing GHG emissions.
3) To evaluate how well the developed modelling tool meets the outlined criteria for an effective ancillary effects evaluation tool.

Once developed, the proposed modelling tool will fulfill the first half of the analytical chain pictured in figure 1.

Methodology

The research objectives outlined in section 1.10 were pursued with an established hybrid energy-economy simulation model, already used to estimate the greenhouse gas (GHG) emissions and the costs associated with Canada's climate policy alternatives as part of the National Climate Change Process (NCCP)[3]. The following section (2.1) further describes the rationale for choosing the CIMS model in this research, and then details the structure and function of the model. In section 2.2 the approach taken to incorporating criteria air contaminants (CACs) into CIMS is described along with a discussion of how CAC emissions are calculated, the data sources used, and the challenges involved. In the closing sections the process of calibration is discussed along with a number of assumptions that were made, and the uncertainty surrounding the representation of CACs is discussed.

The preceding chapter established and discussed both how the ancillary costs and benefits of climate policies are calculated, with the focus on the role of energy-economy models forming the first link of this chain. A number of evaluative criteria were outlined and discussed in terms of how an energy-economy model can be most useful in informing climate policy and ancillary effects estimation (section 1.8). The criteria include:

- technological explicitness (detail),
- preference incorporation,
- disaggregated emission coefficients, and
- spatial disaggregation.

As indicated earlier, hybrid models incorporate the first two criteria, by bridging gaps between top-down and bottom-up approaches. The hybrid simulation model, CIMS, is a Canadian example of an energy-economy model that has been used in the past to evaluate climate policies, and provides a relatively disaggregated representation of

[3] See the report entitled "Integration of GHG Emission Reduction Options Using CIMS" by MKJA (2000) for a synthesis of the work done for the National Climate Change Process.

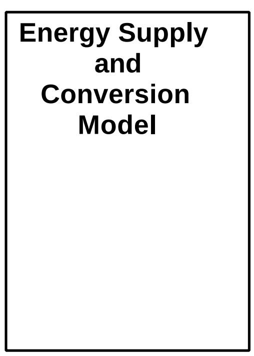
emissions changes to the sector-region level. The third criterion informed how CAC emissions were incorporated into the model, which is discussed in section 2.2. The fourth criterion is the most challenging for CIMS, as it is not a spatial model and may not on its own provide enough detailed information describing where emissions and emission reductions occur. Whether or not the sector-region emission estimates of CIMS are sufficient to inform the evaluation of ancillary costs and benefits will be further addressed in section 3.2.3 of the analysis.

2.1 Introduction to CIMS

CIMS is a simulation model, developed by the Energy and Materials Research Group at Simon Fraser University, which was designed to help policy makers understand feedbacks related to levels of market penetration.

As illustrated in figure 3, CIMS has three major components. The energy service demand component includes the residential, commercial / institutional, industrial and transportation sectors. The energy supply component includes conversion models of electricity generation, petroleum refining and natural gas processing alongside supply curves for fossil fuels and renewables. The macro-economic component includes energy service elasticity parameters that relate product and energy service demands to their costs. Note that for this study, the macro-economic feedback loop was disabled to permit the isolation of the direct emission reductions associated with policy alternatives.

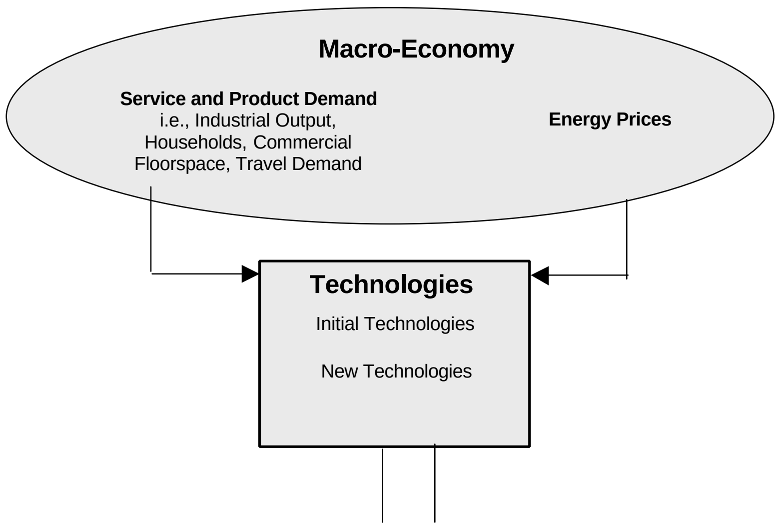
Figure 3. The major components of CIMS



2.1.2 Simulation Process

For this project, a CIMS simulation involves four basic steps (as illustrated in figure 4). First, energy service demand is forecasted in five-year increments (e.g., 2000, 2005, 2010, 2015, etc.).

Figure 4. Basic steps in a CIMS simulation



In each future period, a portion of initial-year equipment stocks is retired, following a time-dependent function. The remaining technology stocks are tested to see if retrofitting is desirable given the economic conditions and service demands (step 2).[4]

If new stocks are required because of the combined effect of equipment retirement and growing service demands, prospective technologies compete to determine which will contribute the remainder of the energy services (step 3). Technologies are allocated market share using a probabilistic function of life-cycle costs, including intangible preference related costs. Section 2.1.3 presents a detailed description of how market share is determined. Next the model iterates between energy demand and energy supply components until energy prices stabilise at equilibrium (step 4). The previous steps are started again in the next time period with an energy forecast demand that reflects the new conditions.

Since each technology has an associated net energy use, net emissions and costs, the simulation ends with a summing of these. The difference between a business-as-usual simulation and a policy simulation provides an estimate of the emission changes and cost of a given policy or package of policies.

2.1.3 Determination of Market Share

The equations that determine the proportion of new market share that a technology will capture are described below. In equation 2, the market share function (MS_{kt}) is a logistic relationship between the life-cycle cost of a given technology and all other technologies that compete to fulfill the same service demand.

[4] Retrofit options are characterized with the same financial and non-financial information as normal technologies, except that the capital costs of residual technology stocks are excluded, having been spent earlier when the residual technology stock was originally acquired.

Equation 2

$$MS_{kt} = \frac{LCC_{kt}^{-v}}{\sum_{k=1}^{z} LCC_{kt}^{-v}}$$

where:

MS_{kt}	=	market share of technology k for new equipment stocks at time t,
LCC_{kt}	=	annual life cycle cost of technology k at time t,
v	=	variance parameter,
z	=	total number of technologies competing to meet service demand.

The slope of the logistic curve is determined by the value of *v*; the magnitude of *v* describes the relationship between life-cycle costs and market share for different technologies. A high value for *v* (e.g. 100) implies that the lowest life-cycle cost technology will capture the entire market share. In comparison, a very low value for *v* (ex. *v* = 1) results in the market share being distributed evenly amongst competing technologies, regardless of their life-cycle costs.

The life-cycle cost for a specific technology is calculated using the following formula (equation 3).

Equation 3

$$LCC_{kt} = \left(\frac{CC_{kt} \times \frac{r}{1-(1+r)^{-n}}}{SO_k} \right) + O_{kt} + E_{kt}$$

where:

CC_{kt}	=	capital cost of technology *k* at time *t*,
SO_k	=	annual service output of technology *k*,
O_{kt}	=	operating cost of technology *k* at time *t* per unit of service output,
E_{kt}	=	energy cost of technology *k* at time *t* per unit of service output,
r	=	discount rate (time preference)
n	=	equipment lifespan

Equation 3 calculates the life-cycle cost (*LCC*) as a function of annualized capital costs, operating, and energy costs. The discount rate (*r*), determines the relative importance of capital costs versus operating costs in the total life-cycle cost of a technology. A higher discount rate places greater weighting on capital costs and results in a high LCC, while a lower discount rate will produce a lower LCC, given equal operating and energy costs. Hence, a high discount rate will hinder the ability of technologies with high capital costs and lower operating and energy costs to gain market share (competitive disadvantage). Because new, energy efficient technologies often have a high capital to operating cost ratio, a high discount rate will impede the market penetration of these technologies (Nyboer, 1997).

Capital costs are calculated using equation 4, which incorporates both financial and non-financial, or intangible costs:

Equation 4

$$CC_{kt} = FC_{kt}\left(1 + i_{kt}\right)$$

where:

FC_{kt}	=	financial cost of technology *k* at time *t*
i_{kt}	=	intangible cost factor of technology *k* at time *t*

The intangible cost factor (i_{kt}) is a value between 0 and 1 that serves as a multiplier, increasing the capital cost beyond simply the financial cost of a technology to reflect one or several factors such as identified differences in non-financial preferences (differences in the quality of lighting from different light bulbs) and perceived risks (one technology is seen as more likely to fail than another) of technologies (Jaccard et al., 2003).

A more detailed description of the CIMS simulation procedure, equilibrium algorithm, and inputs is available in Nyboer (1997) or can be obtained by contacting the author.

2.2 Incorporating Criteria Air Contaminants

The CACs incorporated into CIMS for this study include sulphur oxides (SO_x), nitrogen oxides (NO_x), volatile organic compounds (VOC), carbon monoxide (CO), and particulate matter (PM). Where possible (i.e. where data was available), particulate matter is further characterized by size, where PM_{10} is particulate matter less than 10 microns in diameter, and $PM_{2.5}$ has a diameter less than 2.5 microns.

As indicated in the previous section, emissions are calculated at the end of each time period once the supply and demand models have stabilized at equilibrium. Emission information is summarized in CIMS with the use of emission factors (EFs). Fuel related emission factors (EF_f) are coefficients that indicate the amount of a specific emission generated per GJ of a given fuel demanded. Fuel related emission factors are multiplied by fuel demand coefficients (GJ / unit output) and then by the total material or service output of the specified technology in order to arrive at total emissions generated for the technology. Process related emission factors (EF_p) relate the emissions generated to the service demand, material output or input to a technology or process, and are usually in units of tonnes of pollutant per unit output, input or service. Equations 5 and 6 are a generalized sample of the formulas used to calculate fuel-related and process-related CAC emissions from a technology.

Equation 5. Fuel combustion emissions

$$\text{Kg of pollutant generated} = EF_f \times FC \times O$$

Equation 6. Process related emissions

$$\text{Kg of pollutant generated} = EF_p \times O$$

where:

EF_p = process related emission factor (kg pollutant / unit output)

O = unit of output (ex. tonnes pulp, m2 floor space heated or cooled, etc.)

2.2.1 Fuel-related Emissions

As indicated in equation 5, each specific fuel (natural gas, heavy fuel oil, etc.) has a unique emission factor for each associated pollutant (NO_x, SO_x, etc.). Some EF_f's are further differentiated by sector; For example, industrial combustion of coal has a different emission factor for SO_x than combustion of coal by an electricity generating utility. Table 1 presents the general disaggregation of the fuel-related emission factors by fuel-type, emission, and sector. The letters (A, B, C) represent individual fuels, while the numbers (1, 2…) represent the different emissions. The actual fuel related emission factors used in the model are presented in Appendix B, table 1.

Table 1. Fuel combustion-related emission factors

Emission Type	**Units**	**Industry**			**Transportation**		
		A	**B**	**C**	**A**	**B**	**C**
1	Kg / GJ	$Ef_{f\ (1, A)}$	$Ef_{f\ (1, B)}$	$Ef_{f\ (1, C)}$	$Ef_{f\ (1, A)}$	$Ef_{f\ (1, B)}$	$Ef_{f\ (1, C)}$
2	Kg / GJ	$Ef_{f\ (2, A)}$	$Ef_{f\ (2, B)}$	$Ef_{f\ (2, C)}$	$Ef_{f\ (2, A)}$	$Ef_{f\ (2, B)}$	$Ef_{f\ (2, C)}$
Etc…		…			…		…

(where A, B… = fuels, 1, 2...= pollutants)

2.2.2 Process-related Emissions

As indicated in equation 6, process-related emission factors relate the amount of emissions generated to the material throughput or service demand met by a technology; however, these emission factors are not always exclusive of emissions resulting from fuel combustion. Because emission factors are developed by measuring the emissions that are released at some identified end point of a technology process it is difficult to discern which portion of measured emissions are directly related to fuel combustion and which are related to the process materials or characteristics (e.g. rotary kiln where combustion gases and process materials mix). Hence, process-related emission factors are used to characterize emissions in the following situations:

- The emission production is related to both the combustion of a fuel, the material components of a process and the incremental effect of each cannot be separated.
- The emission production is process dependent, and a small change in some aspect of the process can affect the amount of emissions generated.

Finally, process-related emission factors are truly technology specific, and permit the representation of greater heterogeneity in emission production than fuel-related emission factors.

2.2.3 Data

In order to incorporate CACs into CIMS, technology specific emission factors relating the amount of a pollutant generated to the energy and/or service level of a technology were used. However, due to a lack of sufficiently detailed Canadian data, the majority of the CAC emission data used in this study is adapted from the U.S. Environmental Protection Agency AP-42 and FIRE 6.23 databases of emission factors. The AP-42 databases are public and peer reviewed, and the most recent version available (5^{th} Edition) was consulted for this study.

The emission factors in the AP-42 / FIRE 6.23 are technology specific, and in process emission factor for that technology.

NO_x and VOC fugitive emissions are of particular importance in the natural gas processing and petroleum refining sectors. In the natural gas sector, actions are represented in lieu of specific technologies. An action in this case refers to an alternative way of fulfilling the service output, whether through conventional technologies, or with small process changes such as increased maintenance or feedback looping. No match for

combustion processes do not vary greatly between Canada and the U.S.. The process emission factors, in large part because they are so specific to material use and technology specifications, may vary greatly between Canada and the U.S. and are less reliable.

The sectors with the most data deficiencies include: chemical products, natural gas extraction, and "other manufacturing"[5]. Chemical products and "other manufacturing" are two industrial sub-sectors that have considerable process and fugitive emissions of VOCs, NO_x, and PM. However, information was not found that would enable the development of EFs for the chemical industry. For the "other manufacturing" sub-sector, the nature of aggregation in the CIMS model precluded the use of more disaggregated EFs from the AP-42 database. The natural gas extraction and transmission industry is not represented in CIMS as technologies, but rather is described as distinct possible actions (as described earlier) with consequent related changes in fuel demand and volumes of natural gas transferred. Each action implies some change in CAC emissions, where the change in emissions could not be related to a change in fuel consumption (i.e. in compressors, etc.) best judgment supported by engineering knowledge and relationships contained in the U.S. EPA (1995b) AP-42 document "Protocols for Equipment Leak Emission Estimates" were used to derive estimates in emission changes associated with the actions represented.

Abatement Technology Representation

CAC emissions are often controlled with the use of abatement technologies; however, there is no explicit representation of separate pollution abatement technologies such as venturi scrubbers, electrostatic precipitators, and baghouses in the model. Because the focus of this research project is on the CAC effects of GHG focused policies, the explicit inclusion of control technologies was not necessary.

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[5] "Other manufacturing" includes smaller industries that do not consume enough energy to fit into their own, larger sub-sector classification in the model, and includes activities such as rubber manufacturing, food and beverage production, and wood products manufacturing.

To represent the effect of pollution control technologies without adding them individually to the model, emission factors for various combinations of control technologies were tracked and applied to technologies where the regional context was deemed appropriate. For example, if B.C. cement manufacturers are known to use electrostatic precipitators to control particulate emissions from lime kilns, a lower, representative emission factor from the AP-42 database was applied. This approach ensures that the likely presence of abatement technologies and the effect on CAC emissions is represented in the model, without the complication of adding the individual technologies along with their detailed characteristics (capital cost, operating cost, energy requirements, control efficiency etc.). Including a detailed representation of individual CAC abatement technologies would be a useful future extension to this research as it would create the potential to evaluate the effect of policies targeted explicitly at CAC emissions with the model.

2.3 Calibration

Calibration is the process of evaluating model outputs against an established, external source or inventory and adjusting model parameters to ensure that the estimated baseline approximates the external estimates. The process of calibration is used both to refine the model, but also to ensure that the results can be compared to results from other similar modeling exercises. As part of the work of MK Jaccard and Associates (MKJA) using CIMS to evaluate climate policy alternatives for the National Climate Change Process, the energy demand and GHG emissions in the reference case were calibrated to 'Canada's Emissions Outlook: An Update' for 1999 (Analysis and Modelling Group, 1999). Fuel demand was calibrated to within 5% and GHG emissions to within 10%.

CAC emissions in the baseline were calibrated to Environment Canada's Residual

Discharge Inventory System (RDIS-II) for 1995. (An exception is on-road vehicles in the transportation sector which were calibrated to the SENES inventory of CACs over time, see section 2.2.3). This calibration ensured that the emissions estimated by CIMS for the base year (1995) approximate the values developed by Environment Canada which have been vetted through the provinces and stakeholders. A margin of error of ± 25% was allowed. An initial goal of calibrating to within 10% proved to be unrealistic as the two different methods (RDIS-II vs. CIMS) incorporate different assumptions regarding sector level activity and overall fuel use. Also, CAC emissions were not calibrated to future years because at the time of this report there was no consensus between the government and stakeholders regarding forecasted estimates of CACs over time.

CAC calibration was achieved on a specific sector/region basis by following a number of steps. If the pollutant emissions estimated were determined to differ by more than 25%:

a)

diverging estimates of CAC emissions from CIMS and RDIS-II. Without access to the associated RDIS-II assumptions, the changes made to calibrate the 1995 estimates of

CAC emissions from CIMS were in some cases arbitrary. A better comparison of the assumptions in the two models may provide additional clarity and improve the calibration.

Finally, the RDIS-II inventory records emissions from some sector/regions that are not included in CIMS, therefore only sector/regions included in CIMS were modeled. For example, the RDIS-II includes emissions from a variety of open sources (e.g. agriculture, forest fires, and structural fires) which are not included in CIMS.

2.4 Uncertainty

As discussed in section 1.9, in modelling work of this nature uncertainty is always a factor. The different types of models, the variables and the number of assumptions made regarding their value, all point to the fact that the estimated results reported are within a range of possible values. Uncertainty is further exacerbated by the fact that this is a first attempt at technology specific analysis of this kind in Canada, and that the assumptions of CIMS and RDIS-II could not be compared.

In order to facilitate the characterization of uncertainty involved in estimating CAC emissions a qualitative record of uncertainty for the data used from the AP-42 was established. Table 2 lists each of the indicators and their meaning, ranging from a value of ‘A’ for quite certain, to ‘E’ for very uncertain, and incorporate aspects of variability, bias and representativeness. This information can be used to examine the effect of uncertainty in emission factor data on the estimated changes in emissions.

Table 2. Uncertainty or quality indicators for U.S. EPA emission factors

Emission Factor Rating	**Meaning**
A (Excellent)	Factor is developed from validated source test data taken from randomly chosen facilities in the industry population. The source category population is sufficiently specific to minimize variability. Bias is low.
B (Above average)	Factor is developed from well and sufficiently validated test data from a "reasonable number" of facilities. While no specific bias is evident, it is not clear if the facilities tested represent a random sample of the industry. The source category population is sufficiently specific to minimize variability.
C (Average)	Factor is developed either using unproven methodology or lacking background information, using test data from a reasonable number of facilities. Although no specific bias is evident, it is not clear if the facilities tested represent a random sample of the industry. The source category population is sufficiently specific to minimize variability.
D (Below average)	Factor is developed as per the C rating; however, there also may be evidence of variability within the source population. Bias may be high.
E (Poor)	Factor is developed as per the C-rating and the method may be deemed unacceptable, but provides an order of magnitude estimation of the emissions from Tc -0.016iuA26 T.96 495.g.48 re f 102.72 495.36 126.3l at3

2.5 Estimating the Cost of a Policy

One of the key results from a policy simulation in CIMS is an estimate of the costs

Analysis

As stated in the research objective (section 1.10), the purpose of adding CAC emissions to CIMS is to track and understand the relationship between the GHG-reducing actions stimulated by climate policy, and the corresponding effect on CAC emissions. Thus, the analysis has been designed to focus on determining where synergies and antagonisms exist, the effect on regional CAC emissions, and the ramifications for policy.

Synergies occur when GHG policies stimulate actions that simultaneously reduce GHGs and CACs, while antagonisms are when these actions cause an increase in CAC emissions. A third effect is a neutral response, when very little change in CACs occurs in response to GHG-reducing actions. Antagonisms are of particular concern when considering the effect of climate policies on local air pollution in densely populated urban areas. Understanding the nature of antagonistic increases in CAC emissions can help policy makers explore and design strategies to convert these trade-offs between GHG targeted policies and CAC emissions into synergies.

Three different shadow prices ($10, $30, and $50 / tonne GHG) were evaluated for their effect on GHG emissions, and the associated changes in CAC emissions[7]. Shadow prices are marginal cost signals that approximate the effect of a tax or permit price on emissions. The shadow prices chosen reflect the current consensus in Canada regarding potential prices of domestically traded GHG permits, one mechanism that may be used to pursue Kyoto targeted emission reductions (Government of Canada, 2002). In each shadow price simulation (or policy scenario), the effect of the shadow price is first felt in 2001. Each policy scenario is compared to a business-as-usual (BAU) scenario, and the difference in costs and emissions between the two reflects the effect of the shadow price.

The discussion below begins with a closer look at one specific region that

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[7] For a detailed representation and discussion of the costs associated with these policy scenarios please refer to “Construction and Analysis of Sectoral, Regional, and National Cost Curves of GHG Abatement in

experiences serious local air pollution problems – Ontario, and in particular the Windsor-Quebec Corridor. The overall trend in GHG and CAC emissions in the BAU and the changes that are estimated in the policy scenarios are discussed. Particular attention is paid to understanding where key synergies and antagonisms occur, and the consequence

CAC and GHG-intensity of the different fuels. For example, if demand for coal (GHG and CAC-intensive) falls and is replaced with natural gas (less GHG and CAC-intensive) both GHGs and CACs will be reduced. In contrast, if the fuel switch is from natural gas (which has a moderate level of associated GHG emissions) to biomass (a CAC-intensive fuel which is considered a GHG-neutral fuel in the model), a large increase in CACs may result.

In the following section, actions induced by GHG shadow prices in the policy simulations are separated out for illustrative purposes. However, it should be noted that due to the integrated nature of the model a policy will stimulate a number of actions in the economy, which in turn may have a positive feedback effect, stimulating further actions. For example, an action that increases the demand for electricity may increase the price of electricity which will affect the relative desirability of further actions that might involve further increases in electricity demand. Thus, the total change in emissions and costs associated with a policy alternative is a result of a series, or package, of actions and the nature and relative effect of these actions on the penetration of different technologies will determine the magnitude of change in emissions.

In summary, the total effect that a policy has on the emissions seen in a sector/region will depend on the total package of actions that occur. An increase in CACs will be seen when antagonistic actions outweigh the synergistic actions and vice-versa. Further complicating the matter is the fact that an action may cause a synergistic reduction in one or some of the CAC pollutants, but not all. The same can be said for antagonistic increases in CACs. Therefore, the synergies and antagonisms pointed out in the following sections are specific cases where the response was clear and of considerable magnitude.

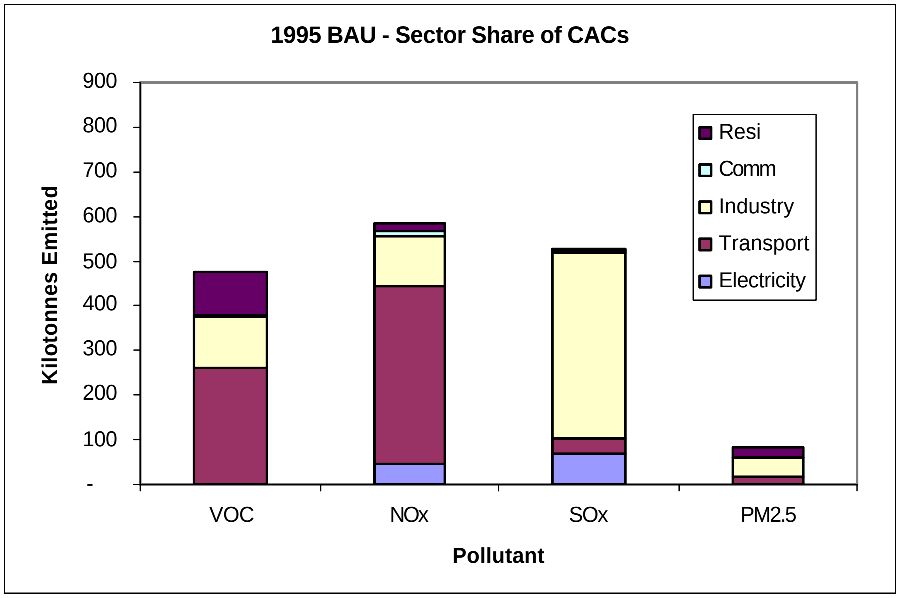
3.2 Ontario

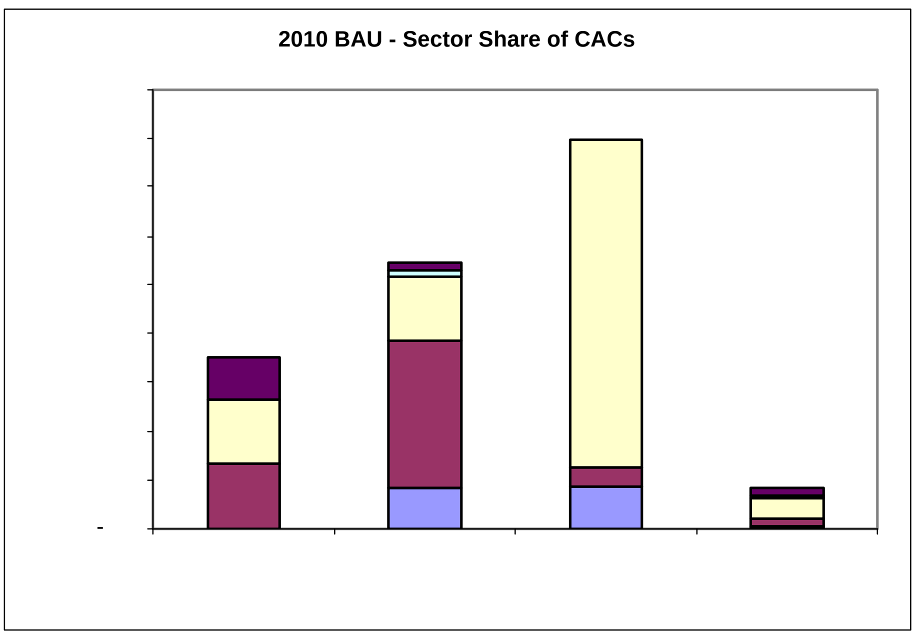
Ontario is a large province in terms of geographic size, population size and GHG emissions, and currently emits more GHGs than any other province (Jaccard et al., 2002). The densely populated southeastern region of the province suffers from poor air quality which is exacerbated in the summer months as sunlight interacts with ozone precursors (primarily NO_x and VOCs) to form photochemical smog (Environment Canada, 2002). The impacts from local air pollution in this region are severe, costing Ontarians $1 billion[8] a year as a result of increased mortality and morbidity (Environment Canada, 2002). Clearly, the effect of climate policy on CAC emissions in Ontario is an important issue.

3.2.1 The Business as Usual Case

In the business as usual or BAU case, which is based on *Canada's Emission Outlook: An Update* produced by Natural Resources Canada (AMG, 1999), the sectors

Figure 6. Sector share of BAU CAC emissions in 1995 and 2010.





By 2010, all but SO_x emissions are predicted to decline. The decline in VOC and NO_x emissions is driven by the transportation sector, and the effect of vehicle fleet turnover (newer, more efficient vehicles) and the sulphur content of fuels regulation. Reductions in $PM_{2.5}$ emissions occur mainly in the residential sector as the demand for oil in space heating falls, replaced with more natural gas and electricity technologies.

The marked increase in SO_x emissions by 2010 is predominantly due to the industrial sector, with contributions from electricity generation as well. In electricity, the increase in SO_x emissions is related to the increased prevalence of coal-fired generation. The sub-sector that contributes the most to growth in industrial SO_x emissions is metal smelting and refining, which depends on a number of SO_x-intensive process technologies.

3.2.2 Scenario Analysis

The total emission changes for Ontario in each of the policy scenarios as compared to the business-as-usual (BAU) case in 2010 are presented in table 3. Note that while GHG emission reductions are greater at higher shadow prices, CAC reductions do not follow a consistent trend.

Table 3. Emission changes in Ontario, 2010

Shadow Price	**Ontario** **Emissions Reduced in 2010 (kilotonnes)** (positive values = reduction in emissions from the BAU, negative values = increase)

In the three shadow price scenarios, the biggest changes in GHG emissions are seen in the transportation (32%), electricity (32%) and industrial (11%) sectors. In comparison, the greatest changes in CAC emissions occur in the industrial, electricity generation and residential sectors for each shadow price (see figure 7).

Note that for most sectors and pollutants, a synergistic response in CAC emissions dominates as illustrated by the number of cases where the policy scenario indicates net reductions (net reductions in a sector are indicated by the shaded area falling below the 0% line on the y-axis, and vice versa for net pollutant increases from a sector). The exception is industry, which registers an increase in $PM_{2.5}$ and VOCs in each policy scenario. In this case the magnitude of increase in the pollutants has overshadowed any synergistic actions, resulting in an overall increase in emissions. By taking a closer look at what happens in specific sectors a better understanding of the nature and magnitude of CAC synergies and antagonisms can be gained.

Electricity

The response to GHG shadow prices in the Ontario electricity generation sector is considerable and results in both GHG reductions and changes in CAC emissions. Because there is a significant amount of thermally generated power in this region (that is predicted to increase over time in the BAU scenario), demand reductions, fuel switching and efficiency improvements stimulated by the shadow prices lead to NO_x, SO_x, and VOC emission reductions with a small increase in $PM_{2.5}$ in the \$50 policy scenario (Table 4). The largest synergy exists for NO_x and SO_x emissions which are reduced by 28kt (34%) and 71.8kt (82%) respectively by 2010, in the \$50 scenario.

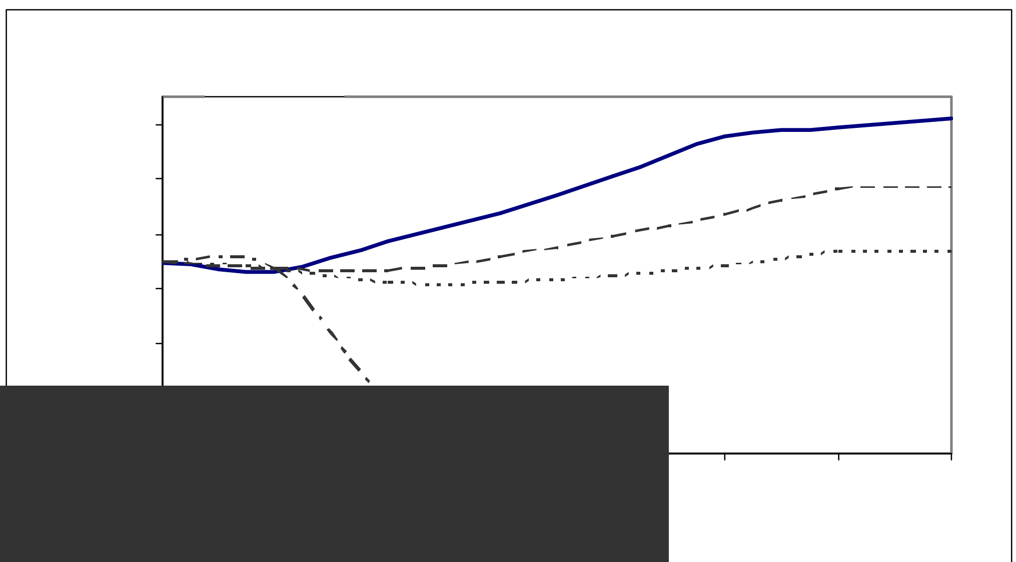
Table 4. Emission changes in Ontario electricity generation, 2010.

Shadow Price	**Emissions Reduced in 2010 (kilotonnes)** (positive values = reduction in emissions from the BAU, negative values = increase)								
(\$ / tonne CO_2e)	**GHGs**	**VOC**	**% change from BAU**	**NO_x**	**% change from BAU**	**SO_x**	**% change from BAU**	**$PM_{2.5}$**	**% change from BAU**
10	5,968.7	0.2	24%	13.8	17%	15.9	18%	0.9	22%

Overall demand for electricity falls at higher shadow prices, with the remaining demand filled by more efficient, less CAC-intensive technologies. At higher shadow prices there is less use of single cycle oil-burning technologies and conventional coal technologies, and more natural gas. In particular, fuel-efficient combined cycle natural gas technologies gain more market share at higher shadow prices. The small increase in $PM_{2.5}$ (and the trend to increased $PM_{2.5}$ emissions at higher shadow prices) is due to an increasing use of small biomass to meet base demand for electricity.

The nature of SO_x reductions over time in each of the scenarios is further illustrated in figure 8. Note that SO_x emission reductions estimated with the $50 shadow price are much greater than those under $10 and $30, plus large reductions are achieved earlier on (with approximately 59 kilotonnes of SO_x reduced by 2005 in the policy versus the BAU scenario).

Figure 8. SO_x emissions in Ontario's electricity production sector over time



[redacted]

actions that contribute to the decrease in SO_x emissions. However, the large, early drop in SO_x emissions associated with the $50 shadow price indicates that a particularly synergistic action is penetrating more strongly in this scenario. In the $50 scenario, extensive retro-fitting of single-cycle coal fired technologies to combined cycle natural gas occurs. More specifically, 83% of base, 73% of shoulder and 69% of peak load single-cycle coal technologies are retrofitted to combined cycle natural gas. Natural gas combustion produces far less SO_x emissions than coal combustion, and switching these technologies to natural gas becomes an economical choice because the marginal cost of continuing to use coal exceeds the cost of switching to natural gas. In comparison, at the $30 shadow price considerably less retro-fitting occurs (5% base load, 5% peak, 14% shoulder) and no retro-fitting occurs in the $10 scenario.

There are a number of factors that together contribute to the large degree of retrofitting seen in the results for this sector, including: an element in the retrofit algorithm used in the model, uncertainty in coal and natural gas prices, the role of trade in electricity, and the ability of firms to access sufficient capital. The effect of these factors arises because the model, as any model, is a simplification of the energy-economy system. Therefore, results must be interpreted with the potential effect of these and other related factors in mind.

As indicated above, a very large amount of retrofitting (up to 83%) is seen in the $50 scenario. This amount is larger than would be expected in CIMS given that it uses a probabilistic distribution to ensure that wholesale switching to cheaper technology options does not occur when the life-cycle cost becomes cheaper[9]. A contributing factor to the large scale retrofit stems from the algorithm that describes the retrofitting competition in CIMS. In order to represent that retrofit decisions may be made in each year of a simulation, rather than only once at the end of a five year period, the retrofit estimates produced at the end of each 5-year iteration are in essence multiplied by 5[10].

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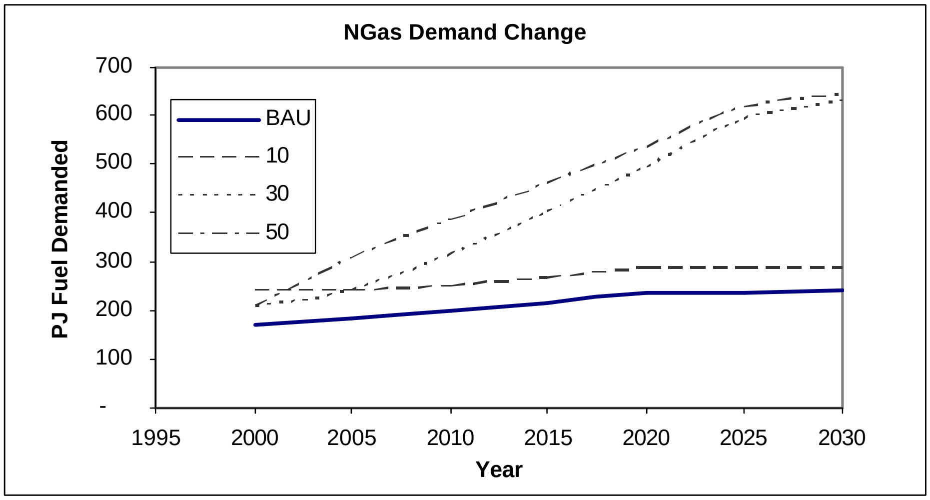
[9] This phenomenon is referred to as "penny-switching' and is often seen in linear programming models.

[10] Contact the author for further details describing the retrofit algorithm (mtisdale@sfu.ca).

Not surprisingly, the demand for coal decreases in the policy scenarios with the far greater decrease in the $30 and $50 case commensurate with the degree of retrofitting to combined cycle natural gas seen is these scenarios. This is mirrored by a greater increase in natural gas demand for the $30 and $50 scenario (see figure 9), and a less pronounced increase from the BAU for the $10 scenario.

(see Analysis and Modelling Group, 1999).

Figure 9. Change in the demand for coal and natural gas in the Ontario electricity generation sector over time



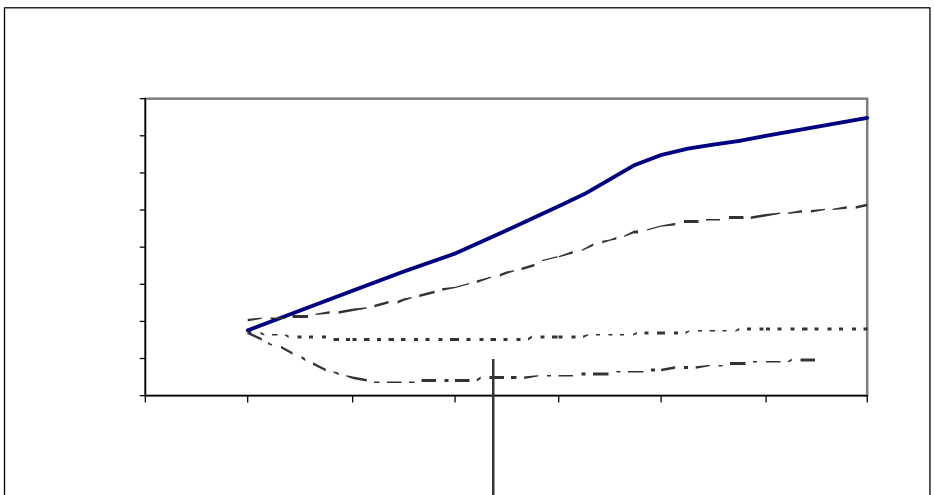


Figure 10. Change in NO_x emissions over time in Ontario electricity generation.

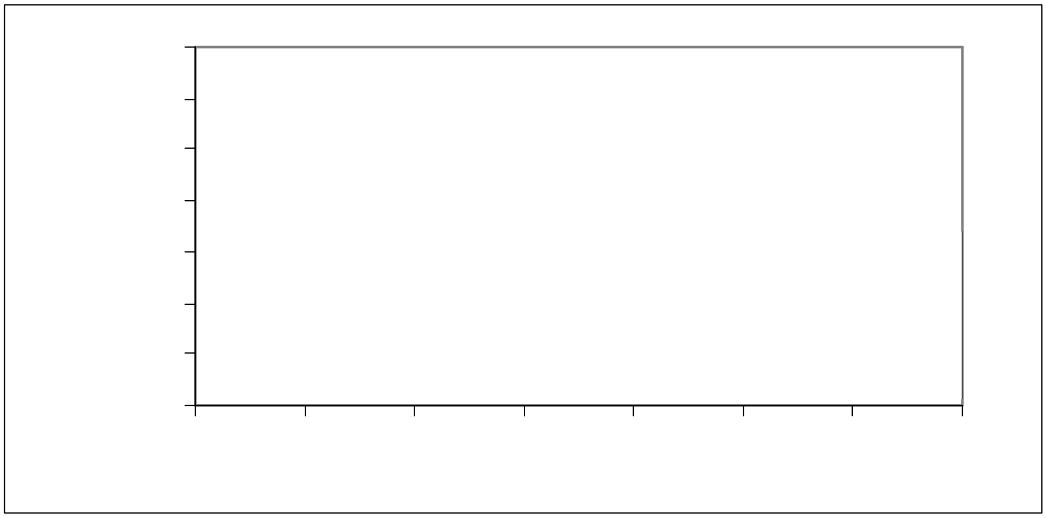


Table 6. Emission changes in the Ontario metal smelting and refining industry.

Shadow Price	Emissions Reduced in 2010 (kilotonnes) (positive values = reduction in emissions from the BAU, negative values = increase)			
($ / tonne CO_2e)	**GHGs[a]**	**SO_x**	**% change from BAU**	**% of net regional reductions**
10	43	2.3	0.5%	7%
30	69	5.8	1.2%	10%
50	79	8.0	1.7%	7%
BAU	**--**	463.2	**--**	**--**

[a] includes indirect and direct GHG emission reductions, where direct emission reductions are caused directly by an action in a sector and indirect refers to reductions associated with a reduced demand for electricity.

To understand why such small reductions are seen in such a large SO_x-producing industry, the key sources of SO_x emissions must be understood along with the actions that occur in the policy scenarios. The actions that cause the small change in SO_x emissions in the policy scenarios are fuel-switching out of coal and oil and into electricity, which is CAC-free in its end-use. For example, one such action is a switch into electric arc furnaces for copper smelting from hearth roasters and fluidized bed technologies which rely more on fossil fuels.

However, because the majority of SO_x emissions from metal smelting and refining result from process sources, fuel-switching actions have little effect on total SO_x emissions. (Figure 11 illustrates just how small the changes in SO_x emissions over time are in the policy scenarios). For example, there are two processes that can be used in metal smelting to separate metal concentrates from sulphur and oxygen compounds in the ore and remove impurities: pyro- or hydrometallurgy. As the names suggest, pyrometallurgy involves the use of heat, while hydrometallurgy uses a chemical leaching process. In hydrometallurgy, sulphur is reduced to its pure elemental form (solid state) and therefore SO_x emissions are negligible when compared to the pyrometallurgical process that uses high temperature reactions with air to concentrate the metal (Nyboer 1997), releasing SO_x in the off-gas.

Figure 11. Change in SO_x emission over time, Ontario metal smelting and refining.

scenarios (illustrated in table 7).

Table 7. Emission changes in the Ontario mining industry

Shadow Price	**Emissions Reduced in 2010 (kilotonnes)** (positive values = emissions reduction, negative values = increase)			
($ / tonne CO_2e)	**GHGs[a]**	**SO_x**	**% change from BAU**	**% of net regional reductions**
10	170.8	27.7	53%	84%
30	182.1	28.4	54%	48%
50	189.0	29.0	56%	26%
BAU	--	52.2		

[a] includes indirect and direct GHG emission reductions, where direct emission reductions are caused directly by an action in a sector and indirect refers to reductions associated with a reduced demand for electricity.

The key GHG-reducing action that causes this drop in SO_x emissions (reduced by up to 56% from the BAU in the $50 scenario) is a change in iron agglomeration technology. Specifically, the shadow prices induce a shift out of sintering and oil-fired pelletization technologies into gas-fired pelletization. Sintering and pelletization are methods of agglomeration used to remove unwanted compounds (such as sulphur) from the ore and form it into larger, cohesive units (Nyboer, 1997). Sintering requires the use of coal and the associated SO_x emissions are much higher, than from natural gas-fired agglomeration. Because the GHG emissions from sintering are higher than those from the natural gas-fired pelletization process, the GHG shadow prices favour the use of the less CAC-intensive pelletization technology. This synergy in process-based CAC emission

manufacturing' sub-sector, and in particular the wood products industry

B.C. and Ontario. In B.C. “other manufacturing”, the increase in biomass fuels accounts for most of a 21 kilotonne increase in $PM_{2.5}$ emissions by 2010 in the $50 scenario. Consequently, the magnitude of this antagonism is sufficient to result in an overall increase in $PM_{2.5}$ emissions in B.C. of 16 kilotonnes (2010).

It is important to note that the actual trend in $PM_{2.5}$ emissions in B.C. or Ontario may change when emissions that are not related to energy are considered. For example, in B.C. beehive burners are used to dispose of wood waste and produce large amounts of $PM_{2.5}$ (and other CACs) but are not included in the CIMS model because they do not demand or produce energy. The increased use of biomass as an energy source in “other manufacturing” means less wood waste is disposed of in bee-hive burners, offsetting the $PM_{2.5}$ emissions from these technologies – which should result in a net decrease in $PM_{2.5}$ emissions in B.C. (or other regions). In this case, these results are partial[13] because they sponsored ‘change-out’ programs and a national regulation is being developed by the Canadian Council of Ministers of the Environment.

Transportation

In contrast to the synergies and antagonisms described above, CAC emissions in the transportation sector show very little response to the GHG shadow prices. Transportation is the second largest source of GHGs in Ontario, and contributes the most GHG reductions in response to the shadow prices (32% of total GHG reductions in 2010). Interestingly, transportation produces significant amounts of VOCs and NO_x emissions as well, yet very small changes in CAC emissions are seen in the policy scenarios. As efficient personal vehicles (cars and trucks) and some high efficiency diesel vehicles, all of which produce less GHG emissions than less efficient, gasoline vehicles. Further of landfill gas to produce energy) was modeled exogenously. However, we can assume that landfill gas capture will offset CAC production to the extent that if offsets the demand for fossil-fuels in this sector. The demand for oil and natural gas barely change in

the policy scenarios, hence the small drop in CACs results primarily from improved insulation and some fuel switching out of oil-fired and into natural gas and electricity driven space heating technologies.

Table 11. Emission changes in Ontario commercial sector, 2010

Shadow Price	**Emissions Reduced in 2010 (kilotonnes)** Positive values = reduction, Negative values = increase								
($ / tonne CO_2e)	**GHGs[a]**	**VOCs**	**% change from BAU**	**NO_x**	**% change from BAU**	**SO_x**	**% change from BAU**	**$PM_{2.5}$**	**% change from BAU**
10	5,218	4	3.0%	415	2.9%	3	0.2%	39	2.9%

Table 12. Emission changes in Ontario Residential, 2010

Shadow Price	**Emissions Reduced in 2010 (kilotonnes)** Positive values = reduction, Negative values = increase								
($ / tonne CO_2e)	**GHGs[a]**	**VOCs**	**% change from BAU**	**NO_x**	**% change from BAU**	**SO_x**	**% change from BAU**	**$PM_{2.5}$**	**% change from BAU**
10	1915.7	11.3	13.0%	0.7	4.1%	0.2	9.1%	2.0	11.2%
30	2115.8	10.1	11.6%	1.0	5.9%	0.3	14.0%	1.9	10.6%
50	2092.5	13.0	15.0%	1.1	6.5%	0.3	14.0%	2.5	14.0%
BAU	--	86.9	--	17.0	--	2.2	--	17.9	--

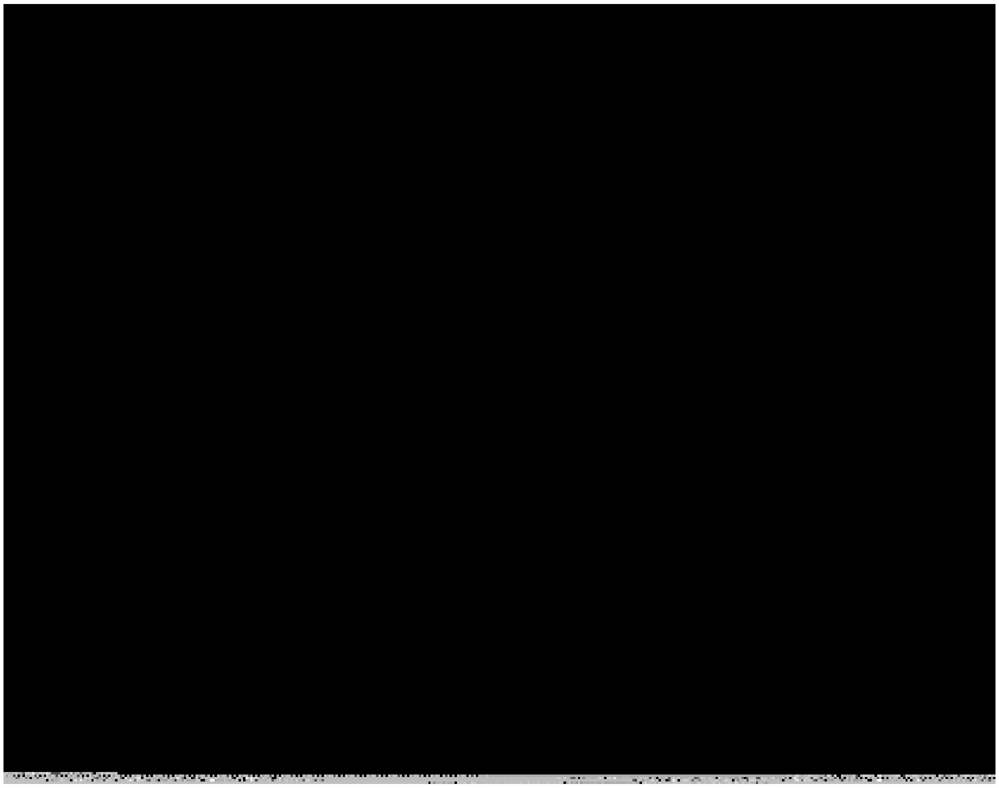
[a] includes indirect and direct GHG emission reductions, where direct emission reductions are caused directly by an action in a sector and indirect refers to reductions associated with a reduced demand for electricity

In the residential sector the specific actions and their effect on CACs are more

As indicated in section 2.1, CIMS is not a spatial model and does not produce pinpoint estimates of the geographic location of emission changes. However, the approximate location and potential impact of emission changes can be deduced by combining CIMS outputs with additional knowledge of the sector in question. The following paragraphs review the changes in CAC emissions in Ontario that were discussed above, with the goal of extracting some understanding of the location and importance of these changes on local air pollution and in particular, human health.

CIMS provides estimates of emission changes associated with GHG policies to the level of specific sectors in the different regions of Canada. While the change in CAC emissions in each of the sectors in Ontario is varied, educated assumptions can be made regarding the proximity of different sectors to urban centres, and hence the potential for emission changes to impact human health. For example, the emissions from the transportation, residential and commercial sectors are released in, or in direct proximity to, urban centres. Therefore, changes in CAC emissions from these sectors have a greater probability of affecting strained urban airsheds and larger populations. Similarly, information detailing the location of large point source emitters can be used to estimate the effect of emission changes from the associated industries. As illustrated in figure 12, the three largest, coal-fired electricity generating plants are located in the densely populated area of southwestern Ontario (Ontario Ministry of Environment, 2002).

Figure 12. Locations of coal-fired electricity generating plants in southeastern Ontario.



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Combining the information about emission reductions in the different sectors with

In comparison, the antagonistic increase in $PM_{2.5}$ emissions in the wood products industry may be less important for local air pollution and human health. These operations are generally located further from urban centers and closer to pulp and paper mills in the northern and western regions of the province (Ontario Ministry of Natural Resources, 1999).

Emission Changes in the Windsor- Quebec Corridor

The type of sector information discussed above can also be used to estimate emission changes numerically, and for a smaller region as well. For example, the Windsor-Quebec corridor in Ontario is the most densely populated area in Canada and suffers from some of the worst local air quality in the country (figure 13).

Figure 13. Map of the Ontario portion of the Windsor-Quebec Corridor.

The dotted line represents the approximate location of the corridor region.

Following the process outlined above, knowledge about the different sectors in Ontario and the location of operations was used to estimate to what extent the estimated emission changes would impact the Ontario portion of the Windsor-Quebec corridor. The results of this estimation, which was carried out using population information (residential, commercial), vehicle-kilometer-travelled data (transportation), location and production information (industrial sub-sectors), are presented in figure 14. Note that the $10 scenario was used for this example.

Figure 14. Relative change in CAC emissions in the Ontario portion of the Windsor-Quebec Corridor (WQC) versus the rest of Ontario ($10 shadow price, 2010)[14]

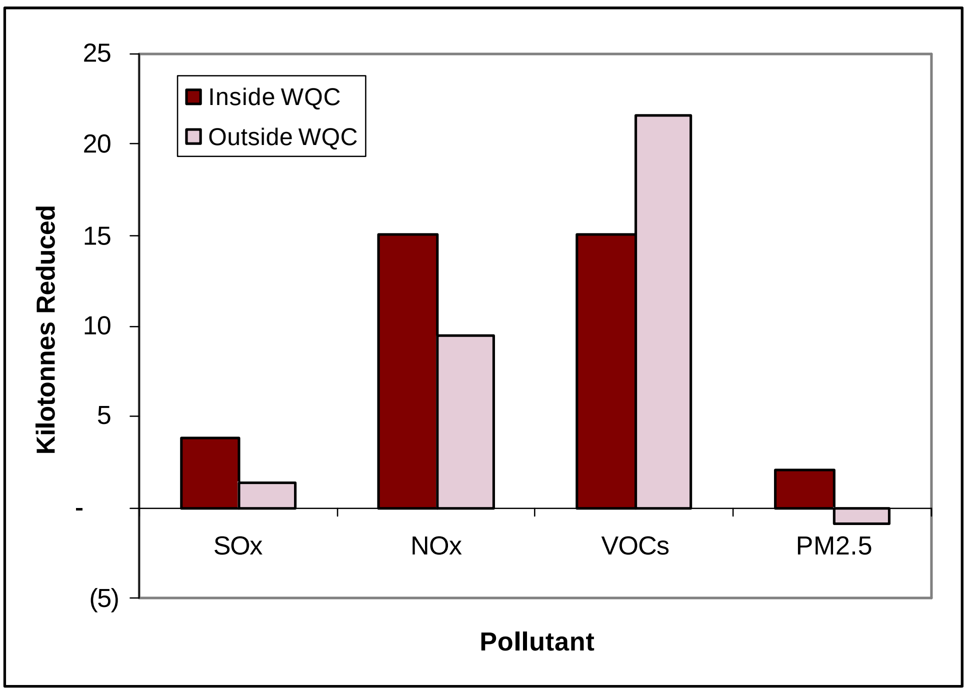


Figure 14 shows that for most pollutants the WQC benefits from a large proportion of total Ontario reductions. For example, of a total provincial reduction of approximately 25 kilotonnes of NO_x, 15 kilotonnes (61%) of these reductions occur in the

WQC. Most of the NO_x reductions in the corridor are attributed to the previously discussed changes in the electricity sector. Similarly, the majority of SO_x reductions occur inside the WQC (73%, or 3.9 of 5.3 total kilotonnes reduced), and can be attributed mainly to changes in the electricity sector.

Interestingly, $PM_{2.5}$ emissions are predicted to increase in the portion of Ontario outside of the WQC, and decrease within the WQC. The reason lies in the geographic distribution of wood product manufacturing facilities. As noted, the increased $PM_{2.5}$ emissions from the “other manufacturing” sector (and wood products in particular) result from increased use of biomass in response to GHG shadow prices. Because most wood product manufacturing facilities are located outside of the WQC, the $PM_{2.5}$ increases that occur in the shadow price scenarios do not affect the region, resulting in a net decrease of $PM_{2.5}$

comprehensive integrated assessment models or meteorological models can provide a more accurate, detailed understanding of the location and extent of the environmental impact of CAC emission changes but often have long computation times. Tools that strike a compromise between the quick, simple method described above and the more computationally intense integrated assessment models may be most useful because they simplify the process of estimating the ambient air quality changes and quicken the progression along the chain of analysis involved in estimating the ancillary costs and benefits of climate policy.

3.3 Alberta Electricity and the Role of Sequestration

The Alberta electricity sector is an interesting case study both because it is GHG-intensive, and because it has cost-effective options to reduce these emissions. The GHG emissions are high from this sub-sector because electricity production relies on fossil fuels

Table 13. Emission reductions in Alberta electricity, 2010

Shadow Price	Emissions Reduced in 2010 (kilotonnes) Positive values = reduction, Negative values = increase								
(\$ / tonne CO_2e)	**GHGs**	**VOCs**	**% change from BAU**	**NO_x**	**% change from BAU**	**SO_x**	**% change from BAU**	**$PM_{2.5}$**	**% change from BAU**
10	29,307.9	(0.4)	26.7%	44.4	40.0%	122.9	83.3%	5.2	64.2%
30	35,340.1	(0.4)	26.7%	51.2	46.1%	134.7	91.3%	5.9	72.8%
50	37,774.0	(0.4)	26.7%	46.0	41.1%	128.7	87.2%	5.6	69.1%
BAU	--	1.5	--	111.1	--	147.5	---	8.1	---

The key action contributing to GHG reductions is improved efficiency in thermal-based generation of electricity. Because much of the electricity is generated through coal combustion, considerable reductions in NO_x, SO_x, and $PM_{2.5}$ can also be attributed to this action.

One might assume that, like GHGs, the CAC reductions should increase under increasing shadow prices in this sector. However, the emission reductions of NO_x, SO_x, and $PM_{2.5}$ are highest in the \$30 scenario. The reason for this counter-intuitive trend stems from the representation of carbon sequestration technologies in the model. The geology in Alberta makes deep aquifer sequestration of CO_2 - a process that involves stripping the carbon dioxide out of the emissions stream and burying it in a deep water aquifer - a possibility in this region (Reeve, 2000; MJKA, 2002). In the shadow price simulations, once the GHG-charge reaches \$50/tonne sequestration becomes an economically viable technology because it allows coal-based generation plants to continue operating by burying their CO_2 rather than further improving efficiency, fuel switching, or paying the shadow price. Hence, under the \$50 scenario CAC emissions are slightly higher because of the role that sequestration plays in freeing up fuel choices and the consequences for emission production.

The potential for decision-makers to continue to rely on ‘dirtier’ fuels, such as coal, when sequestration technologies represent a feasible option represents an important antagonism between GHG policies and CAC emissions. However, there are two important caveats to consider when interpreting the significance of this antagonism for CAC emissions and local air pollution. First, it is possible to sequester SO_x (and potentially more of the waste stream) along with the CO_2; however, not enough is known about how this mixture of gases may react in the reservoir (Reeve, 2000). The model does not currently include this ‘enhanced’ sequestration as a technology option. Realistically, sequestering SO_x could be an important option for emitters who need to control their CAC emissions because of some other policy specifically focused on CACs. Nevertheless, the potential for, and nature of, sequestration technologies is an important consideration when evaluating the ancillary effects of climate policy alternatives.

3.4 Uncertainty

Uncertainty is too often ignored in policy analysis. However, if modelling tools like CIMS are to be useful to decision makers they must provide answers to relevant policy questions, along with an understanding of the nature and effect of the assumptions that were made in reaching these conclusions. Most variables in the CIMS model have some degree of uncertainty associated with them. Therefore, it is important to understand the extent of uncertainty as well as the effect that this uncertainty has on the functioning of, and the results produced by, the model. Explicitly representing the uncertainty in model parameters more clearly illustrates that the model results are not deterministic but indicate a median point in a range of possible outcomes. Furthermore, probing the degree of uncertainty in parameters and understanding the effect of this uncertainty on the models estimates of emission changes can help to target future model development.

and market surveys were conducted to establish the value of each parameter[16], and to associate different values for different sectors and technologies. Nevertheless, because

The ‘true’ value of *r* is something that we do not know for certain, despite extensive research to estimate the most representative value. Further complicating the choice of a value of *r* is the fact that discounting the future is a controversial topic. Different values of *r* could have a significant effect on our estimated change in CAC emissions. Recall from section 2.1.3 that the discount rate establishes the time preference associated with investments. All else remaining equal, an increase in the discount rate favours technologies with lower upfront capital costs over those with higher upfront costs and lower operating costs. For example, a more efficient boiler with high capital costs and low operating costs due to decreased energy consumption will have a harder time gaining market share if the discount rate is high. Thus a lower discount rate could result in faster, and greater decreases in CAC emissions as new, green technologies are adopted more quickly over time.

A lower value for *i* associated with a less energy-intensive technology, or a technology that relies on a renewable energy source, will have a similar effect on estimated changes in CAC emissions over time. If the technology has low intangible costs, (meaning that it is seen as a perfect alternative source of service in the eyes of the consumer) and comparative financial costs to other alternatives, it will more easily gain market share. If the technology has lower CAC emissions, the gain in market share will result in decreased CAC emissions as less efficient, fossil-fuel based technologies fall out of favour with consumers. In comparison, a lower variance parameter indicates that differences in life-cycle costs will bear more strongly on the ability of a technology to gain market share, creating more inertia for the penetration of high cost, potentially less polluting alternatives.

While the qualitative description of different possible values of the *i*, *v*, and *r* parameters above is simple, in reality the manifestation of uncertainty and the effect on model results is more complex. Ideally a full quantitative analysis of the uncertainty in each of the parameters, and in combination (e.g. high *r*, low *v*) should be undertaken to determine the sensitivity of the model results to changes in each parameter.

develop over time. Increased future regulation of CAC emissions would mean that the BAU estimates of CACs would be lower, or may even fall over time. If we assume that future CAC policies are likely, then the CAC emission changes estimated in CIMS are likely too large.

Finally, the BAU scenario does not include the potential for the development of new, effective CAC abatement technologies over time. If these technology options were included in the baseline the related CAC emissions would either fall over time or grow at a slower rate.

3.4.3 Emission Factor Uncertainty

As indicated in section 2.5, there is a great deal of uncertainty in the emission factors used to estimate CAC emission changes. Because the emission factors in this analysis represent an average relationship, the true, observed emission rates will vary with slight differences in operating conditions and other factors that are not captured in the model. This uncertainty is magnified by the fact that U.S. emission factors were used to estimate the CAC emissions from the majority of the technologies in the model, which may or not be representative of actual Canadian emissions. Furthermore, a degree of uncertainty is included in the U.S. data, and varies with the method used to develop the specific emission factor (as described in table 2, section 2.4).

The U.S. EPA qualitative rankings of emission factor uncertainty have been included in the data used in this analysis. The rankings can help inform the uncertainty analysis by indicating the relative range of possible values for the emission factor used, ranging from narrow (rankings A, B) to wide (rankings D, E) as well as the potential for the value to be biased (higher bias in D, E rankings). Factors that may cause variation in an emission factor include: the quality of the feed material or fuel, the presence of abatement technologies, the age and maintenance level of the process or combustion technology and its operating characteristics (e.g. temperature).

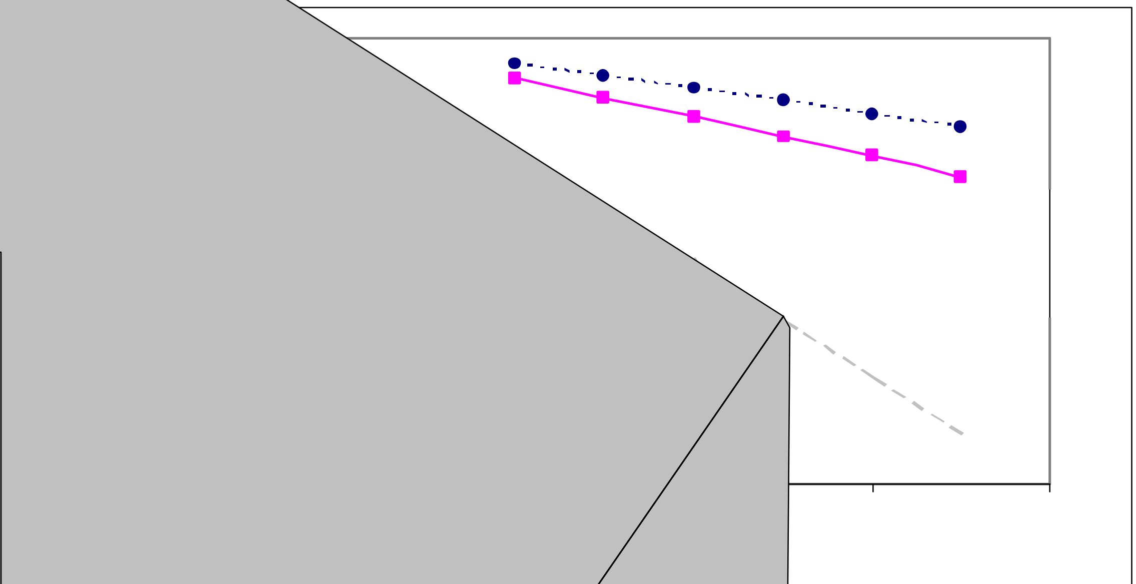
We are most interested in areas where uncertainty might greatly impact our emission predictions (change the magnitude of synergies or antagonisms), or change the relative desirability of the policy alternatives in terms of the level of CAC emission changes associated with the different shadow prices. Therefore, I have conducted sensitivity analyses on key emission factors that relate to synergistic changes in Ontario electricity SO_x emissions, and the antagonism resulting from biomass use in the "other manufacturing" sector.

Uncertainty in Ontario Electricity SO_x Emissions

Recall from the discussion of emission reductions in the Ontario electricity sector (section 3.2.2) that GHG reducing policies produced SO_x emission reductions ranging from 18 to 82% of BAU levels, and the main driver of these reductions was retrofitting existing coal technologies to combined-cycle natural gas. The magnitude of this synergy could be affected by uncertainty in the emission factor used to describe the SO_x emissions from these two fuels. Emissions of SO_x associated with natural gas combustion are so small that they are negligible (natural gas contains barely any sulphur), therefore the sensitivity analysis focuses on the SO_x EF associated with sub-bituminous coal combustion.

To test the sensitivity of SO_x emission projections to uncertainty in the coal combustion EF, the EF was varied by increments of 0.05 kg/GJ around the mean value used in the model. This variation in could illustrate the effect of higher or lower sulphur content in the coal, the presence of an abatement technology, or differences in the level of maintenance. Figure 15 presents the results of the sensitivity analysis.

nsitivity of SO_x emission estimates to uncertainty in the sub-bituminous coal combustion factor.



scenario are relatively insensitive to variation in the emission factor, while the $50 scenario, as evidenced by the steep slope of the line, is quite sensitive. Thus, uncertainty in the SO_x emission factor for sub-bituminous coal combustion has a greater effect on emission changes estimated in the $50 scenario. The effect of emission factor uncertainty is further illustrated in table 16. A very small, 0.1 kg/GJ change in the emission factor results in a large change (~18 kilotonnes, or 0.03% change) in estimated SO_x emission reductions in the $50 scenario and a smaller change (2 and 6 kt respectively, a 0.01% change) in the $10 and $30 scenario.

Table 16. Estimated emission reductions associated with different values of the SO_x emission factor for sub-bituminous coal combustion in Ontario electricity generation, 2010

	Emissions Reduced (kilotonnes)		
Scenario	**Low EF**	**Average EF**	**High EF**
	(0.3 kg/GJ)	(0.4 kg/GJ)	(0.5 kg/GJ)

0.105 Tc -ik 0 -18sSO (0.6 kg/GJ) 4 1 8 0 48 re 1 0.4High EF

Fuel Prices

Another key uncertainty that could affect the synergy between GHG actions and SO_x reductions in the Ontario electricity generation sector is the relative prices of natural gas and coal. The price of natural gas has been highly variable in the last few years, while in comparison the price of coal has been more stable. If natural gas prices were to fluctuate beyond what is represented in the model, this could greatly impact the degree of retrofitting of single cycle coal to combined cycle natural gas. As the price of natural gas increases (all else remaining equal) the penetration of natural gas technologies and the degree of retrofitting should fall –decreasing the magnitude of SO_x reductions associated with GHG-reduction policies. The effect of these fluctuating prices will indicate to decision-makers just how important fuel prices will be on the predicted co-benefits of climate policy.

Similarly, uncertainty regarding the price of coal as it pertains to electricity producers, may affect the CAC estimates in CIMS. The coal price included in CIMS in recent years was based on estimated export coal prices which are fairly high at approximately $230 per GJ. However, many electricity producers have easy access to coal in close proximity to their operations, meaning the price of coal for these producers is actually much lower, and is closer to $0.75 a GJ[18]. This is particularly true for Alberta and Saskatchewan, and less so for Ontario. Overestimating the price of coal (relative to natural gas) would have a similar effect on CAC emission estimates as described above.

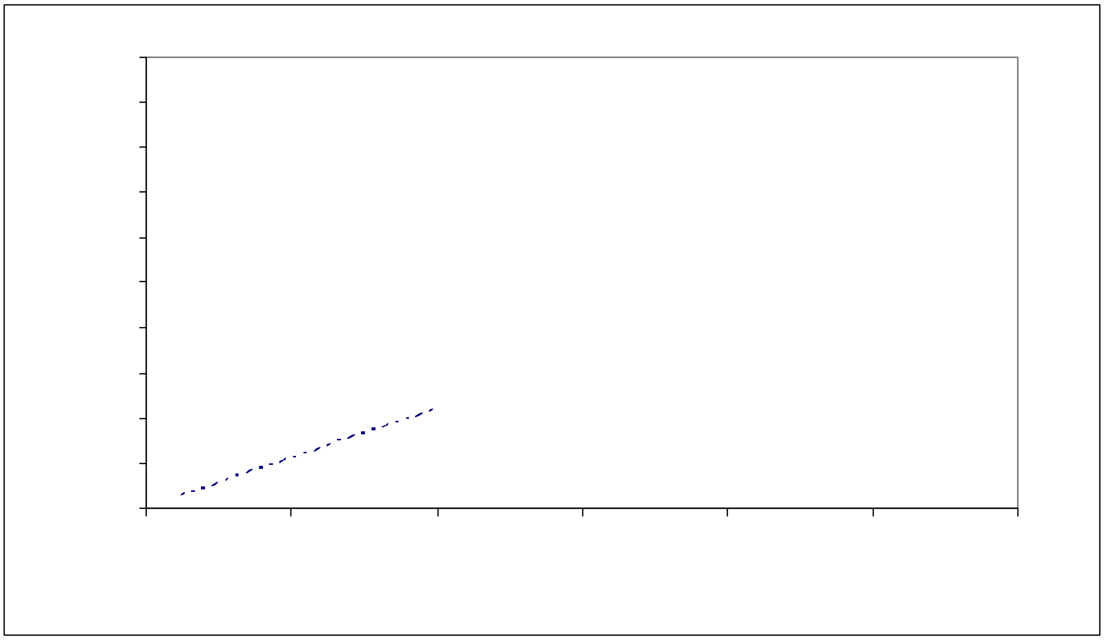
Uncertainty in Ontario Other Manufacturing $PM_{2.5}$ Emission Factor

The antagonistic increase in $PM_{2.5}$ emissions from increased wood combustion in the “other manufacturing” sector of Ontario may also be affected by uncertainty in the related emission factor. A range of different values for the $PM_{2.5}$ emission factor was

[18] The over estimation of coal prices in the electricity sector has been addressed and corrected in the newest version of the CIMS model; however, for this research project the higher coal prices were used.

tested for the resulting change in estimated emissions in the “other manufacturing” sector. Figure 16 and table 17 illustrate the results of the sensitivity analysis.

Figure 16. Sensitivity of the estimated change in $PM_{2.5}$ emissions to uncertainty in the emission factor describing $PM_{2.5}$ emissions from wood combustion



sensitivity are apparent. First, the slope of all three scenario relationships are moderate, meaning that small changes in the emission factor value will translate into moderate variation in emission changes for each scenario. Second, the slopes of the $30 and $50 line are similar, and steeper than that of the $10 line indicating that uncertainty in the emission factor will have a somewhat greater effect on the emissions predicted in the $30 and $50 scenarios than the $10 scenario.

Table 17. Estimated emission reductions of $PM_{2.5}$ associated with different values for the wood combustion emission factor in Ontario "other manufacturing", 2010

	Emissions Reduced (kilotonnes) (negative values indicate an increase)		
Scenario	**Low** (1.4 kg/GJ)	**Mean** (2.43 kg/GJ)	**High** (2.5 kg/GJ)
10	(1.5)	(2.6)	(2.7)
30	(2.4)	(4.0)	(4.3)
50	(2.3)	(3.9)	(4.1)

As indicated in figure 16 and table 17, uncertainty has a greater effect on predicted emission reductions in the $30 and $50 scenarios, where a small change (+/- 0.1 kg/GJ) results in a difference of approximately 0.3 kilotonnes (or an 8% change) of estimated PM to the model. Table 18 describes the relationship between the degree of uncertainty in a parameter, the significance of the uncertainty on the predicted outcome of the model, and the consequent importance of research to reduce the uncertainty. As discussed above, the emission factor for SO_x production from sub-bituminous coal combustion had a small degree of uncertainty (A-rating) and a small effect on resulting estimates of SO_x emissions from Ontario electricity generation. Therefore, further development of this SO_x emission factor would fall in the upper left quadrant of table 18 – indicating that further development of the emission factor to decrease the uncertainty is of a low research concern. In comparison, the $PM_{2.5}$ EF for wood combustion in “other manufacturing” was highly uncertain (E-rating) and had a greater impact on estimated emission changes

Conclusion

4.1 Summary

Anthropogenic production of greenhouse gases (GHGs) and criteria air contaminants (CACs) has contributed to climate change and local air pollution - two serious threats to sustainability in Canada. The production of both GHGs and CACs is linked strongly to economic activities and in particular to fossil fuel combustion. Consequently, policies that target GHG reduction, such as those developed to pursue the

4.2 General Lessons from the Analysis

While the analysis of GHG shadow prices ($10, $30 and $50 / tonne GHG) focused on the CAC changes in Ontario, the model includes all regions and sectors of the Canadian economy. The lessons learned from the analysis of Ontario can be summarized into a general understanding of the potential ramifications of climate policy on CAC emissions.

Synergies and Antagonisms

The analysis showed that, contrary to a common assumption in the literature, climate policy does not always result in CAC emission reductions. Some actions induced by the GHG shadow prices, such as energy efficiency improvements and demand reductions, will cause CAC reductions. The response associated with other actions (fuel-switching, process changes) is less clear, and can be antagonistic. Table 19 summarizes the types of actions that can result in response to climate policies, and the corresponding effect on CAC emissions.

Table 19. Summary of the effect of GHG-reducing actions on CAC emissions

Response to Climate Policy	General Effect on CAC Emissions

In sectors that produce large amounts of process-based CAC emissions, the change in CACs associated with GHG-reducing actions is less clear. The switch from sintering to pelletization in the Ontario mining industry is an example where a process change resulted in GHG reductions and considerable SO_x reductions as well. In contrast, if the CAC-producing process is not also associated with net GHG production, a climate policy will have little or no effect on the market share of the process technology, and will not contribute to a change in CAC emissions from the sector of focus. This phenomenon was illustrated in the Ontario metals industry. Although this sector is a big contributor to SO_x emissions in Ontario, it produced relatively small, efficiency based reductions of SO_x in response to the GHG shadow prices because the main source of SO_x emissions was a process technology with relatively low GHG emissions. Note, however, that process-based GHG-actions can just as easily produce an increase in CAC emissions if the opposite is true, and the process(es) favoured by the GHG-policy are CAC-intensive.

Some specific antagonisms pointed out in the analysis that should be carefully considered when designing climate policy include the use of biomass fuels and the potential for deep-aquifer or underground carbon sequestration. Because biomass is often considered a 'GHG-neutral' fuel over the time scale considered, climate policies can stimulate an increase in biomass-based combustion technologies which may dramatically increase CAC emissions. The use of biomass is more likely in sectors where the fuel is readily available and the technologies are accessible – such as in the wood products industry (other manufacturing) and residential heating. The potential for increased biomass combustion for energy production to offset other, more CAC-intensive technologies (like beehive burners) must also be considered.

Sequestration technologies can produce an antagonistic increase in CAC emissions, if the technologies are characterized as only removing CO_2 from the exhaust gas stream, and become an economic alternative to other abatement options. The potential for sequestration to also capture and store other pollutants (such as SO_x) may offset the need to be concerned with this antagonism.

Finally, some sectors showed a relatively neutral CAC response to the GHG shadow prices. This can occur when the baseline scenario incorporates actions and existing regulations that cause CAC reductions, as in the transportation sector.

Ultimately, the total change in CAC emissions in a sector or region will depend on the relative magnitude of synergistic and antagonistic responses to a climate policy. As summarized in table 20, the net result is simple despite the complex and differing effect of the actions taken to reduce GHGs.

Table 20. Relative magnitude of synergistic and antagonistic response in CAC emissions to GHG-reducing actions and the effect on total sector or region emissions

Relative magnitude of Synergistic/Antagonistic Actions	Cumulative Result on CAC Emissions
S > A	Decrease
S = A	Neutral
S < A	Increase

If the CAC reductions from the sum of all synergistic actions in the region/sector are greater than the sum of CAC increases from antagonistic actions, the net result will be a decrease in CAC emissions and an ancillary benefit associated with the climate policy option. However, it should be noted that the net result may be different for each CAC pollutant; therefore, the guidelines summarized in table 20 apply individually to each pollutant as well.

Regional Interpretation of Results

As discussed in section 3.2.3, the regional relevance of emissions changes can be deduced by combining an understanding of the sectors that realize CAC changes, and the proximity of the sector activity to urban centers. An increase in a CAC pollutant in a sector that is inherently urban in nature (i.e. residential, commercial, transportation) will likely impact urban air sheds more severely with greater consequences to human health.

In some cases, additional information (such as the location of coal-fired generating stations in Ontario electricity production) can be added to further understand the significance of specific actions (i.e. switch from coal to natural gas generation) and the implications for local air pollution.

CAC estimates were disaggregated to a finer geographic level by combining CIMS estimated CAC emission changes with additional information describing the population of the region, the likely geographic location of sector activities, and production levels of facilities in the Windsor-Quebec Corridor. This type of simplified approach is valuable when decision makers require a general understanding of the spatial distribution of emission changes and the potential for human and environmental impact in a timely manner. However, this approach can not replace steps 4-6 of the ancillary effects chain of evaluation (ambient air quality modelling, estimation of environmental and health effects, and valuation) when a complete and comprehensive evaluation of the full costs and benefits of a climate policy is required.

Uncertainty

To more fully explore the limitations of the CAC-CIMS modelling tool, the uncertainty in the model structure and emission factor data was explored. The model parameters that drive the evolution of technologies in the model (i.e. *i*, *v*, and *r)* and the effect of uncertainty in these variables on the estimates of CAC emission changes were briefly and qualitatively discussed.

Because this research is a first attempt at developing technology-specific CAC emission factors (EFs), the uncertainty in emission factors was explored and a process for targeted improvement in the EFs presented. The qualitative rankings of CAC emissions factors, as specified by the U.S. EPA, can be used in conjunction with sensitivity analyses to determine which EFs are most ‘uncertain’ and also have a relatively large impact on the model’s estimate of CAC emission changes. The EFs that meet these two criteria are good candidates for further research to reduce uncertainty regarding their value, because they will contribute the most to improving the overall ‘accuracy’ of the representation of

CAC emission changes in CIMS. Ultimately, considering the uncertainty in the model structure and emission factors and systematically reducing the uncertainty in key parameters will refine the model and make it a more useful tool for policy design and evaluation.

4.3 Recommendations for Future Research

As mentioned, this project represents the first time that CACs have been incorporated into a technology specific, behaviourally realistic, energy-economy simulation model for Canada. As such, there are a number of potential avenues for future research that would improve the quality of the analysis and expand the types of policy questions that can be addressed.

Quantitative analysis of uncertainty in the algorithm parameters

Although the results presented in this report are unique estimates, they are in fact uncertain and should be interpreted as a point in a possible range of emission reductions. A full, quantitative analysis of the magnitude of this uncertainty would be a useful extension as it would provide improved understanding of the range of the ancillary effects of the climate policies tested. In particular, further analysis of the effect of uncertainty in the algorithms that drive technology change should be pursued.

Use of data specific to Canada

The lack of sufficiently detailed Canadian data necessitated the use of U.S. EPA AP-42 EFs. However, some experts feel that for a number of sectors and processes the U.S. values do not accurately represent Canadian emissions. Where possible, and if justified by the sensitivity analysis described above, EFs based on measurements taken from Canadian technologies should be developed and incorporated into the model. The approach to identifying uncertain emission factors with a significant effect on the model’s CAC estimates outlined above will help target the process, and make the best use of the resources required to develop Canadian specific EFs.

Better reconciling of assumptions between CIMS and RDIS-II

As mentioned in the methodology, the underlying assumptions of the RDIS-II CAC Inventory were unavailable to compare with those included in CIMS. Consequently, the calibration of the CAC data used in the model may not be as representative or accurate as it could otherwise have been. A detailed understanding of the assumptions included in the RDIS-II database (i.e. fuel demand, sector output) could greatly refine the calibration of CIMS CAC emission estimates. Also, calibrating to the RDIS-II forecast will improve the CAC reference case over time and decrease the uncertainty in estimated emission changes over time.

Representation of CAC control technologies (extension)

Abatement technologies (e.g. scrubbers, baghouses, etc.) were not included as separate technologies in the model, and as such did not factor into estimates of technology penetration, fuel demand or costs. However, if these technologies were fully represented[20] in CIMS (including their capital and operating costs, fuel demand, and effect on emission) they would impact the competition for market share and the resulting effects of policies. As a result, a larger array of policy questions could be addressed; including a comparison of the effectiveness of CAC-focused versus GHG-focused policies (Gielen, 2002), and the analysis of multiple emission reduction strategies (MERS). Furthermore, the potential for synergies and antagonisms between the two policy targets could be more fully explored (For example, actions to reduce CACs such as the use of a scrubber, which in turn demands energy, can result in increased GHGs if this energy is supplied by fossil fuels).

[20] The range of control technologies available in the model should reflect the actual range of options available to plant operators (i.e. regulation may require the use of a specific technology, in which case there would be no competition required)

Develop an exogenous tool to Improve spatial resolution of estimates

Finally, the ability of CIMS to estimate emission changes at a fine level of spatial resolution was explored. Emission changes are estimated at the level of sectors in a

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Appendix A More about CIMS

Table 1. Regions and sectors included in the CIMS model

An X indicates that the sector/region is included in CIMS.

	British Columbia	**Alberta**	**Saskatchewan**	**Manitoba**	**Ontario**	**Quebec**	**Atlantic**
Energy Demand Sectors							
Residential	X	X	X	X	X	X	X
Commercial	X	X	X	X	X	X	X
Transportation	X	X	X	X	X	X	X
Industrial							
Metal Smelting and Refining	X	--	--	X	X	X	X
Mining	X	--	X	X	X	X	X
Iron and Steel Production	--	--	--	--	X	X	--
Chemical Product Manufacturing	X	X	--	--	X	X	--
Pulp and Paper	X	X	--	--	X	X	X
Industrial Minerals	X	X	--	--	X	X	X
Coal Mining	X	X	X	--	--	--	X
Other Manufacturing	X	X	X	X	X	X	X
Energy Supply and Conversion Sectors							
Electricity	X	X	X	X	X	X	X
Natural Gas Extraction and Transmission	X	X	X	X	X	X	X
Petroleum Refining	X	X	X	X	X	X	X

Appendix B CAC Data Used in the Model

Table 1. Fuel Combustion CAC Emission Factors for Selected Fuels

	Natural Gas			
	Sector			
Pollutant	**Elec**	**Comm**	**Res**	**Ind**
CO	0.17	0.04	0.02	0.04
VOCs	0.0009	0.002	0.002	0.002
NO_x	0.14	0.04	0.04	0.08
SO_x	0.000	0.000	0.000	0.000
PM_{Total}	0.003	0.003	0.003	0.003
PM_{10}	0.003	0.003	0.003	0.003
$PM_{2.5}$	0.003	0.003	0.003	0.003

	Heavy Fuel Oil			
	Sector			
Pollutant	**Elec**	**Comm**	**Res**	**Ind**
CO	0.014	0.014	0.014	0.014
VOCs	0.036	0.003	0.003	0.001
NO_x	0.13	0.157	0.157	0.157
SO_x	0.8	0.8	0.8	0.8
PM_{Total}	0.93	0.06	0.06	0.06
PM_{10}	0.66	0.04	0.04	0.05
$PM_{2.5}$	0.48	0.01	0.01	0.03

	Light Fuel Oil			
	Sector			
Pollutant	**Elec**	**Comm**	**Res**	**Ind**
CO	0.014	0.015	0.015	0.014
VOCs	0.002	0.002	0.002	0.001
NO_x	0.134	0.057	0.056	0.057
SO_x	0.089	0.083	0.083	0.083
PM_{Total}	0.020	0.006	0.006	0.006
PM_{10}	0.014	0.003	0.003	0.003

	Wood			
	Sector			
Pollutant	**Elec**	**Comm**	**Res**	**Ind**
CO	0.64	5.82	5.71	5.82
VOCs	0.0006	0.094	1.312	0.094
NO_x	0.07	0.64	0.07	0.64
SO_x	0.004	0.032	0.032	0.032
PM_{Total}	0.34	3.08	3.08	3.08
PM_{10}	0.33	2.77	2.77	2.77

PM_{Total}	0.833	0.305	0.305	0.305	**PM_{Total}**	0.654	0.654	0.654	0.654
PM_{10}	0.599	0.217	0.217	0.217	**PM_{10}**	0.470	0.470	0.470	0.470
$PM_{2.5}$	0.156	0.054	0.054	0.054	**$PM_{2.5}$**	0.123	0.123	0.123	0.123

Table 2. National Average Sulphur and Ash Content of Fuels

Liquid Fuels	**Average Sulphur Content (%wt.)**	**Average Ash Content (%wt.)**
Aviation Turbo Fuel	.055	—
Motor/Aviation Gasoline	.032	—
Kerosene/Stove oil	.045	—
Low-Sulphur Diesel Fuel	.032	—
Diesel Fuel	.230	—
Light Fuel Oil	.203	—
Heavy Fuel Oil	1.771	—
Plant Consumption	1.668	—

Coal	**Average Sulphur Content (%wt.)**	**Average Ash Content (%wt.)**
Western (B.C., Alberta, Saskatchewan and Manitoba)		
Anthracite	0.64	16.4
Lignite	0.41	15.7

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Appendix C CAC Calibration tables for Ontario

For calibration tables of other regions / sectors contact the author.

Residential					
INITIAL	**RDIS II**	**Unit**	**CIMS**	**Unit**	**Difference**
CO	211,305	t	130,428	t	38%
VOCS	115,024	t	29,355	t	74%
NOX	18,232	t	17,396	t	5%

Electricity					
INITIAL	**RDIS II**	**Unit**	**CIMS**	**Unit**	**Difference**

Iron and Steel Production					
INITIAL	**RDIS II**	**Unit**	**CIMS**	**Unit**	**Difference**
CO	720,434	t	604,598	t	16%
VOCS	27,710	t	12,108	t	56%
NO_x	20,965	t	24,532	t	-17%
PM Tot	16,264	t	67,608	t	-316%
PM<2.5	5,267	t	15,660	t	-197%
PM<10	7,998	t	50,551	t	-532%
SO_x	47,696	t	44,320	t	7%

SCALED CACs	RDIS II	Unit	CIMS	Unit	Difference
CO	720,434	t			

Industrial Minerals					
INITIAL	**RDIS II**	**Unit**	**CIMS**	**Unit**	**Difference**
CO	4,380	t	3,746	t	14%
VOCS	365	t	368	t	-1%
NO_x	11,238	t	17,045	t	-52%
PM Tot	8,556	t	16,012	t	-87%
PM<2.5	2,216	t	1,967	t	11%
PM<10	4,073	t	6,334	t	-56%
SO_x	20,840	t	12,964	t	38%

Petroleum Refining

INITIAL	RDIS II	Unit	CIMS	Unit	Difference
CO	4,711	t	2,042	t	57%
VOCS	25,648	t	17,654	t	31%
NO_x	12,822	t	5,342	t	58%
PM Tot	2,894	t	357	t	88%
PM<2.5	1,314	t	239	t	82%
PM<10	2,176	t	306	t	86%
SO_x	61,595	t	9,405	t	85%

SCALED CACs	RDIS II	Unit	CIMS	Unit	Difference
CO	4,711	t	5,106	t	-8%
VOCS	25,648	t	26,261	t	-2%
NO_x	12,822	t	10,187	t	21%
PM Tot	2,894	t	2,328	t	20%
PM<2.5	1,314	t	1,096	t	17%
PM<10	2,176	t	1,740	t	20%
SO_x	61,595	t	52,070	t	15%