Exhausted with Emissions: Exploring the Link between Anthropogenic Emissions and Urban Pollution on Public Health

By

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Abstract

In the first of the two analyses presented in this thesis, we investigate how reducing transportation emissions and other anthropogenic ("man-made") emissions of nitrogen oxides (NO_X) affects local and national public health in two heavily populated and polluted areas: Los Angeles, CA and New York City, NY. We estimate both the Tipping Point as well as the Break-Even Point in urban areas and show that compounding benefits of NO_X control will eventually compensate disbenefits that occurred with emission abatement. We find that the Tipping Point occurs after abatement levels of 50-52% in Los Angeles and between 7-8% in New York. We find the Break-Even Point occurs at 85-86% in LA and 14-16% in NY. We examine the impact of applying nationalized vs. localized emission control policies in these regions and find that although national abatement scenarios produce the largest benefits, abatement policies targeting local areas may still reach up to 70% of these benefits in NY and 80% in LA.

In the second study, we focus more closely on the transportation sector and examine the current distribution in exhaust emissions of vehicles on an age-segregated basis for both Canada and the United States. Through the combination of these emission factors with data on the per-tonne public health impacts of NO_X and PM emissions from a previous study, we estimate the health benefits of both (1) reducing the distance travelled of vehicles in terms of $/mile, and (2) removing one average vehicle from the road in terms of $/vehicle-year (the per-vehicle health benefit [PVHB]). We characterize the distribution of PVHBs, on a per-year basis, by grouping the vehicles into five age bins within the light-

duty vehicle fleet. For example, we find that the health benefits for an average light-duty gasoline vehicle near Quebec City can range from $62/vehicle-year to $1167/vehicle-year, dependent on age. This study aims to provide more information for coordinated transportation planning and air quality management.

Dedication

I would like to dedicate this thesis to my best friend and the strongest woman I have ever known – my grandmother, Germaine Nolet. I would also like to thank Philippe Genereux and Kathleen Hudson for their endless support. Lastly, I would like to give a shout-out to the Monster Energy corporation for providing me with the energy (via NOS) to write this 185-page document.

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Table of Contents

List of Tables

List of Figures

Abbreviations

ACS	American Cancer Society
ADE	Atmospheric Diffusion Equation
AFWA	Air Force Weather Agency
APEEP	Air Pollution Emissions Experiments and Policy Analysis Model
APHENA	Air Pollution and Health: A Combined European and North American Approach project
AQM	Air Quality Model
BC	Black Carbon
BCON	Boundary Conditions
BEP	Break-Even point
BPT	Benefit per tonne
CAMx	Comprehensive Air Quality Model with Extensions
CAN	Canada
CB05	Carbon Bond V
CDC	Center for Disease Control and Prevention
CG	Calgary, AB, Canada
CMAQ	Community Multiscale Air Quality Model
CONUS	Continental United States
CPS-II	Cancer Prevention Study II
CRF	Concentration Response Function
CTM	Chemical Transport Model

EASIUR	Estimating Air pollution Social Impacts Using Regression
EC	Elemental Carbon
ECCC	Environment and Climate Change Canada
EDGAR	Emissions Database for Global Atmospheric Research
EF	Emission Factor
FAA	Federal Aviation Administration
FLS	Forecasting System Laboratory
FM	Fort McMurray, AB, Canada
FHWA	Federal Highway Administration
GBD	Global Burden of Disease
GVWR	Gross Vehicle Weight Rating
HDDBS	Diesel School Buses
HDDBT	Diesel Transit and Urban Buses
HDDV2b	Heavy-Duty Diesel Vehicles 2b
HDDV3	Heavy-Duty Diesel Vehicles 3
HDDV4	Heavy-Duty Diesel Vehicles 4
HDDV5	Heavy-Duty Diesel Vehicles 5
HDDV6	Heavy-Duty Diesel Vehicles 6
HDDV7	Heavy-Duty Diesel Vehicles 7
HDDV8a	Heavy-Duty Diesel Vehicles 8a
HDDV8b	Heavy-Duty Diesel Vehicles 8b
HDGB	Gasoline Buses (School, Transit and Urban)

HDGV2b	Heavy-Duty Gasoline Vehicles 2b
HDGV3	Heavy-Duty Gasoline Vehicles 3
HDGV4	Heavy-Duty Gasoline Vehicles 4
HDGV5	Heavy-Duty Gasoline Vehicles 5
HDGV6	Heavy-Duty Gasoline Vehicles 6
HDGV7	Heavy-Duty Gasoline Vehicles 7
HDGV8a	Heavy-Duty Gasoline Vehicles 8a
HDGV8b	Heavy-Duty Gasoline Vehicles 8b
HX	Halifax, NS, Canada
ICON	Initial Conditions
InMAP	Intervention Model for Air Pollution
JPROC	Photolysis Rate Processor
LA	Los Angeles, California
LDDT12	Light-Duty Diesel Trucks
LDDT34	Light-Duty Diesel Trucks 3 and 4
LDDV	Light-Duty Diesel Vehicles (Passenger Cars)
LDGT1	Light-Duty Gasoline Trucks 1
LDGT2	Light-Duty Gasoline Trucks 2
LDGT3	Light-Duty Gasoline Trucks 3
LDGT4	Light-Duty Gasoline Trucks 4
LDGV	Light-Duty Gasoline Vehicles (Passenger Cars)
MB	Marginal Benefit ($/tonne)

MC	Motorcycles (Gasoline)
MCIP	Meteorology-Chemistry Interface Processor
NAAQS	National Ambient Air Quality Standards
MOBILE6.2C	Mobile Source Emission Factor Model
MOVES	Motor Vehicle Emission Simulator Model
NCAR	National Center for Atmospheric Research
NEI	National Emissions Inventory
NOAA	National Oceanic and Atmospheric Administration
NOx	Nitrogen Oxides
NPRI	National Pollutant Release Inventory
NY	New York, New York
OC	Organic Carbon
OTC	Ozone Transport Commission
PM	Particulate matter
PM10	Particulate matter smaller than 10 microns
PM2.5	Particulate matter smaller than 2.5 microns
PVHB	Per-Vehicle Health Benefit
QC	Quebec City, QC, Canada
QSSA	Quasi-Steady State Assumption
RH	Relative Humidity
RVP	Reid Vapor Pressure
SCC	Source Category Code

SMOKE	Sparse Matrix Operator Kernel Emissions Model
SO_X	Sulfur oxides (SO_2, SO_4, SO_4^{2-})
SS	Steady State
TIC	Transportation in Canada
TO	Toronto, ON, Canada
US	United States
US EPA	United States Environmental Protection Agency
VMT	Vehicle-miles travelled
VN	Vancouver, BC, Canada
VOCs	Volatile organic carbon
WH	Whitehorse, YT, Canada
WRF	Weather Research Forecasting Model
YK	Yellowknife, NWT, Canada

Symbols

C_i	Mixing ratio
$\boldsymbol{u}$	Vector wind field
ρ	Density of air
K	Diffusivity constant
$\mathrm{R_i}$	Reaction rate of species i
$\mathrm{E_i}$	Emission rate of species i
J	Monetized number of non-accidental mortalities associated with a specific pollutant
ω	Spatial coordinate
λ_i	Adjoint variable at each location and time step
$\mathrm{F_i}$	The *i*th row of the transpose of the Jacobian matrix
φ_i	Adjoint forcing term
V_{SL}	Value of statistical life (country-specific)
$M_{0,\omega}$	The non-accidental mortality rate in a grid cell
P_ω	Population within a specified grid cell
β	Concentration response factor (based on risk)
$\bar{t}$	Number of hours in the exposure metric
C_ω	Max pollutant concentration at each location (dependent on the desired exposure metric)
n	Number of days in each simulation period

1 Introduction

Take a deep breath. Every minute we breathe we inhale approximately 7 litres of air [1], but how much do you know about the air you are breathing? It is abundant, free of charge, invisible to the naked eye, and yet we cannot live without it for longer than a matter of minutes.

In terms of health, the composition of the air we breathe is almost as important as access to air itself and is directly affected by the emissions we introduce into the environment. In fact, a global burden of disease (GBD) study from 2015, estimated that 4.2 million premature deaths occur every year word-wide due to exposure to particulate matter alone, as well as 234,000 deaths attributed to ozone exposure [2]. From a recent Health Canada report in 2017, it is stated that 14,400 deaths in North America every year are due to air pollution [3]. The GBD only considers deaths due to heart disease and stroke, lung cancer, chronic lung disease, and respiratory infections for deaths related to $PM_{2.5}$ and considers chronic lung disease for ozone (O_3).

In addition to these well-established health effects from exposure to these pollutants, there is also an emerging risk of cancer that is not as well understood. Recent epidemiological studies show a potential link between gasoline exhaust and multiple negative health outcomes, including lung cancer [4-13], bladder cancer [4, 6, 12, 14-20], and lymphohematopoietic cancer [4, 6, 12, 14, 21-26]. In addition to these other cancer

types have been listed, including, gastric, pancreatic, liver, kidney, endocrine gland, esophageal, reproductive, and melanoma [4, 6, 12, 14, 23, 27-31].

To reduce the overall public health burden of exposed populations, urban pollution levels can be mitigated with the implementation of more stringent emission control strategies, which has been shown to be effective. In fact, in the United States, aggressive regulatory efforts have reduced nitrogen oxide (NO_X) emissions from a number of emission sources from 1990 to 2010 [32]. These NO_X controls have resulted in a decrease of not only NO_X concentrations, but also served as an effective strategy of reducing nationwide O_3 concentrations in most areas.

At a regional level; however, local concentrations of O_3 have increased in several urban and suburban areas [33, 34]. This increase in O_3 with reductions in NO_X is due to the effect of NO_X titration in a heavily polluted environment. The chemistry behind this is presented in Chapter 2 in detail however, consideration of this effect is key to finding effective and sustained control policies for areas that are NO_X-rich.

In this thesis, the focus is placed on the impacts of urban pollution on public health, specifically nitrogen oxide and particulate matter emissions and mitigation strategies that may be used to reduce exposure to high levels of NO_2, O_3 and $PM_{2.5}$. Here, an adjoint sensitivity analysis is used to render information on the effect of various emission reduction scenarios on domain-wide mortality, as sensitivity analyses provide an effective

approach to provide information on the impacts of emission sources on receptors. Utilizing health as a policy endpoint in this research helps provide information for policy makers, including not only the effect of emission reductions on the population exposed, but also on where the most influential policies should be introduced, including accounting for the effectiveness of either regional or national policies.

As it is well-documented that the largest fraction of NO_X emissions in North America is emitted from the transportation sector, we also quantify the distribution of damage to public health by vehicles on an age-segregated basis. With advances in vehicle technology and the implementation of more stringent emission standards, the magnitude of emissions from the tailpipe of vehicles has changed dramatically since the 1990's. For example, the exhaust emission standard for light-duty gasoline cars has been improved from the Tier 1 standard of 0.4 g/mi to the current (Tier 2) standard of 0.07 g/mi set by the US EPA. The US EPA has proposed future emission standards lower than this, with the implementation of Tier 3 emission regulations. As Canada follows emission standards set by the United States, policies across North America aimed at reducing the emissions from transportation would have different efficiencies if they narrow the target to the oldest vehicles on the road today.

1.1 Objectives

With links established between epidemiology, air quality modeling, policy information, and emission factor tools, this thesis attempts to answer the following questions:

1. How can local (versus wide-spread) emission reductions in heavily populated and polluted areas prove to be favourable policy options? How do either of these emission reduction scenarios affect local public health (vs traditional national health estimates)?
2. At what percentage of abatement in heavily populated and polluted areas do disbenefits become positive (i.e. the Tipping Point)? At what point are cumulative disbenefits compensated by accrued benefits along the abatement pathway (i.e. the Break-Even Point)?
3. How can air quality models be used in conjunction with vehicle emission software to determine the damage to public health by individual vehicles?
4. How does the vehicle age and/or type of vehicle impact the per-vehicle health damage estimates? Can the findings be used to support policy incentives for reducing pollution in heavily populated areas?

1.2 Thesis Structure

This thesis is centered around two manuscripts that address the impact of urban pollution on public health in two unique ways. Prior to these investigations, Chapter 2 provides sufficient background information and context for this work. Following this, the

models and methodologies employed in both manuscripts are discussed in Chapter 3. The remaining chapters consist of two investigations that link urban pollution and exposure to domain-wide health. Chapter 6 concludes the main ideas of the research presented in Chapters 4 and 5 and discusses the current limitations and future work that may be employed to build upon these ideas.

Chapter 4 aims to characterize the non-linearity of anthropogenic NO_X emissions in two heavily polluted urban areas in the United States: Los Angeles, California (LA) and New York, New York (NY). A multi-step analysis of health benefits is performed with three desired outcomes. The first is to identify the Tipping Point (TP), or the abatement percentage that corresponds to the point along the abatement curve where disbenefits become positive (i.e. where the region shifts from NO_X-inhibited (high concentration of NO_X) to NO_X-limited (low concentration of NO_X)) for both LA and NY. Next, this research aims to identify the Break-Even Point, or the abatement percentage that corresponds to the point along the abatement curve where the cumulative health benefits (accrued from progressive emission controls) compensate disbenefits in both LA and NY. At the conclusion of this paper, a comparison of the results of local and national emission reduction policy scenarios are compared (with their respective regional and national societal health damage estimates) to demonstrate how emission control strategies can be effectively applied in heavily polluted areas.

The results presented in Chapter 5 emphasize the capabilities of the adjoint model, including how it can be used in conjunction with vehicle emissions modelling software to estimate the contribution to the overall societal public health burden from on-road vehicles in Canada and the United States, on an age-segregated basis. As the transportation sector is the dominant source responsible for NO_X emissions and remains a significant source of primary $PM_{2.5}$ emissions, the ideas presented in Chapter 5 may serve as a seamless extension of the results and conclusions shown in the previous chapter. The results of Chapter 5 may be used to support policies created to reduce emissions in urban cores or NO_X-rich areas. For example, in order to meet the TP in NY and LA, new policies targeting older vehicles may be enforced – reducing the societal health burden overall.

 1 250 000 People.

The number of deaths from traffic related injuries worldwide every single year.

What is even more salient is

 4 200 000 People.

The number of people killed due to ambient particulate matter (from all-sources) worldwide every single year.

 =1 million deaths

2 Background & Current Literature

In this section, an overview of the chemistry of NO_X and O_3 in the Troposphere is presented, followed by a summary of the most common health outcomes that develop in the population due to exposure to these pollutants. Following this, effect estimates in the literature are defined and listed, along with the application of these estimates and the resultant marginal damage estimates in the current literature.

2.1 Secondary Pollutants in the Troposphere

Direct emissions of nitrogen oxides ("NO_X") include both nitric oxide (NO) and nitrogen dioxide (NO_2) species. The presence of reactive oxygen radicals helps in determining whether the NO_X emission will increase ground-level ozone concentrations and whether NO will react to form NO_2. In the Troposphere, ozone (O_3) is a secondary pollutant, i.e., it is not emitted directly; instead, complex photochemical interactions between NO_X molecules and radicals generated from volatile organic compounds (VOCs) result in the formation of ground-level ozone. In contained, high-temperature combustion, such as gasoline or diesel engines, approximately 10 times more NO is emitted than NO_2 [35].

Under normal conditions, the following reactions occur (in the day-time) when the NO_x cycle in equations (2-1) to (2-3) exists in an equilibrium state, often referred to as photo stationary state [36]:

$$NO_2 \xrightarrow{hv} NO + O \qquad \text{Eq. 2-1}$$

$$O + O_2 + M \rightarrow O_3 + M \qquad \text{Eq. 2-2}$$

$$NO + O_3 \rightarrow NO_2 + O_2 \qquad \text{Eq. 2-3}$$

$$RH + OH + O_2 \rightarrow RO_2 + H_2O \qquad \text{Eq. 2-4}$$

$$RO_2 + NO \rightarrow NO_2 + RO \qquad \text{Eq. 2-5}$$

$$RO + O_2 \rightarrow HO_2 + RCHO \qquad \text{Eq. 2-6}$$

$$NO + HO_2 \rightarrow OH + NO_2 \qquad \text{Eq. 2-7}$$

where RH represents, as an example, a hydrocarbon with a radical R as its base. Equations (2-1) to (2-3) are in equilibrium, with zero net ozone production. Applying the quasi steady-state assumption (QSSA) suggests that ozone production is dependent on the proportion of NO:NO_2 (i.e. if the concentration of NO exceeds 10 parts per trillion). Ozone formation depends largely on the presence of VOCs, as equations (2-4) to (2-7) drive ozone formation. Although VOCs are necessary for ozone generation, NO_X availability is the most important parameter affecting ozone formation due to the following major factors: (1) nitrogen oxides react quickly, (2) they often have a shorter lifetime than VOCs, and (3) require sunlight for the conversion of NO_2 to NO and O. The reaction rate with the QSSA (reaction rate denoted by k) is given by [36]:

$$[O_3]_{SS} = k\frac{[NO_2]}{[NO]} \qquad \text{Eq. 2-8}$$

Throughout the day the forward reactions are favoured, as there are plenty of hydroxide radical species (OH) present to initiate hydrocarbon oxidation reactions. These reactions result in conversion of NO to NO_2 by peroxy radicals (RO_2), which when coupled with the NO_x cycle is equivalent to net O_3 production. During periods with strong solar radiation, this conversion occurs rapidly, and the amount of ozone generated is high [36].

The focus of this thesis is mostly on ozone, but particulate matter (PM) also includes a significant secondary fraction. Primary PM refers to particulate matter that is emitted via anthropogenic (i.e. vehicle exhaust, combustion) or biogenic processes (i.e. dust, sea spray). Secondary PM refers to particles that are formed in the atmosphere but are not directly emitted from exhaust (i.e. nitrates, sulfates). PM is of a mixture of various constituents that have differing size, shape, solubility, surface characteristics, etc. These particles are classified based on their aerodynamic diameter and respective size designation. Particulate matter with an aerodynamic diameter of 2.5 microns or less is termed “fine” particulate matter or $PM_{2.5}$ [37]. A significant fraction of these particles are emitted as a result of gasoline and diesel fuel combustion in vehicles. Particulate matter with an aerodynamic diameter ≤ 0.1 microns is termed “Ultrafine” particles and are also a product of combustion processes (i.e. vehicle exhaust). The composition of particulate matter varies by location and emission source; however, the following species are common constituents: carbon (organic & elemental), water soluble ions (incl. Na^+, NH_4^+, K^+, Mg^{2+}, Ca^{2+}, Cl^-, NO_3^-, SO_4^{2-}), and elemental species (incl. As, Ni, Cd, Pb, Ti, Al, Fe). The proportion

of each constituent in combustion emissions varies with the chemistry of the specific fuel combustion [38].

$PM_{2.5}$ emissions are often represented in models based on prescribed speciation profiles, including elemental carbon (EC) or sometimes referred to as black carbon (BC), organic carbon (OC), and inorganic compounds (e.g. sulfates). Black carbon is the elemental portion of carbon that exists as an aerosol after emission. More recently, emissions of BC are the focus of increasing attention due to their effect on the climate and human health. Black carbon particles aid in facilitating the warming of the atmosphere by settling on snow and ice (in colder regions) and absorbing solar radiation. In 2014, 61.9% of total BC emissions in Canada were attributed to mobile sources. Of this 61.9%, off-road transport accounted for 54% of BC, on-road transport accounted for 32%, followed by rail (8%), marine (4%), and air transportation (2%) [39].

As discussed earlier, in the presence of VOCs and sunlight, emissions of NO_X go on to produce secondary pollutants, including O_3 and secondary $PM_{2.5}$ [47]. These pollutants have been strongly linked in the literature to negative health impacts [48-57]. Every day, the vast majority of us are exposed to traffic-related air pollution. In Canada, roughly 2, 4, and 10 million individuals live within 50, 100, and 250 meters, respectively, from a major road. Karner et al. (2010) assessed 41 monitoring sites and their proximity to roadways. The authors found that the concentration of pollutants emitted from on-road vehicles declined to background levels between 115-570 meters from the roadway [58]. In general,

the most exposed individuals of the population include those subjected to in-vehicle exposures, such as policemen, taxi drivers, chauffeurs, and so on. In addition, studies have found that when metabolic rate is considered, personal exposure is higher for cyclists and commuters who use the sidewalk or lane located directly beside the emission source [59-63].

Particulate matter emitted from the fuel combustion process is largely composed of (on a mass basis) Aitken and accumulation mode particles [64]. The Aitken mode refers to primary particles fresh from combustion, which typically are smaller than 0.1 (μm). Accumulation mode particles are slightly larger, with typical diameters ranging from 0.1 μm to 1 μm [64]. Once emitted, Aitken mode particles coagulate quickly, and the concentrations rapidly decrease with distance from the source [64]. While the composition of $PM_{2.5}$ varies, PM species generally have a relatively longer lifetime in comparison to NO_X, since it does not readily react to change form. Instead, it can travel significant distances and go on to negatively affect public health [39, 65-67]. As the aerodynamic diameter of these particles is so small, $PM_{2.5}$ can be inhaled deeply into the lungs [37].

Ozone is not directly emitted, rather it is formed through NO_X and VOC interactions. The rate and production depend on several pertinent environmental factors, including – local pollutant concentrations, location, relative humidity, and sunlight penetration. These factors combined are the main drivers in determining whether a location is a source or sink

for fresh O_3 molecules. Anthropogenic emissions of NO_X are comprised of NO and NO_2, with approximately 10 times more NO than NO_2 [35]. When the population is exposed to NO_2 and O_3, there are increased levels of negative health impacts [66], including respiratory illnesses and increased hospital admissions [67].

2.2 Quantifying NO_X and $PM_{2.5}$ Emissions in Canada and the US

Anthropogenic emissions of NO_X were estimated to be 2,242 kilotonnes in 2008. These values decreased to 2,076 kilotonnes in 2011 and to 1,928 kilotonnes in 2014 [40]. In Canada in 2016, 52% of total NO_X emissions were attributed to the transportation sector, which has been responsible for contributing more NO_X emissions than any other source since before 1990. Figure 2-1 shows the contribution from each sector to total Canadian NO_X emissions from the year 1990 to 2016. Total Canadian NO_X emissions have decreased by 25% from 1990 to 2016, which was largely driven by the implementation of more stringent NO_X emission standards for light-duty gasoline vehicles and trucks. Emissions from transportation and mobile equipment have decreased by 33% overall since 1990, including a 62% decrease in emissions from off-road vehicles and equipment, a 59% decrease in emissions from light-duty gasoline trucks and vehicles, a 49% decrease in emissions from marine transportation, and a 19% decrease in emissions from heavy-duty vehicles [41].

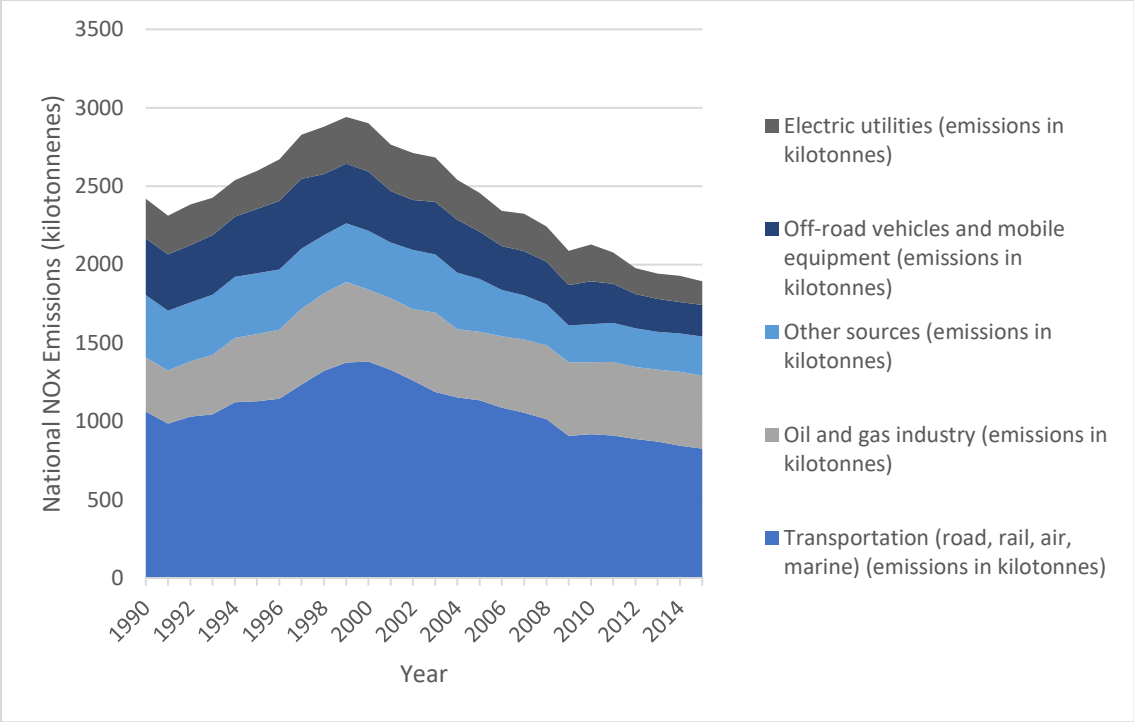

Figure 2-1: Total Nitrogen Oxides Emissions in Canada from 1990-2016 (adapted from [41])

In the United States, anthropogenic emissions of NO_X were estimated to be 16,909 tonnes of a total 17,987 tonnes emitted by all sources in 2008. In 2011, these values decreased to 14,574 kilotonnes of a total of 15,592 kilotonnes, followed by further decrease to 12,643 kilotonnes/13546 kilotonnes in 2014. In 2016, the transportation sector was responsible for 60% of the total annual average NO_X emissions, followed by stationary fuel combustion (26%) [42]. From 1990-2016, NO_X emissions from the transportation sector have decreased by 6500 kilotonnes [42]. The Ozone Transport Commission (OTC) was established under the Clean Air Act in the US to meet the NAAQS requirements for ground-level ozone. The slight increase in NO_X emissions from 2001-2002 is a result of the delayed entry of the Districts of Columbia and Maryland into the OTC protocol [171].

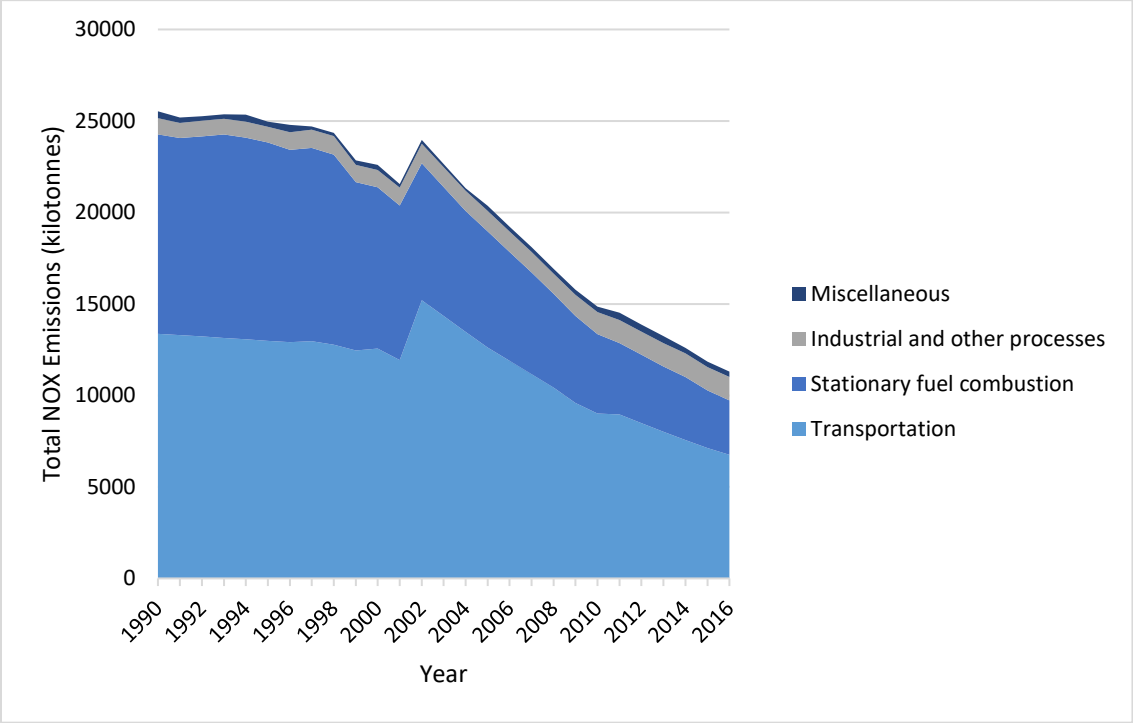

Figure 2-2: Total Annual Average Nitrogen Oxide Emissions in the United States (adapted from [42])

Canadian fine particulate matter emissions have decreased from 1990-2005; however, there has been an increase overall in emission levels between 2005-2014. Anthropogenic emissions of $PM_{2.5}$ were estimated to be 1615 kilotonnes in 2008. In 2011, these values decreased to 1594 kilotonnes and rose to 1646 kilotonnes in 2014 [40]. In 2014, total $PM_{2.5}$ emissions were larger (approximately 107%) than those in 2000 [43]. Open sources remain the largest responsible pollution source of $PM_{2.5}$ (69% in 2000 and 84% in 2014). The increase in emissions from open sources (130% increase from 2000-2014) has been attributed largely to new construction projects and operations as well as dust from roads. With the exception of open sources, emissions from all other sources considered “closed sources” have decreased since 2000 by 43% [43].

In the United States, anthropogenic emissions of $PM_{2.5}$ were estimated to be 6,014 tonnes in 2008. In 2011, these values decreased to 6306 kilotonnes followed by further reduction to 6223 kilotonnes in 2014 [44]. Overall, particulate matter emissions have decreased by 41% between 2000-2017, with an 18% decrease between 2010 and 2017 [45]. From the 2014 NEI, the main source of $PM_{2.5}$ is from stationary sources (61.5%). Fire and mobile sources followed with a share of 32.2% and 6.3% of the total 5.4×10^6 tonnes of $PM_{2.5}$ emitted into the atmosphere [46].

2.3 Linking Health Impacts and Emissions

The term "marginal benefits" (MB) is commonly used in environmental economics to quantify the health effects of reducing 1 unit (i.e. 1 tonne) of emissions. Through this lens, researchers may link health impacts to emissions from sources and further scale these results for any emission reduction scenario. Such linkage relies on economics as well as epidemiology. Epidemiologic investigations establish the relationship between a population's exposure and health impacts and quantify the excess risk of mortality for a change in concentration of a specific pollutant. Effect estimates for O_3, NO_2, and $PM_{2.5}$ are available for Canada and the United States and are presented in this section, followed by an introduction to health damage estimates in the literature.

2.3.1 Effect estimates for O_3

In the United States, one of the most well-known studies on effect estimates for O_3 was published by Bell et al (2004) [68]. This time-series study was conducted for the years 1987-2000 and aimed to investigate the relationship between short-term exposure and non-accidental mortality for the United States population. The published risk estimates refer to a percent change in risk of non-accidental mortality with a part per billion (ppb) change in O_3 concentration. The authors found risk estimates for 24-hour ozone, 1-hour ozone, and 8-hour ozone of 0.052%, 0.034%, and 0.043% for each, respectively. Following this, the effects of short-term exposure to O_3 on non-accidental mortality were quantified for 48 cities in the United States by Zanobetti and Schwartz (2008) [69]. The authors found a 0.053% increase in mortality risk per ppb for 8-hour maximum O_3. Katsouyanni et al. (2009) studied the association between 1-hour maximum ozone and total, respiratory, and cardiovascular mortalities in the Air Pollution and Health: A Combined European and North American Approach (APHENA) project. The project contained mortality information from 90 American cities, 22 European cities, and 12 Canadian cities. The authors found strong, positive associations between 1-hour maximum O_3 and cardiovascular mortality and published effect estimates of 0.076% per ppb for the United States [70].

In Canada, effect estimates have been found to be larger than those reported for the United States. In 2004, Burnett et al conducted a study using 12 Canadian cities to investigate the effect of short-term exposure to 1-hour ozone on mortality [71]. The authors estimated a 0.080% increase in mortality per ppb change in O_3. Stieb et al (2008)

expanded this time period from 1-hour to a 3-hour moving average, and found an effect estimate of 0.061% per ppb change in O_3 [72]. In the APHENA study, Katsouyanni et al. (2009) investigated the association between 24-hour average ozone concentrations were to determine the effect on all-cause mortality in Canada. The authors found an effect estimate of 0.083% per ppb for all-cause mortality [70]. In 2013, Farhat et al. analyzed the association between short-term O_3 exposure and all-cause mortality, respiratory mortality, and cardiovascular mortality in 12 Canadian cities for the years 1980-2001. The authors reported effect estimates of 0.11% per ppb for 1-hour ozone, and between 0.056% to 0.134% per ppb for cause-specific mortality [73].

The risk of death attributed to long-term ozone exposure was investigated by the American Cancer Society (ACS) in the Cancer Prevention Study II (CPS-II). The well-known study conducted by Jerrett et al (2009) is widely-referenced, and reports an effect estimate of 0.104%/ppb for respiratory mortality attributed to daily 1-hour maximum ozone for the United States [74]. In the same year, Krewski et al. (2009) reported an effect estimate of 0.2%/ppb for summertime ozone and all-cause mortality for the US [75]. A later analysis by Turner et al (2016) examined the association between chronic exposure to ozone and all-cause and cause-specific mortality for the United States. This large-scale study included 22 years of follow-up of a total of 669,046 participants (with 237,201 mortalities). The authors found significant associations between chronic O_3 exposure and all-cause mortality (0.2%/ppb), circulatory mortality (0.3%/ppb), and respiratory mortality (1.2%/ppb) [76].

In Canada, a study by Crouse et al (2015) examined the link between long-term ozone exposure and non-accidental mortality [77]. The authors report estimates of 0.188% increased risk of mortality per ppb increase in ozone concentrations.

2.3.2 Effect estimates for NO_2

Worldwide, a lack of consensus exists concerning whether NO_2 is directly linked to mortality. In Canada and in European studies, there is a clear association between NO_2 exposure and mortality; however, in the United States, it was not until recently that evidence of this association has been documented.

In a large-scale time-series study by Moolgavkar et al. (2013), the authors investigated the association between pollutant exposures and all-cause mortality in 108 cities in the United States for the years 1987-2000. The authors report statistically significant results showing strong associations between NO_2 and all-cause mortality (0.094%/ppb) [78]. Turner et al (2016) confirmed this association between NO_2 exposure and mortality, in fact the authors found NO_2 to be a strong, independent predictor of mortality [76]. In single pollutant models, the authors found strong association between NO_2 and all-cause mortality and circulatory mortality.

In Canada, and as early as 2004, Burnett et al. studied the effect of 3-day average NO_2 concentrations on non-accidental mortality for 12 Canadian cities over the period of

1981-1999 [71]. The authors reported effect estimates of 0.10%/ppb increase in 3-day average NO_2 concentration. In 2007, these findings were confirmed as Brook et al. reaffirmed that NO_2 has the strongest association with mortality (stronger than O_3 and $PM_{2.5}$) [79]. In 2008, Stieb et al. found an effect estimate of 0.062%/ppb increase in 3-hour maximum NO_2 concentration in a time-series study for Canada [72]. Crouse et al (2015) was the first to investigate the long-term effects of NO_2 exposure and mortality in Canada. The authors found effect estimates of long-term NO_2 exposure on non-accidental mortality of 0.129%/ppb [77]. This is consistent with the findings of Stieb et al (2002), who reported an effect estimate of 0.12%/ppb for 24-hour average NO_2 exposure on all-cause mortality in Canada [80].

2.3.3 Effect estimates for $PM_{2.5}$

Dominici et al. (2007) published effect estimates of short-term exposure to $PM_{2.5}$ on several health endpoints in the United States [81]. The authors found effect estimates of 0.029%/ $\mu g/m^3$ increase in $PM_{2.5}$ concentration on all-cause mortality and 0.038%/ $\mu g/m^3$ for cardiorespiratory mortality. Zanobetti and Schwartz (2009) analyzed the association between short-term exposure to $PM_{2.5}$ and mortality across 48 US cities. The authors found an effect estimate of 0.098%/ $\mu g/m^3$ for all-cause mortality [69].

The effect estimates related to $PM_{2.5}$ and mortality are consistently higher for Canada [82]. In 2000, Burnett et al found effect estimates of 0.120%/ $\mu g/m^3$ on mortality from short-term $PM_{2.5}$ exposure [83]. In 2003, Burnett and Goldberg published effect

estimates with a similar magnitude, ranging from 0.110-0.117%/ $\mu g/m^3$ for Canada [84]. Stieb et al. (2008) reported an effect estimate of 0.13%/ $\mu g/m^3$ for non-accidental mortality in Canada (using a 3-hour window of short-term $PM_{2.5}$ exposure) [72]. Most recently, Farhat et al (2013) published effect estimates for short-term exposure of between 0.098%-0.177%/ $\mu g/m^3$ in Canada, where all-cause mortality was not as significant as cardiovascular mortality [73]. Crouse et al (2012) quantified the long-term effects of $PM_{2.5}$ exposure and found an increase of 0.95%/ and 1.4%/ $\mu g/m^3$ in non-accidental mortality in Canada [85]. In 2015, the authors reported updated this effect estimate to a 0.22%/ $\mu g/m^3$ increased risk of non-accidental mortality with long-term exposure to $PM_{2.5}$ in Canada [77].

In the work published by Pope et al (2002), the authors investigate the effect of chronic exposure to $PM_{2.5}$ on all-cause mortality in the United States [86]. The authors found an effect estimate of 0.6%/ $\mu g/m^3$ increase in annual $PM_{2.5}$ concentration. Krewski et al (2009) found estimates of between 0.3-0.6%/ $\mu g/m^3$ for all-cause mortality in a reanalysis of Pope (2002) [75].

2.3.4 Marginal Benefits in the literature

In 2013, Pappin et al quantified the societal health benefits of reducing anthropogenic NO_X and VOC emissions in Canada and the United States [87]. The authors found significant differences between the results for both countries and attributed this to disagreements over the link between NO_2 and mortality. To this end, their estimates for

Canada account for health effects of exposure to both NO_2 and O_3; however, estimates are only presented for O_3 for the United States. The authors quantify the benefits of reducing short-term exposure to the aforementioned pollutants over the 2007 ozone season. More detail is presented in section 3.3 on the general formulation of the forcing term used to quantify the health impacts and how effect estimates are used to determine sensitivities in terms of dollar per tonne. The authors apply effect estimates for the United States of 0.00052/ppb (0.052%/ppb) for 24-hour ozone as well as 0.00039/ppb (0.039%/ppb) for 1-hour ozone. For Canada, effect estimates of 0.000839/ppb (0.0839%/ppb) for ozone as well as 0.000748/ppb (0.0748%/ppb) are used for NO_2. In order to quantify the health damage in monetary units, the authors use a value of statistical life of $5.7 million (CAD) for Canada and $8.1 million (USD) for the United States (again, more on this calculation is available in section 3.3). The results show that the total health benefits (through both O_3 and NO_2) are as high as $253,000/day in Hamilton, Ontario from a mere 10% reduction in NO_X emissions. In the United States, the health benefits of reduced exposure to O_3 alone due to 10% reduction is NOx emissions are as high as $181,000/day in Atlanta, Georgia for 24-hour O_3 and estimates for 1-hour ozone were consistently higher than those for 24-hour ozone.

Up to this point, the estimates shown assumed a linear relationship between exposure and health effects; however, more recent studies have shown that the relationship between exposure and health effects is not always linear due to the interactions between NO_X, VOCs, O_3, and $PM_{2.5}$. In fact, the concentration response

function (explained in detail in section 3.3) has been more accurately described with a supra-linear (or "non-linear") fit. Therefore, the relationship between chronic exposure to NO_X and $PM_{2.5}$ is directly non-linear, and complex photochemical interactions between the species when co-emitted may have more significant health implications. Consider an example: a city park with 10000 units of litter versus the same park with 10 units of litter strewn about the grass. In the first case, removing 9 units of litter would make little to no difference in the park; however, removing 9 unit of litter from a relatively clean location would make a significant impact on the state of the park's "cleanliness". This can be applied to the atmosphere, removing a unit of pollution from a clean environment should make a more significant impact than removing a single unit from a "dirty" environment.

Pope et al (2009) suggested that the shape of the CRF over very low concentrations is almost linear; however, at higher concentrations, the relationship is better defined by a supra-linear relationship [88]. The authors extend the analysis to analyze the shape of the CRF associated with lung cancer (linear) vs cardiovascular mortality (supra-linear). Marshall et al. (2015) discuss the shape of the curve of the concentration-response function (CRF) for ambient concentrations in the U.S., which describes the relationship between $PM_{2.5}$ and cardiopulmonary mortality [89]. The author explains how the traditional shape of the CRF curve (linear) is inadequate for modeling the response of PM_{25} exposure over larger concentration ranges and argues that as the baseline becomes cleaner, each additional reduction in concentration would yield a larger benefit. It is suggested that a supra-linear CRF (log-log scale of survival vs concentration) is more appropriate [89]. The authors also

determine the results of the supra-linear CRF assumption in comparison with a linear model and show the overall cardiopulmonary impact of $PM_{2.5}$, the number of premature deaths and the greatest marginal benefits on a per-capita basis. They conclude with the idea that emission reductions should be targeted to clean environments, as these correspond to larger benefits than emission reductions in more polluted areas; hence the title, "blue skies bluer" [89]. As stated in this work, although there is no "safe" threshold for $PM_{2.5}$, the authors suggest that the supra-linear assumption is worth investigating, especially for a broad range of concentrations.

Relating this to other recent literature, Pappin et al (2015) investigated the marginal benefit with emission abatement of NO_X across the United States [97]. The authors use the ozone season of 2007 (May 1st – September 31st) and investigate the sensitivity of reducing short-term exposure to ozone and ultimately quantify the impact of this on domain-wide mortality. In order to monetize these sensitivities, the authors apply a value of statistical life of $7.9 million (USD) and a short-term effect estimate of 0.000427/ppb (equivalent to 0.0427%/ppb) for 8-hour max ozone. The authors separate the health damage estimates into mobile and point source categories. Estimates for mobile sources show the benefit of reducing mobile emissions, as point source estimates show the benefit of reducing emissions from said point sources. The authors find increased monetary health benefits with increased emission abatement. For a reduction of 1 tonne of NO_X emissions, benefits ranged from $-86,000/ton to $87,000/ton for mobile sources (spatially variable) and between $-20,000/ton to $39,000/ton for point sources. The

domain-average values were reported to be $13,000/ton from mobile sources and $14,000/ton for point sources. The authors go on to state that combined reductions of both mobile and point source categories resulted in benefits 3-4 times larger than the original. It is also suggested that since a nonlinear model is more suitable for $PM_{2.5}$, the estimates found in the work are likely to extend to $PM_{2.5}$, yielding critical policy implications.

Furthermore, in 2016, Pappin et al. investigated the marginal benefits of progressive emission abatement in Canada (a "cleaner" environment) [98]. The authors use the same study period as in both 2013 and 2015 publications (May 1st – September 31st, 2007) and apply a value of statistical life of $7.17 million (CAD). The effect estimates used in this analysis are for chronic exposure based on 8-hour ozone and 24-hour NO_2 and vary based on the Cox proportional hazards model. The results showed that with progressive emission controls the marginal benefits increase with increased abatement for NO_2 and O_3, and state that the results may be extended to $PM_{2.5}$ with the supra-linear CRF model [97, 98]. For example, benefits with a linear exposure model were $270,000/ton, $460,000/ton, and $840,000/ton in Ottawa, Vancouver and Toronto, respectively. Applying a log-linear relationship for exposure to each pollutant, these estimates shifted to $500,000/ton, $510,000/ton, and $650,000/ton for Ottawa, Vancouver, and Toronto, respectively. Therefore, the supra-linear shape of the concentration-response function for these pollutants indicates a heightened sensitivity for populations in cleaner environments and suggests that larger benefits may be available in these locations. The idea of non-

linearity with PM has since been cited a number of times and it has been accepted that there is a supra-linear relationship between particulate matter and concentration [91-96], as well as greater marginal benefits in locations where the air is more pristine [91, 95].

Surely we have a responsibility to leave for future generations a planet that is healthy and habitable by all species.

Sir David Attenborough

3 Methodology & Modeling Framework

This section gives an overview of the core programs used to run the US EPA's Community Multiscale Air Quality (CMAQ) model (see Figure 3-1). Each program will be discussed in order, including the preparation in meteorology data in section 3.1, emission file preparation and gridding in section 3.2, followed by an overview of the CMAQ model and adjoint versions in section 3.3. The model updates discussed in the following sections describe the advancements made prior to this publication by the respective institutions and were used to perform the modeling presented later in this thesis.

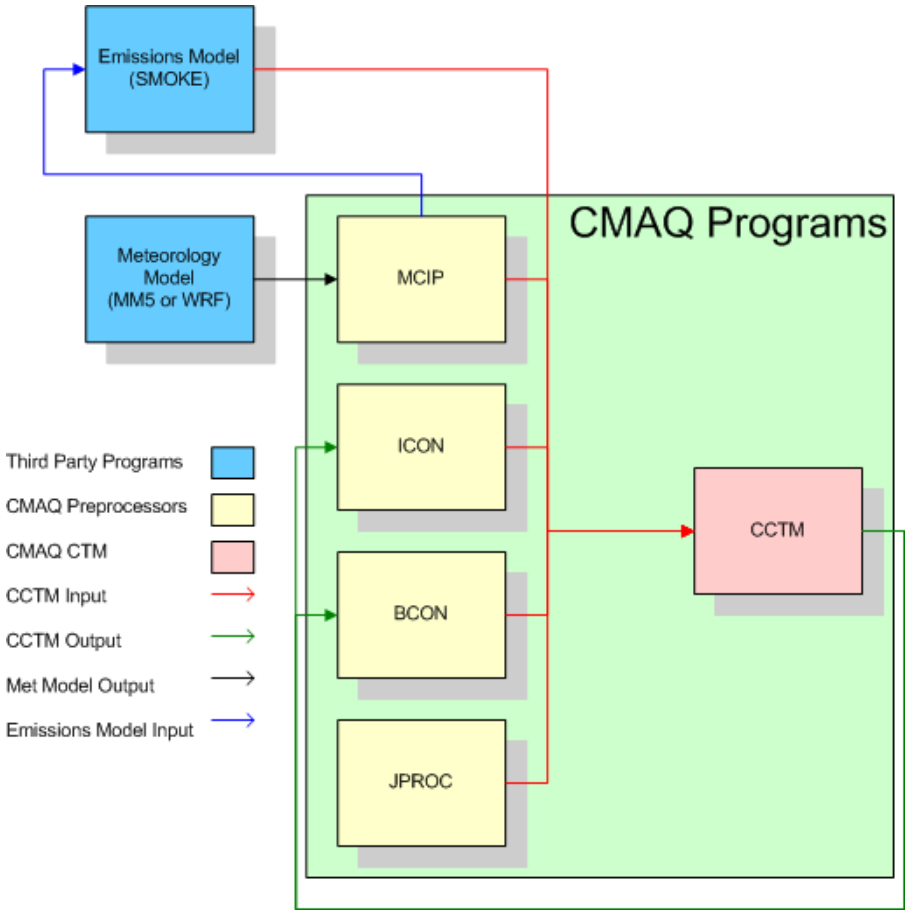

Figure 3-1: Core programs used in the CMAQ modelling process, including third party programs for meteorology (WRF) and emissions (SMOKE) processing, as well as the five major CMAQ modules [99].

3.1 Meteorological Modeling

Meteorological models are formulated based on dynamic atmospheric equations that can simulate the atmospheric circulation parameters (such as wind, temperature, pressure, etcetera), both at the surface and in the upper levels of the atmosphere.

3.1.1 The Weather Research and Forecast (WRF) Model

The Weather Research and Forecast (WRF) model is a regional and non-hydrostatic numerical weather prediction model that was developed by National Center for Atmospheric Research (NCAR), the National Oceanic and Atmospheric Administration (NOAA) groups, the Forecasting System Laboratory (FLS), the Air Force Weather Agency (AFWA), the Naval Research Laboratory, the University of Oklahoma and the Federal Aviation Administration (FAA) collaboration over the last two decades. WRF is a next generation atmospheric model and has two cores – including, the Advanced Research WRF (ARW, mostly used for modeling real case applications) and the Non-hydrostatic Mesoscale Model (NMM, mostly used for modeling ideal case applications). This thesis uses the WRFv3.8.1-ARW model due to the number of physical parameter options, compared to the NMM core [100], and the optimal parameter option was applied for North America. The WRF-ARW model has two fundamental components: the pre-processing system and the real data system (see Figure 3-2).

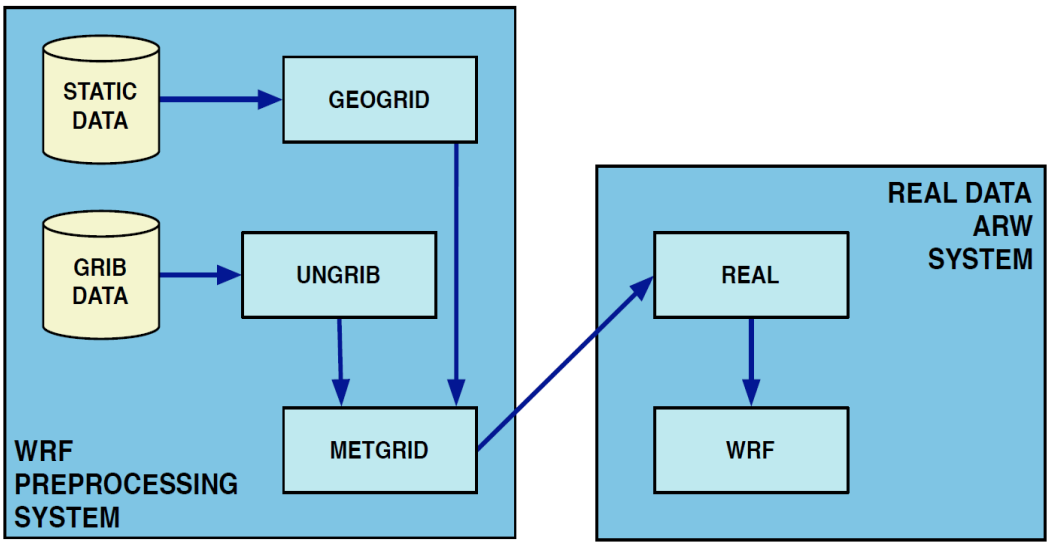

Figure 3-2: A depiction of the processes involved and the data flow within the major components of the WRF-ARW Model [100]

Running the WPS-Preprocessing module is a multistep process and involves creating static variables that may be used by WRF-ARW to simulate weather parameters. The WPS-pre-processing system is composed of three modules: geogrid, ungrib and metgrid. First, the geogrid module is run to prepare terrestrial data for the domain that is required by the numerical core of the model. It interpolates the static input data such as soil categories, land use/cover, terrain height, albedo, etc., according to a user-defined domain. Next, the Ungrib module extracts stacked global meteorological model output files (such as GFS, ECMWF). Lastly, the Metgrid module interpolates inputs and merges them for the defined domain [100].

WRFV3 is the main core of the WRF-ARW model and it contains two modules: real.exe and wrf.exe. The 'real.exe' module initializes real atmospheric data for the

specified domain. Metgrid files from the previous step are used by this module to prepare initial conditions (ICON) and boundary conditions (BCON) for the simulations. This output is fed into the second module, 'wrf.exe', which solves dynamic atmospheric equations and initiates the computation of atmospheric variables for the domain (e.g. temperature, specific humidity, incoming/outgoing radiation [100].

3.1.2 The Meteorology-Chemistry Interface Processor (MCIP)

The Meteorology-Chemistry Interface Processor (MCIP) software is used to ensure that the meteorological information which is received by the chemical transport model is accurate and dynamically consistent [101]. MCIP is used to post-process the raw output from the WRF-ARW model and is used to transform coordinates vertically and horizontally, check atmospheric fields, and imposes grid information, transforming the WRF output into CMAQ-ready data [101]. The shortcoming of this model is in its excessive efforts to interpolate the meteorology data, where in complex or dynamic situations, the accuracy is sometimes lost to computational intensity. This can be resolved by coupling the WRF-ARW meteorology model and the CMAQ air quality models (see Figure 3-3). With this update, a single code is used to run WRF and CMAQ in either coupled or uncoupled modes. Coupling the two models addresses the major limitations of the MCIP module [102].

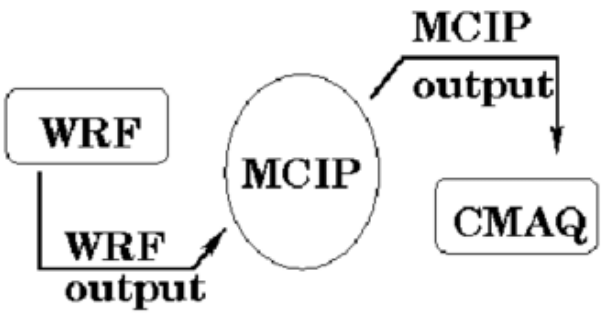

Figure 3-3: Schematic showing the change in high-level modules from the original (uncoupled/offline) WRF-CMAQ (modified from [102]).

3.2 Emissions Modeling

Version 3.7 of the Sparse Matrix Operator Kernel Emissions (SMOKE) Modeling System was used (in conjunction with Spatial Allocator version 4) in this thesis to develop speciated, temporally and spatially distributed emissions data. The SMOKE model is used to process annual emission inventories to create hourly, gridded, and ready-to-use emission files for the CMAQ model. The modeling system consists of three main parts: speciation, gridding, and temporal allocation [103].

First, raw emissions inventory files (ascii format) from the US National Emission Inventory (NEI) System [46] are used as inputs to the SMOKE model and reference files are created (and will be used by other algorithms in a later step). According to pre-defined speciation tables for several chemical mechanisms, the model applies the selected speciation profile for each source classification code (SCC). Next, the SMOKE model distributes the annual emissions across a desired grid resolution. Spatial allocator version 4 [104] is used to generate spatial surrogates for a specified domain projection (ascii files

that have grid numbers and their intersection ratios with your domain), which are used to distribute emissions into grid cells. If the user desires to modify the projection of the data, these files must be regenerated with the spatial allocator model. Temporal allocation is the last step of the SMOKE modeling system. To do this, the model calls information from pre-defined tables containing hourly, weekly, and monthly temporal profiles and activity ratios for each SCC. At the end of these three steps, The SMOKE model merges the intermediate files to create ready-to-use emission files in the format of the M3Models system (the main system employed in running CMAQ) [103].

3.3 Chemical Transport and Air Quality Modeling

Conventional (or "forward") atmospheric chemical transport or air quality models (CTMs or AQMs) are designed to numerically solve the atmospheric diffusion equation (ADE, Eq. 3-1 [105]) by simplifying the 3-dimensional ADE into a series of process-based equations.

$$\frac{\partial C_i}{\partial t} = -\boldsymbol{u} \cdot \nabla \mathrm{C_i} + \left(\frac{1}{\rho}\right) \nabla \cdot (\rho \mathrm{K} \nabla \mathrm{C_i}) + \mathrm{R_i} + E_i \qquad \text{Eq. 3-1}$$

Where **u** is the vector of wind field , C_i is the mixing ratio, ρ is the density of the air, **K** is the diffusivity constant, R_i is the reaction rate for nonlinear chemical and thermochemical processes, and E_i is the emission rate of a species *i*. With the definition of initial and boundary conditions, CMAQ solves the differential of the ADE in Eq. 3-1 to generate a concentration vector.

The CMAQ model was first released in 1998. Since then, the model has continued to be the main CTM used in air quality modeling and policy management in the United States [106]. A timeline showing the evolution of the model and the updates since 1998 is shown in Figure 3-4 [106].

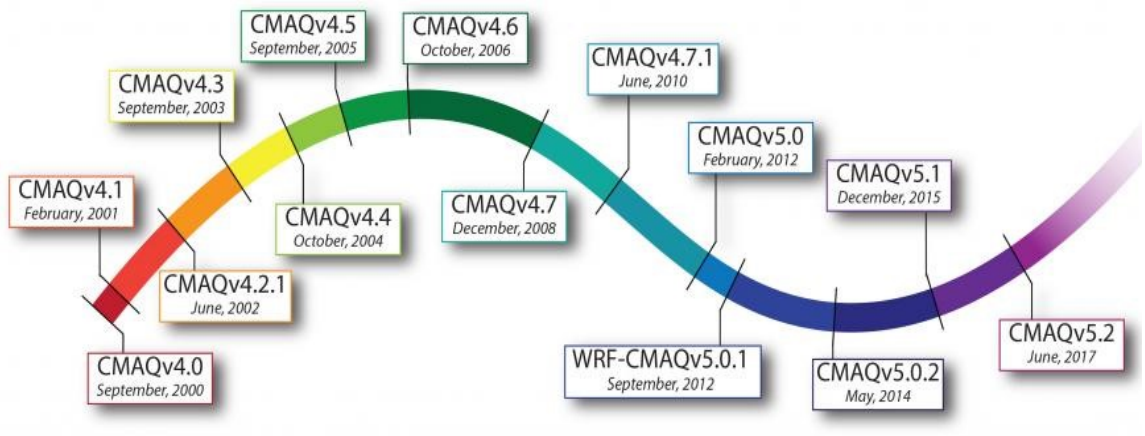

Figure 3-4: Timeline showing all of the updates made to the CMAQ model since 1998 [106].

The CMAQ model is a collection of science modules that work together to estimate pollutant interactions, transformations, and deposition over space and time. CMAQ is an open-source CTM that can be used to simulate various policy or research applications. The model contains five major modules, including MCIP (Meteorology-Chemistry Interface Processor), JPROC (Photolysis Rate Processor), BCON (Boundary Condition Processor), ICON (Initial Condition Processor) and CCTM (The CMAQ-Chemical Transport Model). An overview of the CMAQ modeling methodology is presented in Figure 3-5.

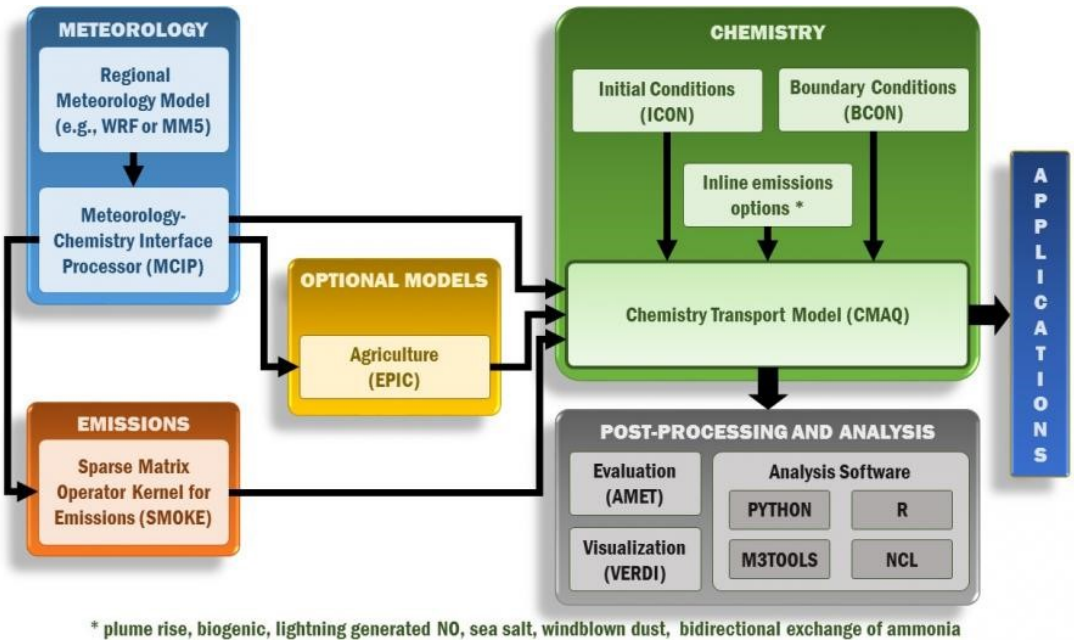

Figure 3-5: Flow diagram showing the multiple processes involved in the CMAQ model and the relationship between processes [106].

After MCIP processes and converts meteorology information for CMAQ, the JPROC module prepares clear sky photolysis rates from pre-defined longitudinal datasets. The ICON and BCON modules prepare the initial and boundary conditions for the chemical transport model simulations. Both modules have two options: to either use a pre-defined longitudinal dataset for each pollutant, or to use dynamic conditions that have been generated using a larger (parent) domain in a nested series of CMAQ simulations. In both cases, dynamic initial and boundary conditions from global transport models are used in the model runs for the simulations presented in this thesis. Lastly, the CCTM module is the numerical core of CMAQ and is responsible for dynamically solving pollutant deposition, transportation, and reaction rates for each simulation [107].

CMAQ-Adjoint

In an adjoint sensitivity analysis, the opposite (or reverse) approach to more conventional forward sensitivity analysis is employed. In this approach, a perturbation is made to a receptor and the model is run in the reverse direction to trace the impact back to all sources and inputs of the model. This type of analysis allows for additional "what-if" analyses that cannot be feasibly studied with a conventional forward model.

To perform an adjoint analysis, the CMAQ model must be run in the forward (conventional) direction to generate checkpoint files as well as concentration vectors. Checkpoint files from the forward run are used by the adjoint model to represent the model state in non-linear processes as the adjoint model traces impacts backward through time. In order to attribute health impacts to the changes in concentration with emissions, a scalar cost function must be defined on a basis of change in concentration, to investigate its sensitivity to various parameters.

Recap of the Adjoint Derivation

In this short section, an overview of the adjoint derivation is presented, as published by Hakami et al (2005). The example derivation shown here is for black carbon inversion specifically and is unique as black carbon has no chemistry; however, the full derivation and more details are available elsewhere [109]. Modeling the dynamics of non-reactive species with the use of an atmospheric chemical transport model (CTM) is governed by the following equation:

$$\frac{\partial C}{\partial t} = -\nabla \cdot (\boldsymbol{u}C) + \nabla \cdot (\boldsymbol{K}\nabla C) + E - k_W C \qquad \text{Eq. 3-2}$$

Where **u** denotes a vector wind field, C is the concentration of the given species, **K** represents the diffusivity tensor, E is the elevated emissions, and k_W is the first order rate constant (wet removal). In an adjoint sensitivity analysis, a set of inputs are defined and the gradients of a predefined cost function, J, are calculated. Eq. 3-3 provides a definition of an objective cost function for inversion for example (other types of cost functions are discussed in the original publication and elsewhere [109]):

$$J(C,\alpha) = \frac{1}{2}\left[\frac{1}{\mu}\int_{t_0}^{t_F}(\alpha^b - \alpha)^T N^{-1}(\alpha^b - \alpha) + \int_{t_0}^{t_F}\left(\hat{C} - C\right)^T R^{-1}\left(\hat{C} - C\right)\right] \qquad \text{Eq. 3-3}$$

Here, $\hat{C}$ is a vector containing the observed concentrations, C is a vector containing simulated concentrations, t_0 is the start time of the simulation, t_F is the final time of the simulation, α is the predefined input parameters, μ is a weighting factor that places emphasis on either observed or background estimates, and a^b is a vector containing background estimates for the parameters in α.

Since the desired outcome of this modeling approach is to incite a perturbation in one of the predefined inputs (denoted as $\delta\alpha$) and to determine the influence on the model simulated concentrations (δC) and on the cost function (δJ). Through the addition of a lagrange multiplier to Eq. 3-3, allows for the calculation of the gradients of the cost

function. Once computed, the gradients are both location- and time- dependent; however, Eq. 3-4 must be satisfied by the lagrange multiplier, λ, and the conditions listed in Eq. 3-5 to Eq. 3-8 must be satisfied [109].

$$-\frac{\partial \lambda}{\partial t} = \nabla \cdot (\boldsymbol{u}\lambda) + \nabla \cdot (\boldsymbol{K}\nabla\lambda) - k_W\lambda + \phi(\omega, t) \qquad \text{Eq. 3-4}$$

$$\lambda(t_F) = 0 \qquad \text{Eq. 3-5}$$

$$\lambda|_{\omega\in\Gamma^{\text{out}}} = 0 \qquad \text{Eq. 3-6}$$

$$K\frac{\partial \lambda}{\partial \omega}|_{\omega\in\Gamma^{\text{out}}} = 0 \qquad \text{Eq. 3-7}$$

$$\left(K\frac{\partial \lambda}{\partial \omega} - v_d\lambda\right)|_{\omega\in\Gamma^{gr}} = 0 \qquad \text{Eq. 3-8}$$

Further details on the derivation of adjoint equations and boundary conditions for a CTM are available elsewhere [108-112]. Following this, the cost function is defined below for the simulations in this work.

Adjoint Cost Function Definition

The identification of emission sources and the related health impacts is an inter-disciplinary problem, involving the gathering and processing of information from economics, epidemiology, and air quality modeling. The general equation for monetizing marginal benefits can be simplified in the form of Eq. 3-9 [113].

$$\left(\frac{\Delta\$}{\Delta Emissions}\right) = \left(\frac{\Delta\$}{\Delta Mortality}\right) \cdot \left(\frac{\Delta Mortality}{\Delta Concentrations}\right) \cdot \left(\frac{\Delta Concentrations}{\Delta Emissions}\right) \qquad \text{Eq. 3-9}$$

The first term in the equation (Δ$/ΔMortality) is derived from economics and is used to monetize health impacts through the incorporation of a parameter referred to as the value of statistical life, V_{SL}. The V_{SL} is an estimation of an individual's willingness to pay to avoid death. As population and income demographics may vary drastically between countries, and the V_{SL} applied in this equation must be representative. The second term of Eq. 3-9 (ΔMortality/ΔConcentrations) links data from epidemiology on mortality rates due to changes in pollutant concentrations and contains pollutant-specific information about the risk, exposure rates, and effect estimates of exposed populations. The third term in Eq. 3-9 (ΔConcentrations/ΔEmissions) incorporates air quality modeling into the health impact estimation process. To generate this term, CTMs are used to investigate how a perturbation in emissions affects concentrations.

To determine the societal public health benefits across Canada and the United States for the purposes of this thesis, an adjoint cost function must be defined (and must exist as a function of concentration). As discussed in section 2.3, the health impacts of NO_2 are not considered to be directly linked to mortality in the United States. Although a single-pollutant model may be the more common form of existing concentration response functions, single pollutant models do not account for co-pollutant interactions (such as the effect of NO_X on secondary $PM_{2.5}$, for example), and the single pollutant health impact function may overestimate the effects of individual pollutants. As data on the health

effects of NO_X, O_3, and $PM_{2.5}$ is available in Canada and current research supports the effect estimates of a joint non-linear model, a 3-pollutant model is employed in this thesis for Canada when the impact of all pollutants is considered. Although the general equation used in both analyses are similar, the results for Canada and the United States should be considered separately, as the epidemiology, emission magnitudes, and economic valuation vary between the two countries (disagreements over the health effects of NO_2, differences in the value of statistical life, etc.). The health impact models and parameters are discussed separately below.

In the United States, health benefits from chronic exposure to O_3 from emissions of NO_X are estimated using the US EPA's CMAQ Model, followed by the CMAQv4.5.1 gas-phase adjoint model (equipped with the SAPRC99 chemical mechanism). Health benefits were modeled for each pollutant separately over 34 vertical layers and at a 36-km horizontal resolution over the continental US (CONUS) for July 1-31st, 2007. The cost function used to estimate the domain-wide societal health benefits is shown in Eq. 3-10 [98]. More information on the derivation of this function is available elsewhere [98].

$$J = V_{SL} \sum_{\omega} M_{0,\omega} P_{\omega} \left(1 - e^{-\beta \overline{C_{\omega}}}\right) \qquad \text{Eq. 3-10}$$

Here, in Eq. 3-10, J is the monetized number of non-accidental mortalities associated with O_3 (or $PM_{2.5}$) exposure in the US on a per-year basis. A value of statistical life, V_{SL}, is estimated based on a populations' willingness to pay to avoid mortality and is used to

monetize the health sensitivities to pollution. This value varies by country and is assigned to be $7.9 million in the U.S [114]. M is the non-accidental mortality rate in grid cell w, for populations, P_W, between the ages of 33-99. The equation is written separately for emission reductions of NO_X and $PM_{2.5}$, as separate pollutant-specific concentration response factors, β, are applied for chronic exposure to O_3 and $PM_{2.5}$ [74]. The sensitivities across the domain are estimated using the forcing term, as shown in equation Eq. 3-11 [98].

$$\varphi = \frac{V_{SL} M_{O,W} P_W \beta e^{-\beta \overline{C_W}}}{\bar{t} n} \qquad \text{Eq. 3-11}$$

Where φ is the adjoint forcing term, n is the number of simulation days, t is the number of hours in the exposure metric (1 for 1-hour ozone), and C_w is the 1-hour max ozone concentration at each location, w.

In the analysis for Canada, health benefits from chronic exposure to emissions of NO_X and $PM_{2.5}$ are estimated using the CMAQ adjoint model, equipped with the Carbon Bond-V (CB05) chemical mechanism and the AERO5 aerosol module with aqueous chemistry [115, 116]. The Hemispheric CMAQ model is run first at a 108-km horizontal grid resolution and 35 vertical layers first to create initial and boundary conditions (ICON and BCON files) for the forward model simulations using EDGAR emissions for the study period. These ICON and BCON files are used by the CMAQ forward model (version 5.0) and pollutant concentrations are estimated for the modeling period (July 1-16th, 2010). The

concentration outputs from the forward model are passed as inputs to the CMAQ-adjoint model. The adjoint model simulation is performed at a 36-km horizontal grid resolution for the full domain encompassing Canada over 35 vertical layers. Estimates of health damage are generated for Canada using a joint (3-pollutant) health cost function that accounts for exposure to NO_2, O_3, and $PM_{2.5}$ (simultaneously). The multi-pollutant model (Eq. 3-12) is used to form a scalar adjoint cost function that is non-linear in concentration for both NO_2 and $PM_{2.5}$. Detailed information on the generation of the cost function is available elsewhere [87, 97, 110]. The adjoint cost function and forcing equation have been adapted from a recent publication by Pappin et al. in 2016 [98].

$$\Delta M = M_0 \cdot P \cdot V_{SL} \cdot \left(1 - \frac{1}{RR}\right) \qquad \text{Eq. 3-12}$$

Here, ΔM is the monetized mortality across Canada that can be attributed to emissions of a specific pollutant for the year 2010. The equation incorporates information about the annual, non-accidental mortality rate, M_0 as well as location- and age-specific population data, P. The equation has been updated to include a complex translation of concentration, based on [117] and [118]. We apply pollutant-specific effect estimates, β, for chronic exposure to NO_2, O_3, and $PM_{2.5}$. To achieve the monetization of benefits for each location, we multiply the function by the current estimate of the Value of Statistical Life, V_{SL}, for the domain.

It's not that the world hasn't had more carbon dioxide, it's not [that] the world hasn't been warmer. The problem is the speed at which things are changing. We are inducing a sixth mass extinction event kind of by accident and we don't want to be the 'extinctee,' if I may coin this noun.

Bill Nye

Aug 7, 2014 Interview with Big Think

4 Estimating the Tipping Point of Urban NO_X Control in Major US Cities

This chapter is a manuscript in preparation for publication. It consists of original research for which Angele Genereux is the main contributor.

4.1 Introduction

One of the most proven and effective strategies for reducing ozone in most environments is through the control of nitrogen oxide (NO_X) emissions. Aggressive regulatory efforts have been made to reduce NO_X emissions from a number of emission sources from 1990 to 2010 [32]. Although these NO_X controls have resulted in a decrease in nationwide ozone concentrations, local concentrations have increased in several urban and suburban areas [33, 34].

In the troposphere, ozone is a secondary pollutant that is formed through photochemical interactions primarily between two precursors: NO_X and volatile organic compounds (VOCs). The relationship between these precursor species and ozone is important in environmental policy. This relationship is illustrated by ozone isopleths, more information on which can be found in more detail elsewhere [119].

The ozone formation process is highly nonlinear with respect to NO_X and VOCs. This non-linear relationship has been well-documented in previous publications concerning

ozone formation in heavily populated and polluted areas [97, 98, 119, 120]. The rate of formation is largely dependent on the local availability of NO_X as well as the number of free radicals generated by sunlight. A NO_X-limited regime refers to the area on the ozone isopleth that exists below the ozone ridge and can be described as a region with a low background NO_X concentration. In this regime, increases in NO_X emissions result in increased rates of ozone formation and additional increases in VOCs have little to no effect on the rate of ozone formation. NO_X-limited regimes are commonly found in rural or clean air environments, where there are fewer pollution sources [119].

A NO_X-inhibited regime is depicted by the area on the ozone isopleth that is above the ozone ridge and is best described as a region with high background NO_X concentrations. These regimes are common for heavily populated and polluted areas (e.g. city centers and urban cores). In this regime, additional NO_X would decrease the rate of ozone formation. Conversely, reducing NO_X in this environment would initially increase the rate of ozone production, until the region reaches the ozone ridge. Therefore, policies aiming to reduce ozone levels must do so by targeting reductions in precursor emissions of NO_X or VOCs, based on the photochemical regime [119].

In recent literature, the health impacts from air pollution exposure have been commonly referred to as “Marginal Benefits” [87, 121-124]. Marginal benefits are defined as the public health benefit resulting from emission reductions in a location and are estimated based on averted mortality in most cases. In most locations across North

America, reducing NO_X results in a reduction of O_3, and positive marginal benefits. Conversely, reducing NO_X emissions in NO_X-inhibited locations (urban cores/population centers), would initially increase the rate of ozone formation, resulting in a negative health benefit referred to as a "disbenefit" [87, 125-127].

In the analysis conducted by Pappin et al. (2013), the authors use the Community Multi-scale Air Quality (CMAQ) model to estimate marginal benefits through reduced short-term ozone exposure resulting from a 10% reduction in NO_X emissions in the United States. In most locations across the US, reducing NO_X emissions by 10% resulted in positive marginal benefits (e.g. $181,000/day in Atlanta, Georgia); however, the authors estimated substantial disbenefits in heavily populated cities, with disbenefits reaching up to $-681,000/day in New York, NY, and up to $-244,000/day in Los Angeles.

A later analysis by Pappin et al. (2015) used the CMAQ model to investigate how the marginal benefit estimations change with progressive emission abatement [97]. The authors estimate the sensitivity of domain-wide mortality due to short-term ozone exposure for multiple abatement scenarios. At baseline emission levels, positive benefits are wide-spread across the domain, while local disbenefits are concentrated in urban cores and population centers. As NO_X emissions are reduced in intervals of 20% from 0-100% abatement, the atmospheric regime shifts away from NO_X-inhibited. At some point along the abatement pathway, this shift can result in a change in the health damage estimate from negative to positive (disbenefits to benefits). This is apparent around Los Angeles,

where the mobile marginal benefit at 0% abatement is $-17,000/ton. With progressive emission abatement, the regime shifts away from NO_X-inhibited and the health benefits maximize to $152,000/ton (after 100% abatement).

In this work, we aim to extend the research on the nonlinearity of NO_X in urban areas. To do this, we estimate the marginal benefit of NO_X control due to chronic exposure to 1-hour maximum ozone on a per-ton basis over emission abatement levels of 0 to 80%, in two NO_X-inhibited regions in the US: Los Angeles and New York City. As suggested in Pappin et al. 2013, to adequately account for the non-linearity in ozone response, a multi-step analysis of health benefits is more appropriate in modelling the benefit-per-ton (BPT) response with emission abatement [87]. Therefore, in this study, NO_X emissions are reduced in 10% intervals from baseline 2007 emission levels. We generate a marginal benefit curve (see Figure 4-1), showing the benefit at each successive abatement scenario, and we identify two key parameters: the Tipping Point (TP) and the Break-Even Point (BEP).

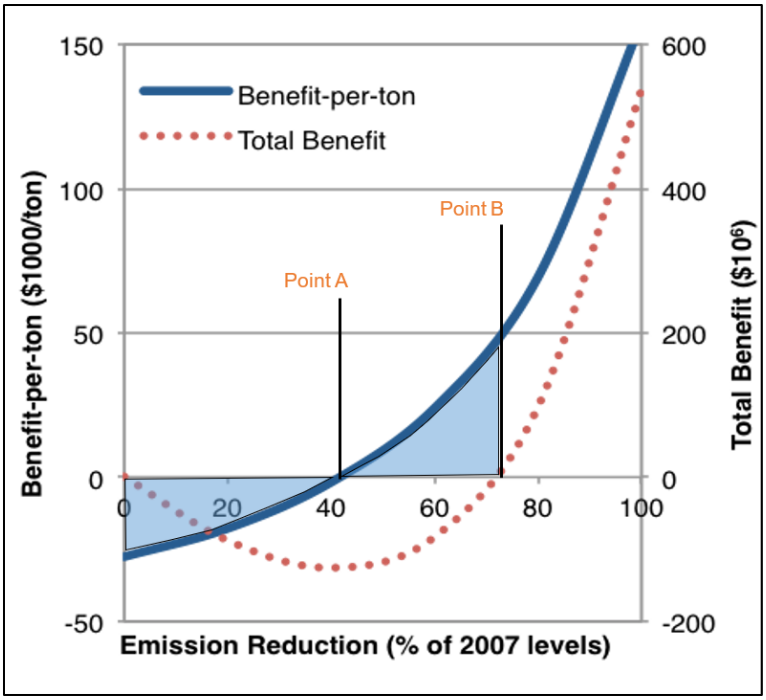

Figure 4-1: Conceptual plot of the benefit-per-ton over 0-100% emission abatement.

The TP is point A in Figure 4-1 and refers to the abatement percentage that corresponds to where disbenefits become positive (i.e. where the region shifts from NO_X-inhibited to NO_X-limited). We identify Point B in Figure 4-1 to be the point at which cumulative benefits compensate immediate disbenefits along the abatement pathway and refer to this as the BEP. The break-even point is a theoretical location of the point at which benefits are compensated if the pace at which emissions are abated remains constant. Theoretically, one could reduce emissions up to the tipping point and remain in a state of cumulative benefits for each year thereafter, shifting the location of the break-even point. We aim to address a unique question for policy makers: "how can local (vs. widespread) emission reductions and controls prove to be favourable policy options in a NOx-rich city where NOx-control disbenefits are prevalent?". To answer this, we estimate the benefits

for both national and local emission control policies, the total benefit of either control policy to the entire US domain, and the regional benefit (local benefit in LA & NY only). These two parameters allow us to ultimately characterize the nonlinearity of NO_X marginal benefits, providing useful incentive to policy makers and providing guidance on how national versus regional control policies will affect the immediate areas.

4.2 Methodology

We generate the MB curves for two major anthropogenic abatement scenarios through the utilization of the US EPA's CMAQ Model [107]. We use the CMAQv4.5.1 gas-phase adjoint model with the SAPRC-99 chemical mechanism to perform our analyses. A detailed description of the adjoint model is provided elsewhere [105, 110]. Our domain consists of the continental US at a 36-km horizontal grid resolution and with 34 model layers. Our analysis consists of 2 abatement cases generated from a forward and backward CMAQ simulation. Due to the computational requirements of a large number of simulations, we limit our analysis to one month of the ozone season: July 1 to 31, 2007. We use emission inventories from the National Emission Inventory (NEI) for the US domain and the National Pollutant Release Inventory (NPRI) for Canada, and the Sparse Matrix Operator Kernel Emissions (SMOKE) model to distribute the emission inventories over the continental U.S [103]. We supply the model with meteorological inputs from the Weather and Research Forecasting model (WRF) [100].

We apply a linear in concentration adjoint cost function, as defined in Eq. 4-1. More detailed information is available [98]:

$$J = V_{SL} \sum_{W} M_{0,W} P_W \left(1 - e^{-\beta \overline{C_W}}\right) \qquad \text{Eq. 4-1}$$

Here, J is the monetized number of non-accidental mortalities associated with O_3 exposure in the US on a per-year basis. We assign the value of statistical life, V_{SL}, to be $7.9 million USD [114]. M is the non-accidental mortality rate in grid cell w, scaled down from rate per year to rate per day. The non-accidental mortality rate is given for populations between the ages of 33-99. We apply a concentration response factor, β, of 3.92 x 10^{-3} based on Cox proportional-hazard models for chronic exposure to 1-hour O_3 in the US [74]. We supply the adjoint model with the forcing term as shown in Eq. 4-2 (more information is available [98]), in order to determine sensitivities across the domain [105].

$$\varphi = \frac{V_{SL} M_{O,W} P_W \beta e^{-\beta \overline{C_W}}}{\bar{t} n} \qquad \text{Eq. 4-2}$$

Where φ is the adjoint forcing term, t is the number of hours in the exposure metric (1 for 1-hour ozone), n is the number of simulation days, and C_w is the 1-hour max ozone concentration at each location, w.

4.2.1 Case 1: Abatement in NY & LA Only

We analyze the BPT curves for emission abatement in New York, NY, and Los Angeles, CA, by running the forward CMAQ model with the implementation of local emission reduction strategies. To do this, we generate a mask file, using US population census data to identify the areas of interest. We choose a representative domain for each region, based on population information. For each scenario, we use fixed-percentage abatement of anthropogenic emissions of all species in these cells (everywhere else emissions remain at baseline levels) of 0% to 80%, in 10% intervals.

4.2.2 Case 2: Nation-wide Abatement

We model the BPT curves for nationwide emission abatement in the US. We use fixed-percentage abatement of anthropogenic emissions of all species across the domain in intervals of 10%. In both cases, we assume biogenic emissions to be constant in our analysis as they are predominantly a function of vegetation cover and meteorological conditions [98, 128].

At the conclusion of this modelling process, our results allow us to estimate the tipping point (where BPTs become positive) of local NOX emission control in NY and LA, as well as the break-even point, where the cost of abating emissions is equal to the benefit gained (case 1). Our second set of simulations allows us to compare the cost of local versus national emission reduction strategies and to estimate how the location of the tipping point and the break-even point (optimal abatement percentage) vary based on these

emission reduction implementations. The inter-comparison of these results gives an indication of the influence of surrounding cities on the concentrations and health damage attributable to 1-hour chronic ozone exposure in NY and LA.

4.2.3 Influence of Pollutant Transport on the Independent Consideration of Sensitivities

To consider the influences over each regional domain independently, our results include the inherent assumption that the emissions in LA and NY do not have a significant impact on each other over the study period. In order to justify the assumption that emissions in these locations can be considered independently within the same CMAQ run, a sensitivity analysis is required to determine the extent of the impact of pollutant transport in the atmosphere.

To do this, we use the adjoint of CMAQ model to generate a set of concentration outputs at baseline emission levels for the same study period (July 1st-31st, 2007). We use spatial allocator version 4 [104] and ArcMap10.5.1 to generate a new mask file to identify the cells within the New York region only. We define our forcing term as given in Eq. 4-2 and supply the model with the command to force only the cells within the New York domain and to otherwise set the forcing term in the remaining cells to zero. We run the CMAQ-adjoint model for the study period and post-process the results by employing an emission-weighted averaging approach.

4.3 Results

The results in this section show the sensitivity of the single pollutant model (based on the linear epidemiological model for O_3) for both LA and NY separately, for two emission reduction scenarios: regional and national abatement. The health benefits, or marginal benefits, are calculated for both the national domain (entire United States) and the local domain (region-based). The national results show the national monetary public health damage from reducing emissions in a given location. As discussed in Pappin et al 2016, MB values are location specific and reducing emissions in a given location results in benefits beyond local, state, or national boundaries [98]. In this way, the benefits are calculated for the contiguous U.S. as a whole and emission reductions in a given location influence this national benefit. The local results show the local monetary public health damage from reducing emissions in a given location; a metric that may be more significant for local policy making. Here, the benefits of reducing emissions are calculated for two smaller domains (LA and NY regions), and therefore the benefits are calculated and show the influence of reducing emissions for the LA and NY regions, separately.

4.3.1 Influence of Pollutant Transport on the Independent Consideration of Sensitivities

We analyze the extent of the influence of pollutant transport from the Los Angeles region on the New York domain to justify the assumption that the results presented in this work for LA and NY may be considered independently. To perform this analysis, we provided a mask file to identify the grid cells corresponding to the NY region. The CMAQ and CMAQ-adjoint models were run for the period of July 1st -31st, 2007. The forcing of the

adjoint model were performed for cells within the NY domain (exclusively) to investigate the sensitivity of the NY region to emission reductions in all locations across the CONUS domain. The results are shown in Figure 4-2.

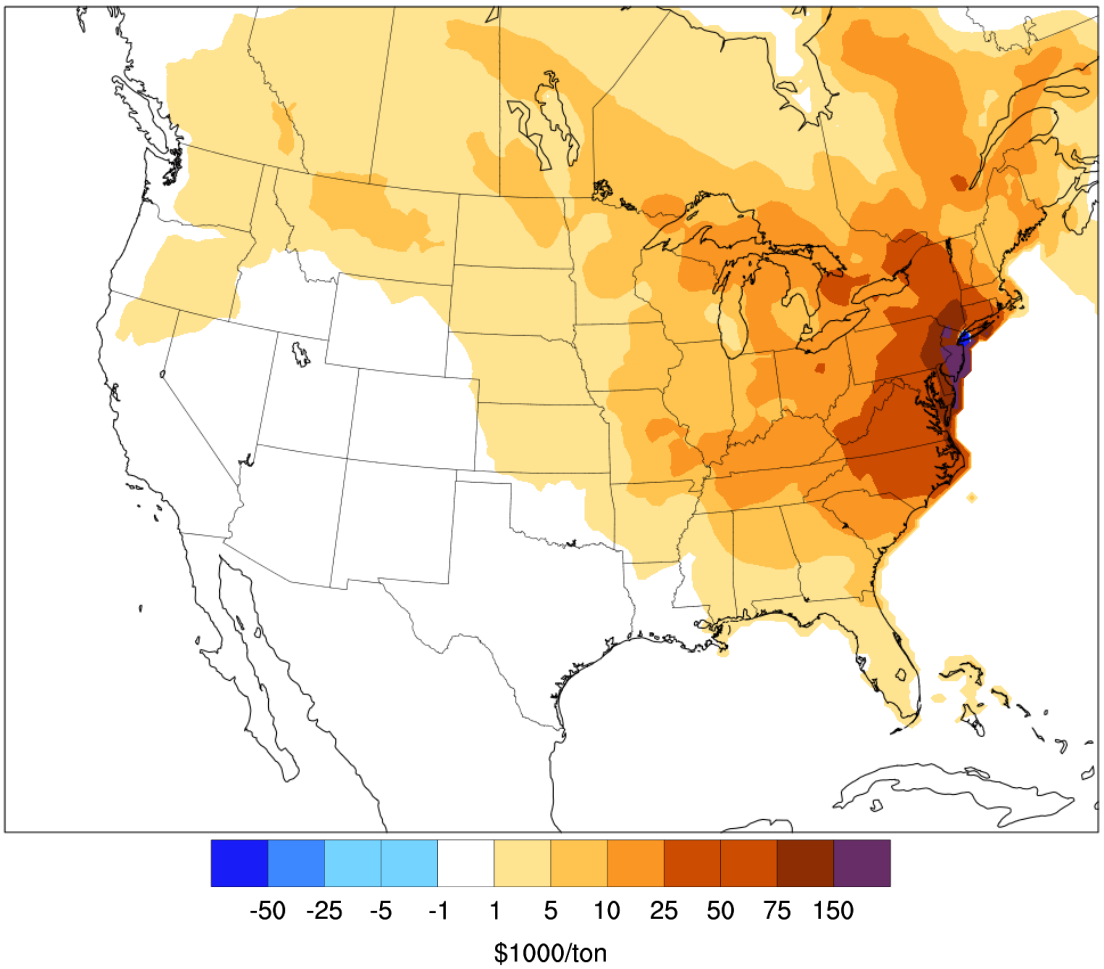

Figure 4-2: Sensitivity of the New York domain to emission reductions in grid cells across the CONUS domain. The results are presented in terms of $/ton of NO_X emission reductions.

As expected, we find that the sensitivity of neighboring cells around New York to have a more significant influence (ranging from -$300/ton to $150 000/ton) on the NY region, and found that these sensitivities decrease approaching the west of the domain. We found a sensitivity of $108/ton in Los Angeles and an average sensitivity of $7079/ton

for the entire U.S. domain. Therefore, the influence of reducing NO_X emissions by 1 tonne in the grid cell corresponding to LA will contribute $108/ton to the NY domain, accounting for a mere 1.5% of the influence on the NY region. Given the predominantly westerly direction of winds in the continent, we did not estimate the impact of NY emissions on LA mortality. Therefore, we conclude that this influence can be considered negligible and the domain-specific results presented in this paper are can be considered independently for both LA and NY. This assumption allows us to save significant computational time by forcing both regional domains in the same simulation without significant loss of accuracy.

The estimates presented in the following sections are reported for a single grid cell corresponding within the boundaries of Los Angeles and New York, based on the maximum population in each 36x36-km grid cell. The result of reducing emissions in each abatement scenario is therefore reported for a single grid cell and is presented as a nation-wide average benefit (encompasses the entire CONUS domain) as well as a regional average benefit (averaged over a smaller domain).

4.3.2 Case 1: Regional Abatement in New York and Los Angeles

In this case, we analyze the sensitivity of the model to regional emission perturbations in 10% intervals in NY and LA only. In every other location across the U.S., the emissions are held constant at baseline levels. Figure 4-3 contains spatial plots over LA and NY at each abatement interval.

At baseline 2007 emission levels, the nation-wide societal public health benefit of reducing NO_X in LA is estimated to be \$-505,000 tonne^{-1}, indicating that a reduction in NO_X emissions by 1 tonne in LA would initially cause more damage to public health. At each successive abatement level, the benefits are calculated and are found to consistently increase with progressive emission reductions. These curves are shown in Figure 4-4. In LA, these nation-wide benefits increase to \$-386,000 tonne^{-1} at 20%, \$-186,700 tonne^{-1} at 40%, \$228,700 tonne^{-1} at 60%, and \$790,700 tonne^{-1} at 80% abatement. This shift from disbenefits to benefits is shown in the five spatial plots in the first column in Figure 4-3 (LA – National MB, Figure 4-3a-e). Similarly, in NY at baseline emission levels, the marginal benefits are initially negative, starting at \$-11,000 tonne^{-1} and increase to \$16,900 tonne^{-1} at 20%, \$49,000 tonne^{-1} at 40%, \$82,000 tonne^{-1} at 60%, and \$127,900 tonne^{-1} at 80% (NY – National MB, Figure 4-3a-e).

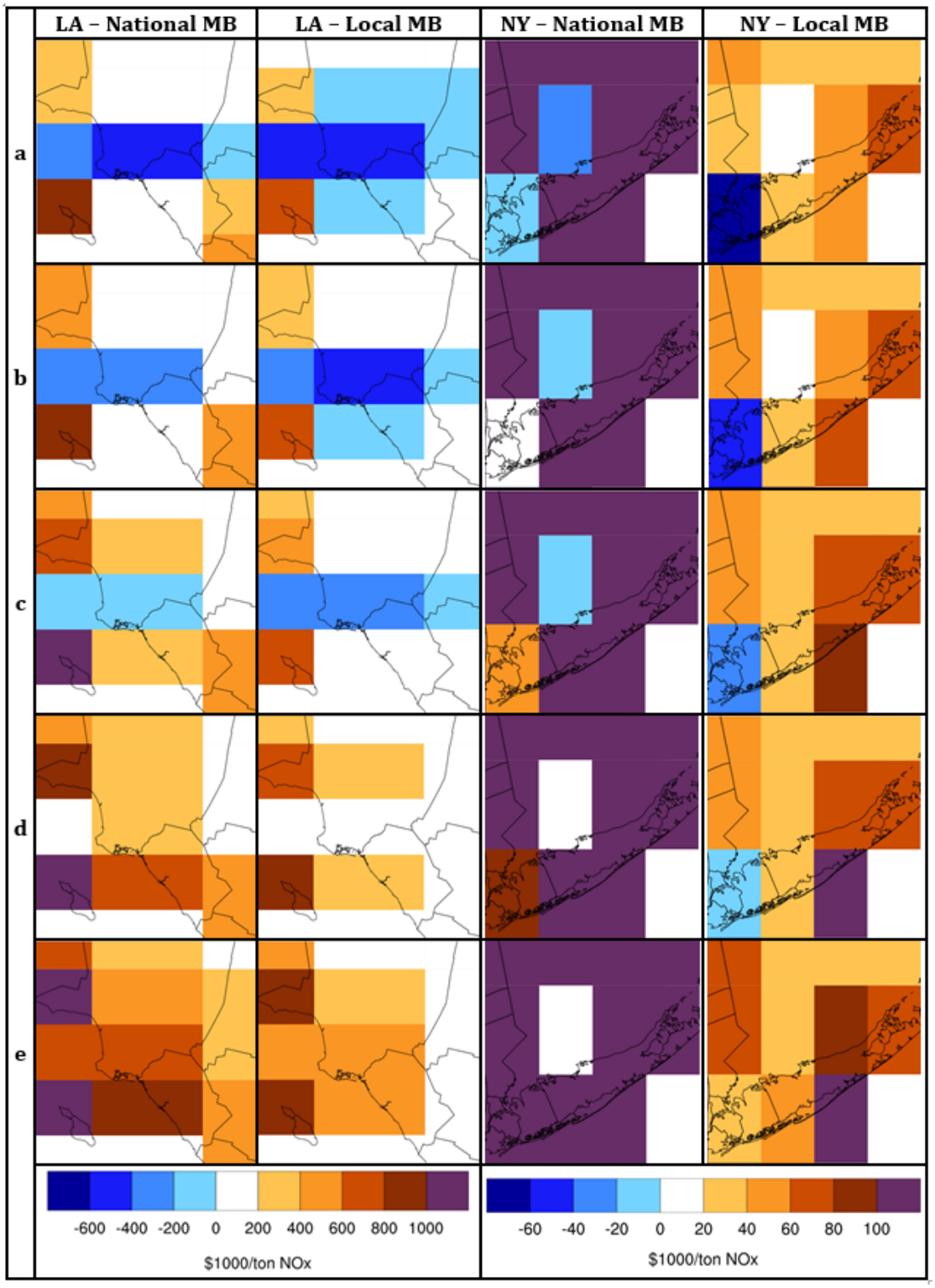

Figure 4-3: Spatial plots showing O_3-based marginal benefits for regional abatement (Case 1) in Los Angeles and New York at (a) 0%, (b) 20%, (c) 40%, (d) 60%, and (e) 80% abatement levels, showing both the national and local health benefit of removing 1 tonne of NO_X.

The local benefits in LA and NY are shown in Figure 4-3a-e in columns 2 and 4 (LA – Local MB & NY – Local MB). At baseline emission levels, the local benefit is more negative than the national benefit for both NY and LA, indicating that at the same emission level, a reduction in NO_X emissions by 1 tonne would cause more damage locally than nationally, due to the availability of NO_X in each location. This is expected as the NOx-inhibited regime leading to disbenefits is present in urban areas, whereas further downwind the urban plume is more likely to have shifted to a NOx-limited regime with positive O3 to NOx sensitivity. At baseline emission levels, the local marginal benefit in LA is \$-508,800 tonne^{-1} and increases to \$-415,100 tonne^{-1} at 20%, \$-245,000 tonne^{-1} at 40%, \$118,000 tonne^{-1} at 60%, and \$596,400 tonne^{-1} at 80%. In comparison, local marginal benefits in NY start at \$-78,500 tonne^{-1} and increase to \$-56,900 tonne^{-1} at 20%, \$-32,300 tonne^{-1} at 40%, \$-7,100 tonne^{-1} at 60%, and \$29,300 tonne^{-1} at 80%.

The tipping point has been defined as the point at which the region shifts from NO_X-inhibited to NO_X-limited and the health outcome shifts from a disbenefit to benefit. Our results show that the TP for a single location occurs at separate levels of abatement for national and local marginal benefit curves. This point is clearly identified in Figure 4-4 for each location. For example, in LA the nation-wide benefits begin at 50% abatement (the TP); however, this would still result in local disbenefits until 54% abatement. Likewise, in NY, as nation-wide benefits begin at 8% and local benefits do not appear until 64% abatement. This substantial difference emphasizes important policy implications when considering the optimal emission reduction point and strategy.

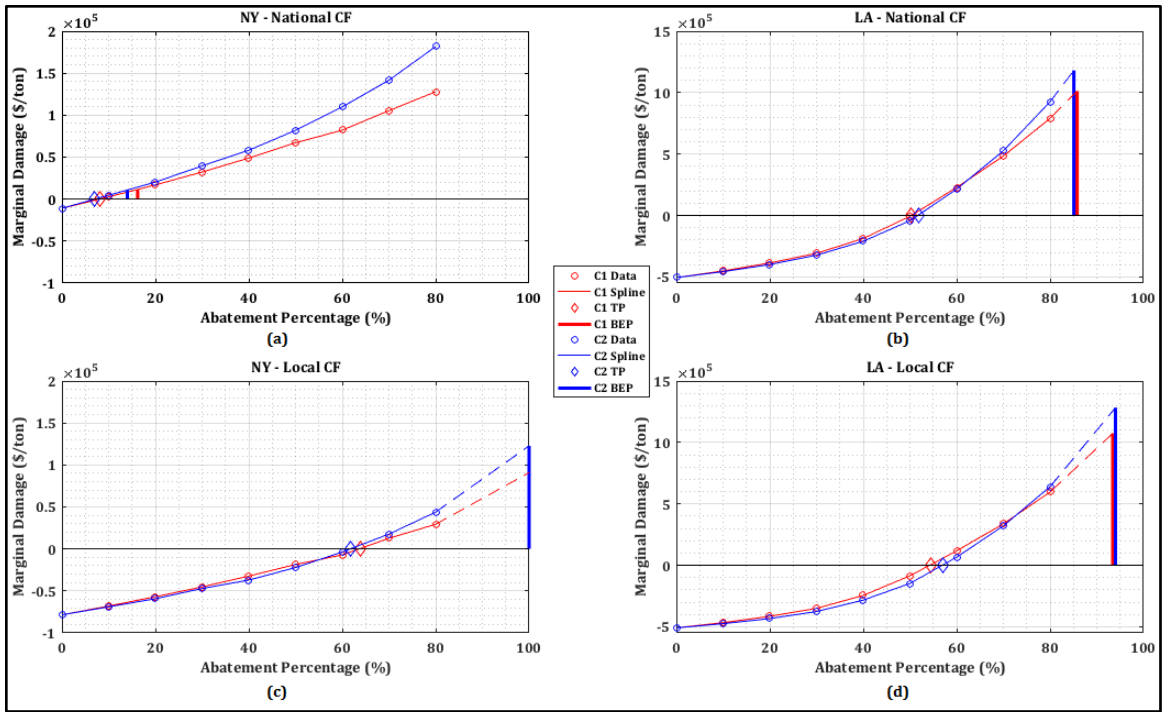

Figure 4-4: Marginal damage curves for 0-100% abatement of anthropogenic emissions (results of Cases 1 and 2). The plots show the location of the Tipping Point and the Break-Even Point of both nation-wide marginal benefits in (a) NY and (b) LA as well as local benefits in (c) NY and (d) LA.

The break-even point is the point at which cumulative benefits compensate immediate disbenefits from emission reductions (Figure 4-4), i.e. the abatement point where the area under the MB curve on the negative and positive sides are equal. This point also exists at separate levels of abatement for national and local marginal benefits. In LA, the national benefits break even at a BEP of 86% abatement and local benefits reach this point at 93% abatement. In NY, national benefits reach this point at 16% abatement, whereas local benefits do not break even at 100% abatement. It is important to note that the break-even point does not accurately represent the actual point where benefits make up for disbenefits, as the exact location of that point will depend at the pace of abatement. BEP estimates represent the overall impact assuming the years spent in the NOx-inhibited

(disbenefit) and NOx-limited (benefit) regimes are equal, i.e. a consistent abatement pace. In reality, in years after the shift to NOx-limited regime, the benefits will continue to accumulate resulting in overall net benefits in the long-term.

4.3.3 Case 2: National Abatement

In this case, we analyze the sensitivity of the model to national emission perturbations in 10% intervals in all locations. This is an important consideration as the chemical regime in each region/city, it is also impacted by the national changes in emissions that the country experiences. As a result, the marginal benefit curves shown in Figure 4-4 would be different under nation-wide abatement scenarios. Figure 4-5 contains spatial plots over the U.S. at 0%, 20%, 40%, 60%, and 80% intervals. The five figures show the progression of benefits with more aggressive abatement. Positive benefits exist across the domain, especially in areas with low NO_X concentrations; however, disbenefits can be seen localized in urban areas (city centers). A comparison of Figure 4-5(a) and Figure 4-5(e) shows that with progressive emission abatement, we see these negatives disappear and become positive in most locations (as in each perturbation, NO_X availability decreases and there is less NO_X titration).

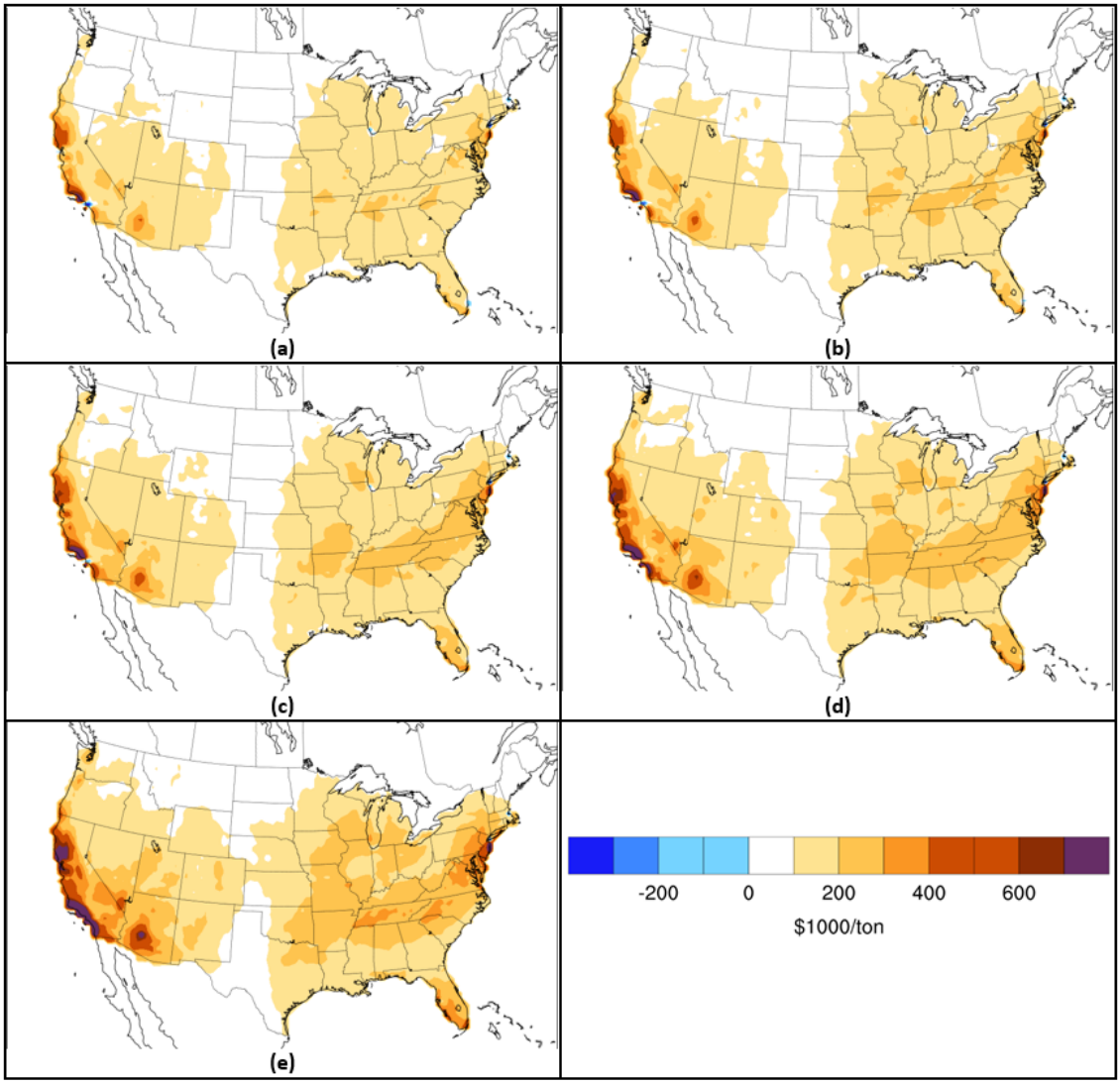

Figure 4-5: Spatial plots showing the results of Case 2 (National abatement) at (a) 0%, (b) 20%, (c) 40%, (d) 60%, and (e) 80% abatement levels.

At baseline levels, the national benefits of emission reductions in LA and NY begin at the same value as the base case, \$-505,000 tonne^{-1} in LA and \$-11,000 tonne^{-1} in NY. In LA, these benefits increase to \$-399,000 tonne^{-1} at 20%, \$-209,400 tonne^{-1} at 40%, \$214,500 tonne^{-1} at 60%, and ultimately \$925,800 tonne^{-1} at 80% with nation-wide emission controls. In NY, the benefits are \$20,200 tonne^{-1} at 20%, \$58,200 tonne^{-1} at 40%,

$109,700 tonne^{-1} at 60%, and $181,700 tonne^{-1} at 80%. Figure 4-4 contains the complete set of curves for 0-80% abatement.

Comparing these results with Case 1, the benefit of reducing emissions over the entire domain by 80% would achieve additional benefits of $135,100/ton in local emission controls in LA, as the maximum benefit in LA at 80% is $925,800, whereas the national benefit of regional abatement is $790,700/ton. In New York, additional benefits are found to be as high as $53,800/ton with national abatement compared to regional abatement.

The local benefits in LA follow a similar trend, beginning at $-508,800 tonne^{-1} and increasing to $-433,600 tonne^{-1} at 20%, $-283,000 tonne^{-1} at 40%, $66,200 tonne^{-1} at 60%, and $640,000 tonne^{-1} at 80% abatement. In NY, benefits start at $-78,500 tonne^{-1} at baseline and increase to $-59,300 tonne^{-1} at 20%, $-37,100 tonne^{-1} at 40%, $-3,600 tonne^{-1} at 60% and $43,800 tonne^{-1} at 80% abatement.

A comparison of these results with Case 1 suggests that the benefit of reducing emissions over the entire domain by 80%, would achieve additional local benefits of $43,600/ton in LA. As the maximum benefit at 80% is $640,000/ton, whereas the local benefit of regional abatement is $596,400/ton. In New York, additional benefits are found to be as high as $14,500/ton with national abatement compared to regional abatement ($43,800/ton vs. $29,300/ton).

In Case 2, the results again show that the TP for a single location occurs at separate levels of abatement for national and local marginal benefit curves. In LA, the nation-wide benefits begin at 52% abatement (the TP); however, this would still result in local disbenefits until 57% abatement. Similarly, in NY, nation-wide benefits begin at 7% and local benefits do not appear until 62% abatement. In LA, the national benefits break even at a BEP of 85% abatement and local benefits reach this point at 94% abatement. In NY, national benefits reach this point at 14% abatement, whereas local benefits do not break even until after 100% abatement.

4.3.4 Overarching Trends in National versus Local Emission Abatement

The results presented in this study stress the substantial difference in health outcomes between policies that account for local versus national emission control strategies and demonstrate the importance of considering the local implications of an emission control strategy for areas that are NO_X-inhibited, as the immediate health damage may be underestimated in a nationalized approach.

The results are consistent with the findings of compounding benefits in previous studies, showing that each additional tonne of NO_X control ensues a larger benefit than the previous tonne. Our findings also suggest that there are substantial benefits to urban NO_X control for NO_X -rich cities, where emission reductions appear unfavourable in the short term.

With the upward sloping MB curves, we show that compounding benefits of NO_X control will eventually compensate immediate disbenefits in heavily polluted urban areas. Our results also suggest that both national and local emission control policies are effective ways to mitigate health damage due to NO_X and O_3 exposure in both LA and NY. The results of the BPT curves in both NY and LA suggest that national emission abatement yields the largest benefits to both national and regional domains; however, it appears that targeted emission control policies that are specific to heavily polluted, urban areas can still result in up to 70% of these benefits in the NY region, and 80% of these benefits in the LA region (at 80% abatement).

There's one issue that will define the contours of this century more dramatically than any other, and that is the urgent threat of a changing climate.

Barack Obama

On Climate Change

5 Exhausted with Emissions: Linking Tailpipe Emissions to Public Health

This chapter is a manuscript in preparation for publication. It consists of original research for which Angele Genereux is the main contributor.

5.1 Introduction

Although nationwide emissions of nitrogen oxides (NO_X) have declined over the past decade in both Canada and the United States, the transportation sector has continuously been the largest single sector contributing to total anthropogenic emissions. In 1999, the Government of Canada enforced more stringent emission standards for passenger vehicles, light-trucks, and heavy-duty vehicles under The Regulations Amending the On-Road Vehicle and Engine Emission Regulations and Other Regulations Made Under the Canadian Environmental Protection Act [129]. Under this act, Tier 3 emission standards are introduced for the 2017 model year, and vehicles are given a useful life estimate of 240,000 km or 15 years, whichever occurs first. This useful life estimate was increased by 150% and the limit for kilometers has increased by 125% of the original useful life estimate [129]. According to the latest Transportation in Canada report, Canadians in 2017 drove an average of 16, 509 kilometers per year – roughly 7% of their estimated useful life [130]. With new models also constantly being placed on the market, the age of vehicles can vary substantially in Canada. As these vehicles age, regular maintenance is required, and alongside the increased brake and tire fatigue, the emission control systems begin to

deteriorate, and so on. The combined effect is often the increased pollutant emissions with the age of the vehicle contributing to the overall public health burden.

Nitrogen oxides and particulate matter are emitted as primary species after fuel combustion; chronic exposure to these criteria pollutants has been linked to adverse health outcomes [48-50] – including, respiratory morbidity and mortality (with acute symptoms such as coughing, wheezing, asthma, and bronchitis). What exacerbates this problem are the side reactions that lead to a net production of ozone and secondary PM, resulting in additional health effects as the population is exposed to both primary and secondary pollution.

The monetized health impacts of air pollution exposure are often referred to as "Marginal Benefits" or "Benefit per tonne" [87]. Marginal Benefits are defined as the societal health benefit of reducing the emissions (of a given criteria pollutant) and are presented as an annual, dollar per-ton value. Alternatively, these estimates may be expressed as "Marginal Damages", referring to the damage to societal public health due to the emissions of a given criteria pollutant. Quantifying the impacts of emissions, or in this case transportation emissions on public health is not an easy task. To perform a risk (i.e. cost/benefit) assessment, spatially distributed emissions data have been used in traditional integrated assessment models to estimate the effects of air pollution on health for policy applications [121-124]. Many air quality models have been used to produce

health damage estimates due to air pollution exposure for North America, five of which are described following this.

In 2007, Muller and Mendelsohn applied an integrated assessment model to measure health damages due to air pollution in the United States [120]. Similar to previous integrated assessment models used by the US EPA, the Air Pollution Emissions Experiments and Policy Analysis Model (APEEP) was used to link emission levels to health damage estimates. To estimate marginal damages, the model uses baseline emission information to calculate pollutant concentrations, exposure levels, physical effects of pollution, as well as dollar damages for the United States. The baseline emissions are provided to the model for 10,000 sources, including a total of 6900 point sources and 3110 area sources. The baseline emissions are then replaced with a set of emissions data that has been incrementally increased, the model is run again, and the difference between the two scenarios is interpreted as the marginal damage corresponding to the specified emission increment (for example, the authors use an increment in emissions of 1 tonne). The authors report gross annual damage estimates (summation of damages from all sources) of between $71 billion to $277 billion per year. Although the model has been used in subsequent analyses [107, 131, 132], the main limitation with these estimates is that the model is a Gaussian dispersion-based model that does not account for secondary pollutant interactions in the atmosphere.

Heo et al. (2016) studied the public health costs of exposure to inorganic PM for metropolitan areas of the United States using a new method called Estimating Air pollution Social Impacts Using Regression (EASIUR) [134]. The EASIUR model was developed as an alternative to other CTM methods for estimating marginal societal costs. The authors report that the EASIUR model estimates these costs using parameterizations which provide high spatial resolution information with while being less computationally expensive as more traditional techniques [105, 112, 126, 133]. All sources of inorganic $PM_{2.5}$ for 14 Metropolitan Statistical Areas (MSA) in the United States were analyzed with the EASIUR model, and the contribution of each to the societal health burden was estimated. The results showed that up to 60-80% of the health impacts due to the emissions in a single MSA will occur outside of that MSA due to pollutant transport in the atmosphere. Again in 2016, the authors applied the same method to parameterize marginal health costs and derived models for additional pollutants, including – NH_3, NO_X, EC, and SO_2. The authors found comparable results to the performance of other CTMs, with low fractional bias [135]. The major limitation of the EASIUR model is that it is a regression-based model that relies on multiple simulations and does not fully account for pollutant interactions in the atmosphere.

In 2017, Tessum et al. published an alternative approach to estimating health impacts related to emission reductions with the Intervention Model for Air Pollution (InMAP) in the United States [136]. The model analyzes changes in yearly primary and secondary $PM_{2.5}$ concentrations and estimates the monetized health damages related to

these changes. The authors claim that the model's variable spatial resolution allows it to perform simulations, while being less computationally expensive than previous air quality models. In 2018, Thakrar et al. applied the InMAP model to estimate the increase in mortality due to long-term exposure to both primary emissions of $PM_{2.5}$ (direct), as well as secondary $PM_{2.5}$ emissions formed from precursors such as NO_X, sulfur oxides (SO_X), ammonia (NH_3), and VOCs [137]. The area-weighted, monetized health impact due to emissions varies by location and is expressed in terms of dollar per megagram (10^6 g) of switchgrass, a vegetative plant similar to hay. These estimates range from $3.71/Mg of switchgrass for Maine to a maximum benefit of $82.60/Mg in Pennsylvania. The domain-average health benefit of reduced exposure to $PM_{2.5}$ emissions was reported to be $45/Mg of switchgrass.

Fann et al. (2009) also used the CMAQ model to estimate the health benefits related to changes in emissions of $PM_{2.5}$. As the constituents of $PM_{2.5}$ vary by location and source type, the authors implement 12 combinations of sources for nine urban areas and one nation-wide area to estimate the significance of both location and source type on marginal damage estimates. The marginal benefits of reducing emissions of $PM_{2.5}$ was determined to be most significant for area source emissions, with an average national health benefit of reducing $PM_{2.5}$ emissions of $720,000/ton. In individual urban locations, the health benefits were found to increase beyond this, reportedly reaching up to $2,500,000/ton in Phoenix. The mobile source emissions were found to produce the second largest health benefits, with a domain-wide average of $550,000/ton and benefits

of up to $1,700,000/ton in Phoenix. The authors conclude that the two sources showing the highest marginal damage results suggest a correlation between $PM_{2.5}$ emissions and population centers [126]. Following this, Fann et al. (2012) used a simplified air quality model to estimate the health benefits and impacts of reduced emissions of $PM_{2.5}$. The authors use the Comprehensive Air Quality Model with Extensions (CAMx) to characterize the effect of reducing primary and precursor $PM_{2.5}$ emissions from 17 sectors across the US on marginal damage estimates. The authors found that the benefit of reducing $PM_{2.5}$ emissions is between $1300/ton NO_X ($PM_{2.5}$ precursor) for Ocean-Going Vessels, up to $450,000/ton $PM_{2.5}$ (primary) emissions from Iron and Steel facilities [133]. The main limitation in this work is that the authors model specific urban areas and geographic regions, and therefore the estimates provided do not represent the full nation.

In 2013, Pappin et al. used the CMAQ model to trace the sensitivities of reducing emissions in each location on domain-wide, non-accidental mortality in Canada and the United States [87]. The authors use the CMAQ model to quantify the public health benefits of reduced short-term exposure to NO_2 and O_3 for Canada, and O_3 for the United States. The health effects of NO_2 are not reported for the United States due to differences in national epidemiologic studies and disagreements over the health effects of NO_2. For a 10% reduction in NO_X emissions, the authors find significant total health benefits for Canada through both NO_2 and O_3, including up to $253,000/day in Hamilton, Ontario. In the United States, a 10% reduction in NO_X emissions was reported to result in benefits of up to $181,000/day in Atlanta, Georgia. The authors reported health damages on a per-

vehicle basis for one hypothetical, average vehicle existing in Canada and found estimates of $440/year in Mississauga (Ontario), $450/year in Vancouver, and $770/year in Montreal.

Following this, Pappin et al. (2015) applied a similar methodology to investigate how the marginal benefit estimates change with progressive NO_X emission abatement in the United States [97]. The authors used the CMAQ model and its adjoint counterpart (explained further), to investigate the sensitivity of short-term ozone exposure on domain-wide mortality ($/ton). A reduction of NO_X emissions by 1 tonne was reported to result in marginal benefits of up to $87,000/ton for mobile sources and up to $39,000/ton for point sources. Although these estimates are for averted NO_2 and O_3 exposure, the authors conclude that the nonlinear model would likely extend for $PM_{2.5}$ as well.

A later analysis was conducted by Pappin et al. (2016) on the marginal benefits of progressive NO_X emission abatement in Canada [98]. The authors investigated the effect of reducing NO_X emissions on chronic exposure to 8-hour ozone and 24-hour NO_2. The authors used the CMAQ model to analyze multiple abatement scenarios on domain-wide mortality. Its adjoint counterpart was used to investigate the effect of using an exposure model that is linear in concentration and compare these with a log-linear exposure model for each pollutant. The linear exposure model resulted in marginal damages of up to $270,000/ton in Ottawa, $460,000/ton in Vancouver, and $840,000/ton in Toronto. The log-linear exposure model showed a heightened sensitivity for populations in cleaner

environments, with marginal damage estimates of up to $500,000/ton in Ottawa, $510,000/ton in Vancouver, and $650,000/ton in Toronto. The authors report that the marginal benefits for Canada increase with increased emission abatement, regardless of the exposure model used.

In this paper, a new, innovative approach is used to estimate the health impact of individual vehicles on an age-segregated basis within each fleet in both Canada and the United States. We use the US EPA's CMAQ model to track the movement of air pollutants from a source over space and time. To determine the sensitivity of reducing emissions on a per-ton basis (Marginal Benefits), we use its adjoint counterpart to trace the influence back to the original emitters. The adjoint approach is particularly attractive in this case, as it provides spatially resolved data for transportation impact.

We follow the approach in Pappin et al. (2013); however, as the age of vehicles (as well as their respective pollutant emissions) varies drastically across Canada and the United States, we extend the granularity of their representation of the fleet. We aim to address the limitation in the past work by proposing a new modeling methodology to segregate the emissions by vehicle type and vintage/age. To do this, we use emission factor models with domain-specific information for both Canada and the United States (MOBILE6.2C and MOVES, respectively) to estimate individual emission factors for each region, vehicle type, and vehicle age. This distinction of cost on a per-vehicle basis will allow new insight for policy makers in targeting the transportation sector in upcoming emission control policies.

5.2 Data and Methodology

5.2.1 Adjoint Formulation

In this analysis, we quantify the public health impacts of emission reductions of both nitrogen oxides (NO_X) and particulate matter less than 2.5 microns ($PM_{2.5}$) due to chronic exposure to pollutants in Canada and the United States. Following the combustion of fuel, nitrogen oxides are emitted as NO and NO_2. In Canada, nitrogen dioxide (NO_2) is considered to be damaging to human health and the environment. These species are reactive and in combination with volatile organic matter (VOCs) and/or NH_3, often lead to the formation of additional secondary pollutants (e.g. secondary particulate matter, ground level ozone) [138]. In the United States; however, NO_2 is not considered to be directly linked to mortality. Therefore, the estimates for Canada include the benefit for averted chronic exposure to NO_2, O_3, and $PM_{2.5}$, and solely O_3 and $PM_{2.5}$ for the United States. We present the results separately due to differences in epidemiology and health, economics, pollutant emissions, and transportation classifications.

5.2.1.1 Canada

In an effort to estimate the impact of individual sources on Canadian public health, we use an approach incorporating the adjoint of the US EPA's Community Multiscale Air Quality model, CMAQ [107]. A detailed description of both the adjoint model development and more recent applications are available elsewhere [87, 97, 98, 105, 110]. We estimate Marginal Benefits for mobile sources using the CMAQ version 5 with the CB05 chemical mechanism. We run the model at a 36-km horizontal grid resolution for the full domain

encompassing Canada (shown in Figure 5-1), with 34 vertical layers. Meteorological information is processed with the Meteorology Chemistry Interface Processor (MCIP) within the Weather Research and Forecasting (WRF) model [100]. We use the Sparse Matrix Operator Kernel Emissions (SMOKE) model [103] to process emissions from the National Emission Inventory (NEI) for the US and the National Pollutant Release Inventory (NPRI) for Canada.

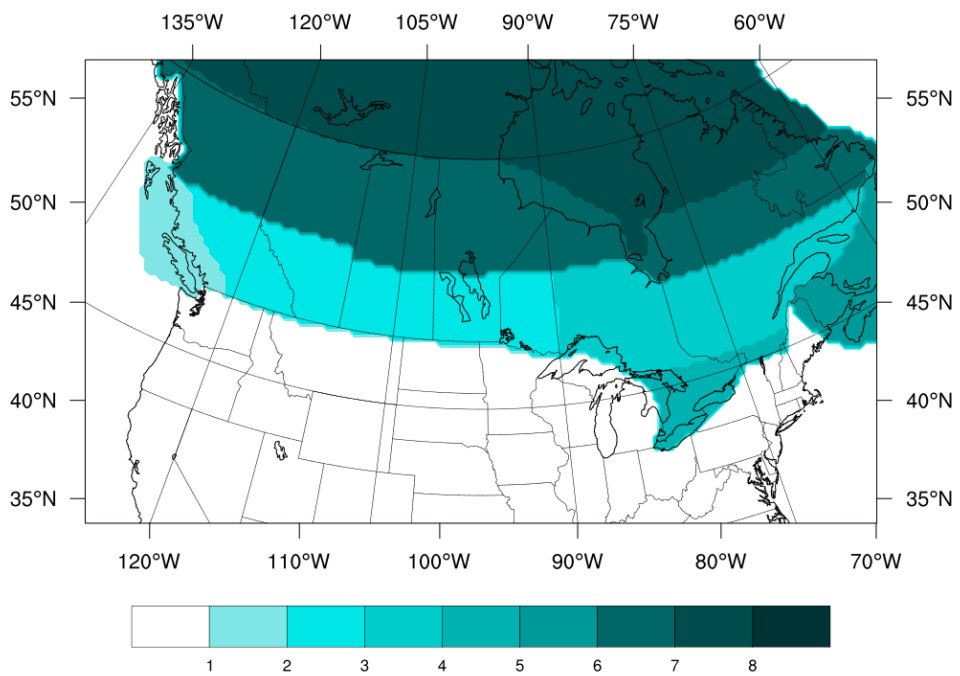

Figure 5-1: The domain used to model the 8 individual subregions within Canada at a 36-km horizontal resolution, with region numbers corresponding to those presented in Table 5-1.

To determine the societal public health benefits across Canada, an adjoint cost function must be defined (and must exist as a function of concentration). Estimates for Canada are generated using a joint (3-pollutant) model that accounts for exposure through co-pollutant emissions of NO_2, O_3, and $PM_{2.5}$ simultaneously. The multi-pollutant model

(presented in Eq. 5-1) is the basis for a scalar adjoint cost function that is non-linear in concentration for both NO_2 and $PM_{2.5}$. The equation has been comprised with information from multiple disciplines, including economics, epidemiology and air quality. Detailed information on the derivation of the cost function is available elsewhere [87, 97, 110]. The adjoint cost function and forcing equation have been adapted from a recent publication by Pappin et al. in 2016 [98].

$$\Delta M = M_0 \cdot P \cdot V_{SL} \cdot \left(1 - \frac{1}{RR}\right) \qquad \text{Eq. 5-1}$$

Here, ΔM is the monetized mortality across Canada that can be attributed to emissions of a specific pollutant for the year 2010. To achieve the monetization of benefits for each location, we incorporate a Value of Statistical Life, V_{SL} and the annual, non-accidental mortality rate, $M_{0,w}$. We incorporate estimates at county-level for the all-age population, P, for each location. The equation has been updated to include RR, or relative risk from exposure to an additional unit of concentrations and incorporate pollutant-specific effect estimates, β, for O_3, NO_2, and $PM_{2.5}$, based on [117] and [118].

5.2.1.2 United States

The US EPA's CMAQ Model [107] is used in this analysis to track pollutant concentrations over space and time at baseline emission levels to generate a set of concentration outputs. These output files are used by the CMAQ-adjoint model to estimate the health sensitivities across the full domain. A detailed description of the adjoint model

is provided elsewhere [105, 110]. The domain is set for the continental US (CONUS), modelled over 34 vertical layers at a 36-km horizontal grid resolution. Due to the complexity of the air quality model and the computational requirements, we limit the model run to one month of the ozone season (July 1 to 31, 2007), and scale the results to yearly estimates. Emission inventories were downloaded from the National Emission Inventory (NEI) for the US domain. The emission inventories are distributed over the continental US with the Sparse Matrix Operator Kernel Emissions (SMOKE) model [103]. We supply the model with meteorological inputs from the Weather and Research Forecasting model (WRF) [100].

We apply a linear epidemiological model in the adjoint cost function, as defined in Eq. 5-2. More information on the derivation of this function is available elsewhere [98].

$$J = V_{SL} \sum_{W} M_{0,W} P_W \left(1 - e^{-\beta \overline{C_W}}\right) \qquad \text{Eq. 5-2}$$

Here, in Eq. 5-2, J is the monetized number of non-accidental mortalities associated with O_3 (or $PM_{2.5}$) exposure in the US on a per-year basis. A value of statistical life, V_{SL}, is estimated based on a populations' willingness to pay to avoid mortality and is used to monetize the health sensitivities to pollution. This value varies by country and is assigned to be $7.9 million in the U.S [114]. M is the non-accidental mortality rate in grid cell w, for populations, P_W, between the ages of 33-99. The equation is written separately for O_3 and $PM_{2.5}$ as we apply separate concentration response factors, β. We apply a β of 3.92 x 10^{-3}

for chronic exposure to 1-hour O_3 in the US that is based on Cox proportional-hazard models [74] and apply a pollutant-specific risk estimates, β, for chronic exposure to $PM_{2.5}$. The adjoint model is driven using the forcing term, as shown in equation Eq. 5-3 [98].

$$\varphi = \frac{V_{SL} M_{O,W} P_W \beta e^{-\beta \overline{C_W}}}{\bar{t} n} \qquad \text{Eq. 5-3}$$

Here, φ is the adjoint forcing term, n is the number of simulation days, t is the number of hours in the exposure metric (1 for 1-hour ozone), C_w is the 1-hour max ozone concentration at each location, w, and all other variables are as defined above.

5.2.2 Emission Factor Estimation

Vehicle emission simulator models are important tools for regulators and researchers, as countless operating scenarios can be studied without the need for time-consuming and prohibitively expensive physical testing. With the use of traffic data, vehicle registration information, and environmental records, large emission datasets can be generated to provide information about a region's vehicle fleet.

5.2.2.1 Canada

Emission factors for the Canadian vehicle fleet are estimated with MOBILE6.2C, a mobile-source emission factor model that was adapted for Canada in 2003 by Environment Canada to provide a benchmark for modeling of on-road emissions [139]. Previous versions

of the MOBILE model have been published by the US EPA and were adopted for Canada shortly after [139]. The model is capable of estimating gram per mile emission factors for emissions of 21 pollutants and sub-species in total – including, sulfur oxides (SO_X), nitrogen oxides (NO_X), primary emissions of particulate matter ($PM_{2.5}$ & PM_{10}), volatile organic compounds (VOCs), carbon monoxide (CO), ammonia (NH_3), carbon dioxide (CO_2), as well as air toxics such as benzene, 1,3-butadiene, acetaldehyde, formaldehyde, and acrolein [139]. In 2003, the MOBILE6.2C model was the best vehicle emission simulator model available in Canada. The model has since been replaced with the Motor Vehicle Emissions Simulator (MOVES) which is not yet publicly available. In 2011, Environment Canada submitted a Request for Proposal with the objective of updating the MOVES underlying datasets to more accurately reflect Canadian conditions [140]; however, neither the input data nor the revised model have been made publicly available to this date.

Although the MOBILE6.2C model can produce estimates for ten separate emission types (running, start, hot soak, diurnal, resting, run loss, crankcase, refueling, brake wear, and tire wear [141]), this investigation considers only the exhaust running emissions. Therefore, the health damage estimates for populations exposed to vehicle exhaust (presented in the results section of this paper), do not fully quantify (and may underestimate) the impact of vehicles on public health as only a fraction of the lifecycle emissions of an individual vehicle are accounted for.

Input Data and model runs

The MOBILE6.2C model is used to predict running emission factors in grams of pollutant per distance travelled (g/mi) for NO_X, $PM_{2.5}$, NH_3, SO_2, Pb, and SO_4. The vehicle emission factors produced by the model depend on multiple input factors, which may be user-defined or left as national default values.

The updates to MOBILE6.2C in 2003 included replacing the US national-level default data included with the software with new defaults that are a better representation of the Canadian climate and vehicle fleet [142]. A list of input parameters available for modification are summarized in the MOBILE6.2C Canadian Supplemental Users' Guide [139]; however, users are only required to provide the input data including the calendar year, minimum and maximum daily temperature, and fuel volatility for national-level runs (at a minimum). The amount of information required by the model depends on the specific modelling needs of the user. It is suggested that the default data be replaced for city-based or small-scale simulations (where national averages may not apply) to achieve the best emission factor estimates (a roadway, intersection, city, etc); however, the use of default information for the majority of the parameters is adequate for national or regional level simulations [139, 142]. Therefore, the national level defaults were used for all except the following input parameters: calendar year, evaluation month, hourly temperatures, fuel volatility (RVP), diesel sulfur content, relative humidity, barometric pressure, and particle size cutoffs.

Therefore, to increase the accuracy of the emission factor model, the single domain chosen to represent Canada is sub-divided into 8 regions based on the division of fuel RVP values outlined in the National Automotive Gasoline Standards, published by the Government of Canada in 2016 [143]. The regions are shown in Figure 5-1. For each region, the city with the largest population (based on population census data from [144]) was chosen to be representative of each domain. The chosen cities and their respective region IDs (corresponding to those depicted in Figure 5-1) are presented in Table 5-1.

Table 5-1: RVP values and corresponding cities in all 8 regions

Region	ID	City	RVP (psi)	Barometric Pressure (mm Hg)	Diesel Sulfur Content (ppm)
A	1	Vancouver	6.530	30.14	15.00
B	2	Calgary	7.034	26.44	15.00
C	3	Quebec City	8.775	29.55	15.00
D	4	Toronto	7.760	29.36	15.00
E	5	Halifax	7.760	29.39	15.00
F	6	Fort McMurray	8.775	28.55	15.00
G	7	Whitehorse	7.760	27.46	15.00
H	8	Yellowknife	7.760	29.01	15.00

The fuel RVP [143], barometric pressure [145], and diesel sulfur content [146] for each region is listed in Table 5-1, along with the respective values used for each. Region-specific information was also supplied for the hourly temperatures and hourly relative humidity (see Table 5-2) based on daily observations from climate data published by the government of Canada at each city's respective airport weather station for July 15th-16th, 2017 [145]. The evaluation month and calendar year were set to be July of 2017.

Table 5-2: Hourly temperature (°F) and relative humidity (%) values supplied to MOBILE6.2C for Vancouver, Toronto, Calgary, Quebec City, Halifax, Fort McMurray Whitehorse, and Yellowknife

Hour	Hourly Temperature (°F)								Relative Humidity, RH (%)							
	VN	TO	FM	WH	YK	CG	HX	QC	VN	TO	FM	WH	YK	CG	HX	QC
1	58.82	65.48	53.42	49.82	61.7	58.1	55.4	62.06	79	78	94	78	63	71	85	89
2	61.34	66.92	55.76	51.98	65.48	61.34	56.66	61.7	70	73	93	75	56	65	88	93
3	63.32	69.08	60.98	52.34	67.64	64.76	57.38	61.52	62	67	83	74	53	52	88	95
4	65.48	70.7	67.28	53.96	69.8	67.82	61.7	64.04	55	63	71	72	46	43	84	90
5	67.28	73.58	70.16	56.12	70.52	70.52	66.38	66.2	52	57	69	67	43	44	72	85
6	69.62	76.1	73.94	56.66	70.88	71.42	69.08	67.46	47	53	54	68	45	43	66	82
7	71.78	78.08	75.2	56.48	70.88	73.76	72.5	70.34	49	49	50	67	43	42	57	80
8	73.22	78.62	79.34	57.02	70.7	75.2	72.68	72.68	43	47	47	71	44	43	59	74
9	73.4	80.6	81.14	57.02	67.64	77	73.22	74.66	44	49	44	71	59	42	65	69
10	74.12	78.98	84.02	55.04	67.46	77.9	72.32	75.2	44	46	41	79	62	41	67	67
11	74.12	81.68	87.44	54.68	66.38	77.9	72.14	76.1	47	44	36	85	67	34	67	64
12	71.24	80.6	89.24	53.42	68.36	79.88	70.16	76.1	50	45	31	90	58	32	72	66
13	69.44	79.88	90.14	52.7	71.78	80.42	68.9	75.2	54	47	26	92	49	33	75	69
14	66.2	77.72	88.7	52.16	70.88	77.72	64.58	72.68	70	44	28	93	50	41	85	73
15	65.12	74.3	85.1	51.44	68	74.3	61.88	70.16	66	40	34	94	61	49	92	80
16	64.4	69.98	80.78	50.72	63.86	72.32	61.16	67.64	53	43	42	94	74	49	96	87
17	63.5	68	75.56	50.36	60.26	70.16	60.8	67.28	56	50	53	94	83	51	97	87
18	62.42	67.46	73.58	49.82	58.1	66.92	60.62	65.48	59	51	49	93	89	56	98	92
19	61.16	65.66	72.32	49.46	57.02	64.22	60.62	64.22	63	54	62	93	92	61	99	93
20	60.08	66.38	70.16	49.46	57.02	62.42	60.62	62.96	67	54	66	92	93	65	100	94
21	60.8	63.68	68.54	48.92	55.76	61.88	60.44	60.26	64	72	70	95	95	67	100	95
22	60.8	64.58	69.44	48.92	54.86	57.56	60.62	61.34	65	76	56	93	98	78	100	96
23	60.26	63.32	65.48	48.38	55.22	60.44	60.98	64.04	65	84	66	95	96	71	100	95
24	60.62	63.68	63.32	48.2	58.46	56.3	61.16	64.58	68	87	73	94	91	78	100	92

The output from the MOBILE6.2C model contains emission factors, generated for each of the 28 vehicle types outlined in Table 5-3 [141]. The emission factors are grouped by facility type and are averaged into age bins – including 0-2 years, 3-5 years, 6-9 years, 10-13 years, and >14 years, as defined in previous Transportation in Canada reports [147]. See the supplementary material for a description of the calculation process.

Table 5-3: The 28 vehicle classes (along with the abbreviation and long description) as defined in the MOBILE6.2C model (modified from [141])

Number	Abbreviation	Description
1	LDGV	Light-Duty Gasoline Vehicles (Passenger Cars)
2	LDGT1	Light-Duty Gasoline Trucks 1
3	LDGT2	Light-Duty Gasoline Trucks 2
4	LDGT3	Light-Duty Gasoline Trucks 3
5	LDGT4	Light-Duty Gasoline Trucks 4
6	HDGV2b	Heavy-Duty Gasoline Vehicles 2b
7	HDGV3	Heavy-Duty Gasoline Vehicles 3
8	HDGV4	Heavy-Duty Gasoline Vehicles 4
9	HDGV5	Heavy-Duty Gasoline Vehicles 5
10	HDGV6	Heavy-Duty Gasoline Vehicles 6
11	HDGV7	Heavy-Duty Gasoline Vehicles 7
12	HDGV8a	Heavy-Duty Gasoline Vehicles 8a
13	HDGV8b	Heavy-Duty Gasoline Vehicles 8b
14	LDDV	Light-Duty Diesel Vehicles (Passenger Cars)
15	LDDT12	Light-Duty Diesel Trucks
16	HDDV2b	Heavy-Duty Diesel Vehicles 2b
17	HDDV3	Heavy-Duty Diesel Vehicles 3
18	HDDV4	Heavy-Duty Diesel Vehicles 4
19	HDDV5	Heavy-Duty Diesel Vehicles 5
20	HDDV6	Heavy-Duty Diesel Vehicles 6
21	HDDV7	Heavy-Duty Diesel Vehicles 7
22	HDDV8a	Heavy-Duty Diesel Vehicles 8a
23	HDDV8b	Heavy-Duty Diesel Vehicles 8b
24	MC	Motorcycles (Gasoline)
25	HDGB	Gasoline Buses (School, Transit and Urban)
26	HDDBT	Diesel Transit and Urban Buses
27	HDDBS	Diesel School Buses
28	LDDT34	Light-Duty Diesel Trucks 3 and 4

5.2.2.2 United States

A base set of emission factors of NO_X and $PM_{2.5}$ in units of g/mi were obtained from a publication by the Argonne National Laboratory (Cai et al., 2013). The authors use the US EPA's Motor Vehicle Emission Simulator (MOVES) model [148], to estimate the lifetime mileage-weighted average emission factors in g/mi for vehicles corresponding to model years 1990 – 2020 [149]. With advancements in vehicle technology (enhanced emission control systems), and strict emission standards, data collection has improved in the United States. The MOVES model was developed by examining millions of results from light-duty vehicles. In an effort to separate the MOVES model from its MOBILE6.2 counterpart, the US EPA conducted what became known as the largest study ever conducted, considering 500 light-duty cars and trucks in Kansas City, Missouri in an effort to examine $PM_{2.5}$ emissions. When the MOBILE6.2 model was developed, there was little information available on heavy-duty diesel crankcase evaporation and extended idling. With data from the past decade, the MOVES model was updated to contain more representative information and processes (Federal Highway Administration travel data, EPA studies, vehicle surveys conducted by the census bureau, etc.), replacing the now outdated MOBILE6.2 model in the United States [148, 150].

The authors use the MOVES2010b model [151] to estimate emission factors for light-duty gasoline and diesel vehicles, light- and heavy-duty trucks, and motorcycles for carbon monoxide (CO), volatile organic compounds (VOCs), oxides of sulfur (SO_X), nitrogen oxides (NO_X), as well as PM_{10} and $PM_{2.5}$, methane (CH_4), nitrous oxide (N_2O), and carbon

dioxide (CO_2). The particulate matter emissions were specified to be disaggregated into the major components – including, organic carbon (OC), black carbon (BC), and sulfate [149]. The vehicle categories have shifted from gross vehicle weight rating (GVWR) and vehicles are classified by source/operation type in the MOVES model [152]. Although the vehicle classes have changed, the emission factors within the model are still aggregated on a class/weight basis and can be mapped back to the original 28 vehicle categories using the Source Bin Generator module within MOVES to obtain the vehicle type mapping (see the supplementary material for a detailed description of the calculation process).

Emission factors for NO_X and $PM_{2.5}$ were extracted and were disaggregated back into the original 28 vehicle types to maintain some consistency with the analysis performed for Canada. The emission factors were then grouped into the same age bins as those estimated for Canada (shown again later in Table 5-4), including – 0-3 years, 3-5 years, 6-9 years, 10-13 years, and 14+ years. The reader is encouraged to refer to the supplementary material for the emission factor tables and an overview of the grouping process.

5.2.3 Per-Vehicle Health Benefit Estimation

The per-vehicle health benefit (PVHB) can be quantified in units of dollar per vehicle per year ($/vehicle-year) or dollar per thousand miles travelled ($/$10^3$ mi). The first is a quantification of the annual contribution to mortality on a per-vehicle basis, estimated

using the average distance travelled by a vehicle of a specific age per year. The PVHB in terms of dollar per thousand miles travelled is shown in Eq. 5-4.

$$PVHB\left(\frac{\$}{10^3 mi}\right) = \frac{MB \cdot EF}{10^3} \qquad \text{Eq. 5-4}$$

The equation consists of combining marginal benefits ($/ton) with emission factors (g/mi) to estimate the benefit of reducing emissions on a vehicle-mile basis. Presenting the PVHB in terms of dollar per thousand miles may be most applicable to heavy duty vehicles, which are known to be heavy polluters. The average distance travelled by a vehicle (vehicle-miles travelled, VMT) of a given age can be incorporated into the equation to produce distributions of the per-vehicle health benefit in terms of dollar per vehicle, per year ($/vehicle-year).

5.2.3.1 Canada

A shapefile containing the new domain definitions was created using ArcMap10.5.1 software. A base map of North America was imported into ArcGIS software and the new domain boundaries were traced. A new attribute was added, and a region ID was assigned to each of the 8 subdomains. Spatial allocator version 4 was used to convert the shapefile to a gridded netcdf file [104]. The resulting file contains the respective region ID for each 36 km grid cell in a domain encompassing a total of 148 columns and 87 rows. The map is presented in Figure 5-1.

For each individual 36-km grid cell, the corresponding emission factor was multiplied with the location's corresponding marginal benefit estimate, producing ($/$10^3$ mi) estimates for all vehicle types, fuel types, and model years. To estimate the annual PVHB in terms of dollar per vehicle, VMT must be accounted for in the equation. Vehicle driving characteristics were extracted from Transportation in Canada reports [153] and are presented in Table 5-4.

Table 5-4: Domain-wide, annual vehicle-mileage estimates (in miles per vehicle per year) by age and country (estimated from [153, 154]))

Age	Vehicle-Miles Travelled, VMT $\left(\frac{\mathbf{mi}}{\mathbf{vehicle \cdot year}}\right)$	
	Canada	United States
0-2	12564	13851
3-5	11481	12042
6-9	10197	10741
10-13	7968	7401
>14	7968	7401

5.2.3.2 United States

The corresponding emission factor for each vehicle type was multiplied with the marginal benefit estimate in each 36-km grid cell across the CONUS domain, producing ($/$10^3$ mi) estimates for all vehicle types, fuel types, and model years. A conversion factor of 10^6 g per tonne was applied to satisfy the units within the calculation. To estimate the annual PVHB in terms of dollar per vehicle in the US, we supply a distribution of VMT, based on American driving characteristics (see Table 5-4) [154].

5.3 Results

5.3.1 Emission Factors: Canada

Emission factors of NO_X and $PM_{2.5}$ were estimated for all 28 vehicle classes defined in Table 5-3. The NO_X emission factor estimates are presented for light-duty gasoline vehicles in Figure 5-2(a) and for light-duty diesel vehicles in Figure 5-2(c). The emission factor estimates for LDGV average between 0.0251 gNO_X/mi within the first age bin (0-2 years) to 0.7464 gNO_X /mi within the last age bin (>14 years). LDDV estimates range from 0.0226 gNO_X/mi for new models and up to 0.2875 gNO_X/mi for vehicles aged 14 years or more.

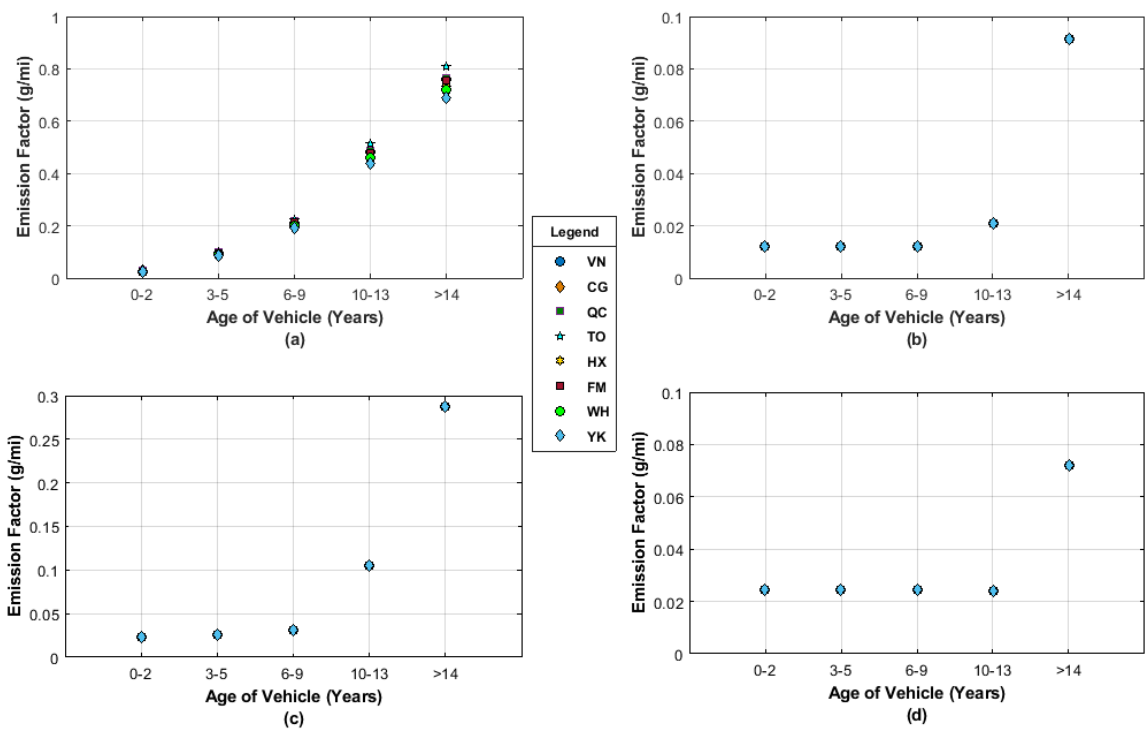

Figure 5-2: Emission factors are shown over five age bins, showing the results for light-duty gasoline (LDGV) emissions of NO_X (a) and $PM_{2.5}$ (b), as well as light-duty diesel (LDDV) emissions of NO_X (c) and $PM_{2.5}$ (d) in Canada. The emission factor results have been post-processed from MOBILE6.2C output for 8 separate regions within the domain encompassing Canada.

Emission factors of $PM_{2.5}$ are presented for LDGV in Figure 5-2(b) and for LDDV in Figure 5-2(d). The model produced emission factor estimates of between 0.0121 $gPM_{2.5}$/mi

to 0.0913 $gPM_{2.5}$/mi for light-duty gasoline vehicles in age bins 1 and 5, respectively. For light-duty diesel vehicles, emission factors are similar, with 0.0245 $gPM_{2.5}$/mi for new models up to 0.0721 $gPM_{2.5}$/mi for models aged 14 years or more.

Comparing the emission factors for all 28 vehicle types, 5 age bins for each, for 8 differing regions, and 2 pollutant types can become overwhelming. Therefore, the results presented in this section will show a comparison between the results of NO_X and $PM_{2.5}$ for vehicles aged 0-16 years within the following vehicle classifications: LDGV, LDDV, HDGV8a, and HDDV8a; however, the results for the remaining vehicle classes are given in the Supplementary Information document that accompanies this paper.

Emission factors of NO_X are presented for HDGV in Figure 5-3(a) and for HDDV in Figure 5-3(c). The emission factors of NO_X for HDGV8a average between 0.2221 gNO_X/mi for models aged 0-2 years, up to 5.8392 gNO_X/mi for vehicles aged 14-16 years. Vehicles classified as HDDV8a were estimated to emit an average of 0.5303 gNO_X/mi for newer models, and up to 10.7413 gNO_X/mi for vehicles in the last age bin. The model estimates of $PM_{2.5}$ emission factors are presented for HDGV in Figure 5-3(b) and for HDDV in Figure 5-3(d). HDGV8a emit an average of 0.0335 $gPM_{2.5}$/mi for vehicles aged 0-2 years and can emit up to 0.1849 $gPM_{2.5}$/mi (vehicles aged 14-16 years). The emission factors for HDDV8a were found to range from 0.0175 $gPM_{2.5}$/mi to 0.2507 $gPM_{2.5}$/mi for vehicles within age bins 1 and 5, respectively.

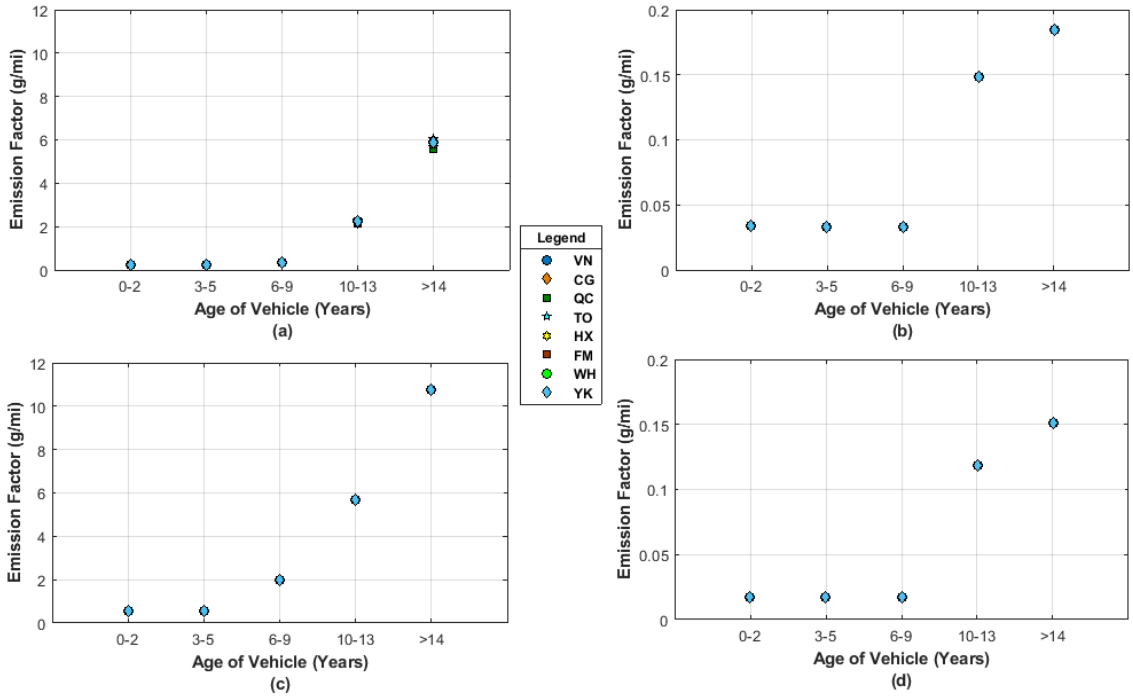

Figure 5-3: Emission factors are shown over five age bins, showing the results for heavy-duty gasoline (HDGV8a) emissions of (a) NO_X and (b) $PM_{2.5}$, as well as heavy-duty diesel (HDDV8a) emissions of (c) NO_X and (d) $PM_{2.5}$ in Canada. The emission factor results have been post-processed from MOBILE6.2C output for 8 separate regions within the domain encompassing Canada.

5.3.2 Per-Vehicle Health Benefit ($/10^3 miles): Canada

Estimates of the per-vehicle health benefit are spatially heterogeneous and show a strong sensitivity to population density and location. These benefits are reported in dollar per thousand vehicle-miles travelled and represent the domain-wide societal health benefit of reducing the distance travelled by a single vehicle in a specific location. The results for the PVHB (through NO_X) are shown for vehicle age bins 1-5 in Figure 5-4 where it is clear that the health benefits are found to maximize near population cores and decrease in rural areas. This association has been directly linked to be a function of population and exposure.

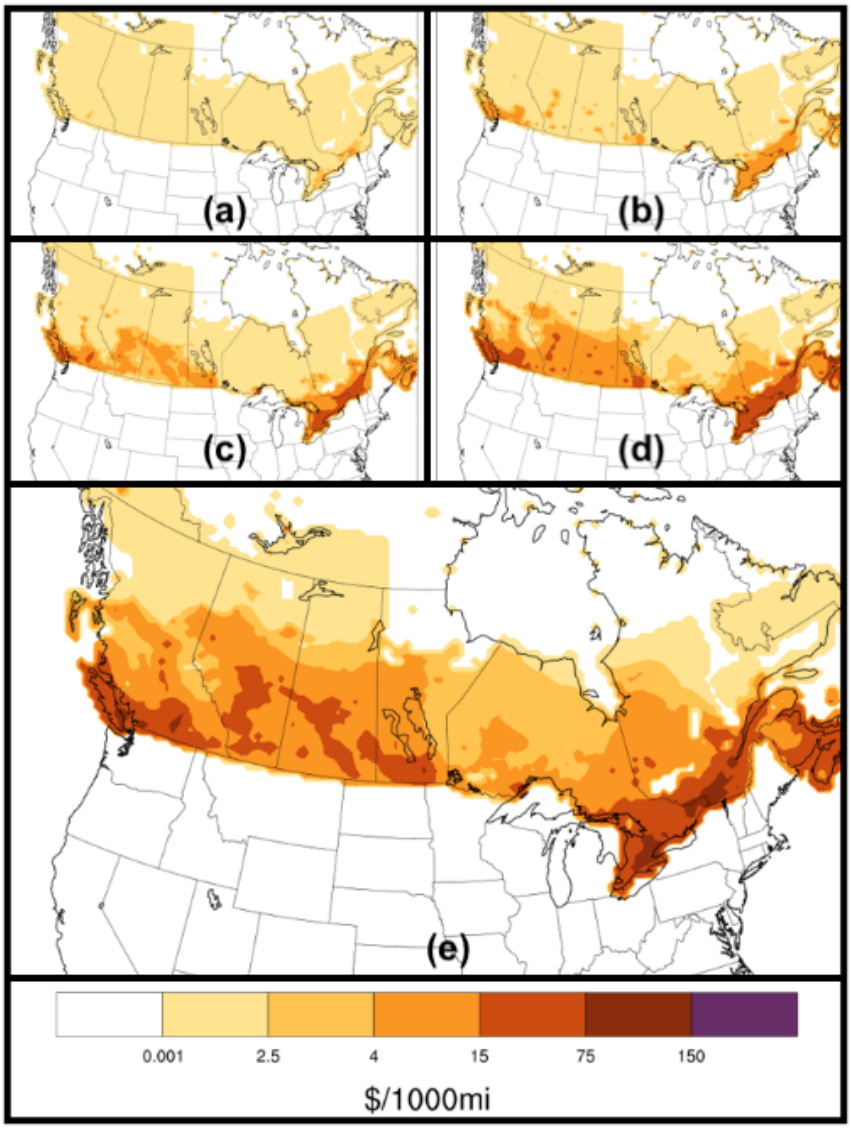

Figure 5-4: The per-vehicle health benefit results, PVHB(NO_X), are shown over five age bins, including - (a) 0-2, (b) 3-5, (c) 6-9, (d) 10-13, and (e) >14 years. The spatial variation of the PVHBs for light-duty gasoline vehicles in Canada is depicted.

For LDGV in Toronto, we find that the PVHB through NO_X increases from \$1.90/10^3mi for new models to \$52/10^3mi for vehicles aged 14-16 years. In Quebec City, the PVHB ranges from \$5/10^3mi for a new model to \$147/10^3mi for a vehicle in age bin 5. The PVHBs for LDGV through $PM_{2.5}$ are shown in Figure 5-5. We find additional benefits through $PM_{2.5}$ in Toronto of \$2.10/10^3mi to \$16/10^3 mi for vehicles in age bins 1 and 5, respectively. In Quebec City, the PVHB through $PM_{2.5}$ ranges from \$2.00 to \$15/10^3 mi.

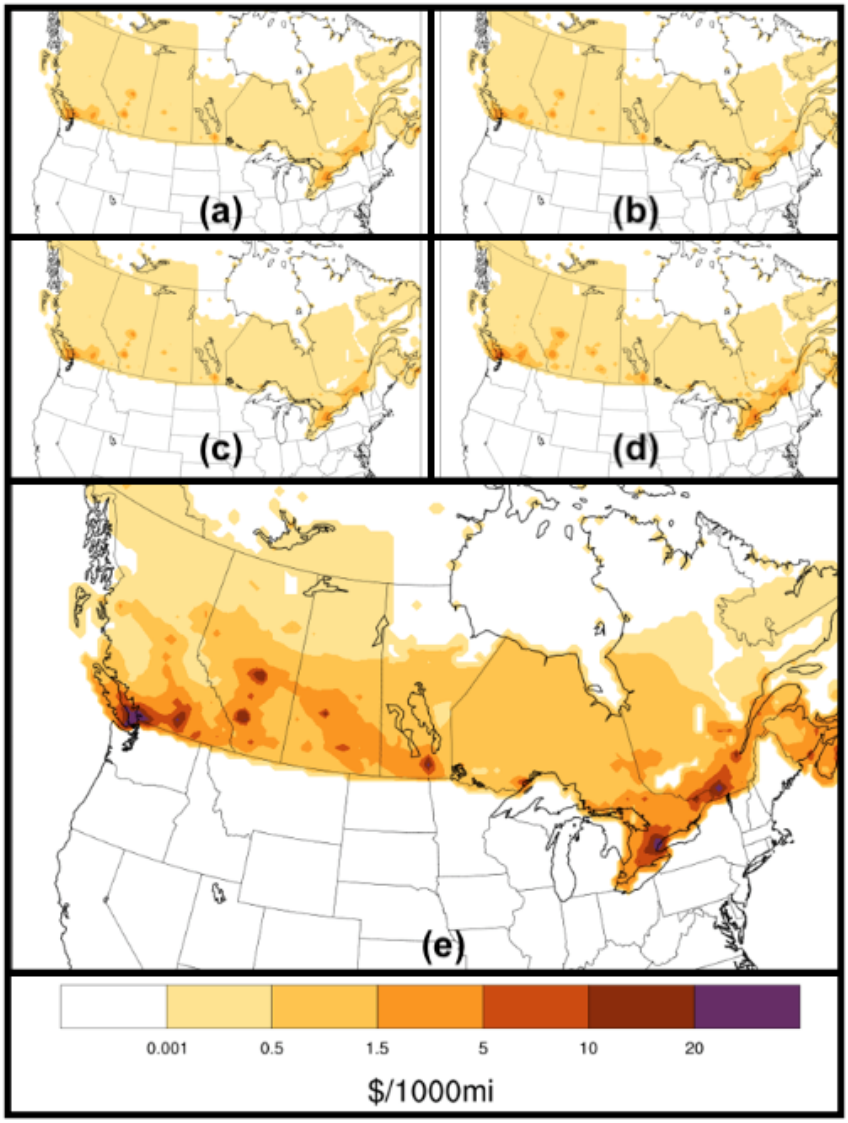

Figure 5-5: The per-vehicle health benefit results, PVHB($PM_{2.5}$), are shown over five age bins, including - (a) 0-2, (b) 3-5, (c) 6-9, (d) 10-13, and (e) >14 years. The spatial variation of the PVHBs for light-duty gasoline vehicles in Canada is depicted.

For comparison, Figure 5-6 shows the spatial distribution of health benefits through NO_X for LDDV, HDGV, and HDDV. For LDDV in Toronto, we find that the PVHB through NO_X increases from \$1.60 for new models to \$21/10^3mi for vehicles aged 14-16 years. In Quebec City, the PVHB ranges from \$4.80 for a new model to \$61/10^3 mi for a vehicle in age bin 5.

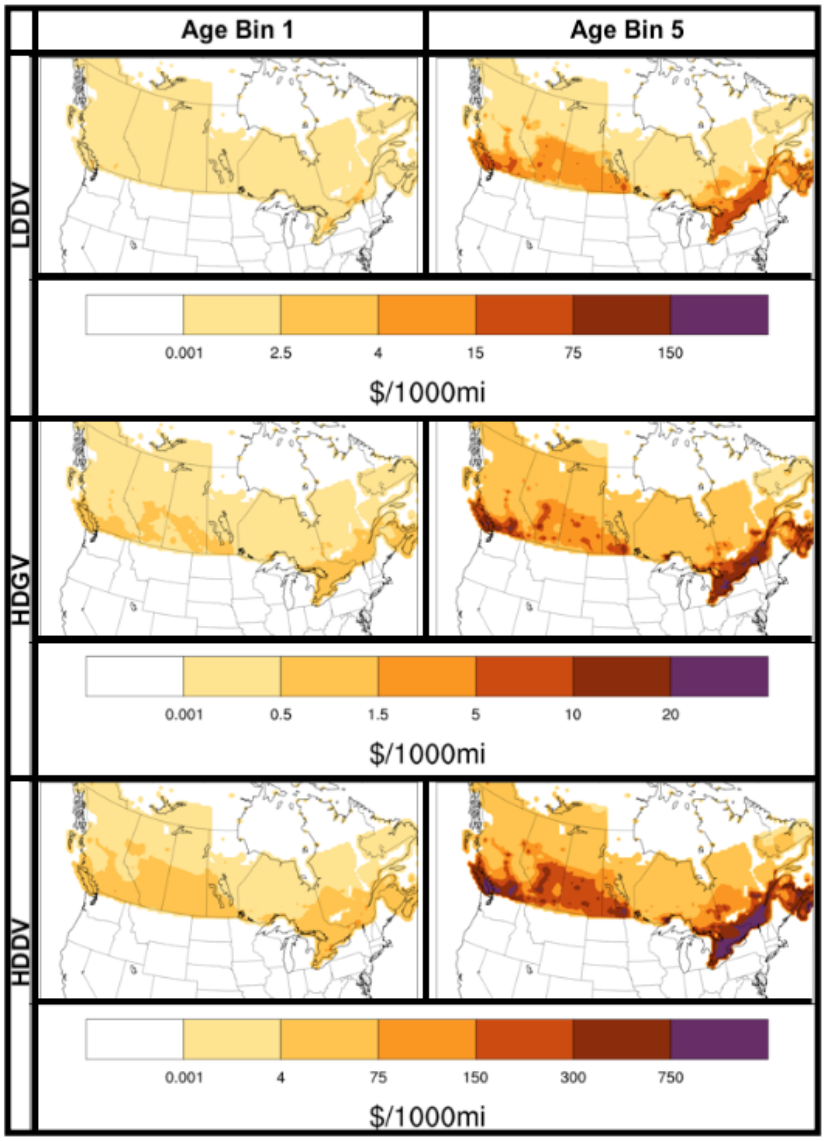

Figure 5-6: The per-vehicle health benefit results, PVHB(NO_X), are shown for light-duty diesel (LDDV), heavy-duty gasoline (HDGV), and heavy-duty diesel (HDDV) vehicles in Canada. The plot depicts the differences between spatial variation of health benefits for vehicles in Age Bin 1 (0-2 years) and Age Bin 5 (>14 years).

Figure 5-7 shows the spatial distribution of health benefits through $PM_{2.5}$ for LDDV, HDGV, and HDDV. We find that for LDDV, benefits through $PM_{2.5}$ increase from \$4.30 /10^3 mi for new models to \$13 for vehicles aged 14-16 years. In Quebec City, the PVHB ranges from \$4/10^3 mi for a new model to \$12/10^3 mi for a vehicle in age bin 5.

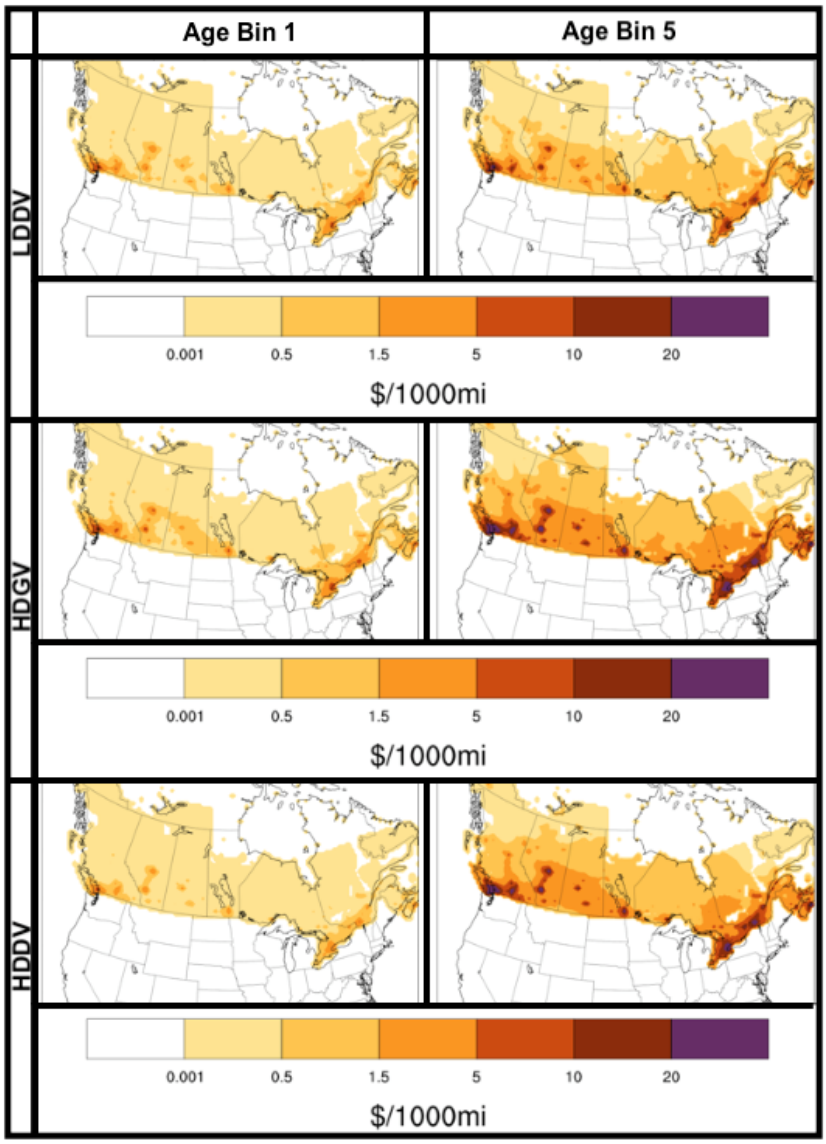

Figure 5-7: The per-vehicle health benefit results, PVHB($PM_{2.5}$), are shown for light-duty diesel (LDDV), heavy-duty gasoline (HDGV), and heavy-duty diesel (HDDV) vehicles in Canada. The plot depicts the differences between spatial variation of health benefits for vehicles in Age Bin 1 (0-2 years) and Age Bin 5 (>14 years).

The benefit of removing the emissions of a heavy-duty vehicle from the road, regardless of fuel type, is sizably larger than either light-duty category shown. For HDGV in Toronto, these benefits through NO_X start at \$16/10^3 mi for a new model and increase up to \$412/10^3 mi for a vehicle older than 14 years. The additional health benefits through $PM_{2.5}$ vary between \$5.50/10^3 mi (0-2 years old) to \$33/10^3 mi (14 years or more). In Quebec City, estimates through NO_X begin at \$48/10^3 mi and maximize for vehicles within

the last age bin at \$1249/10^3 mi. Additionally, benefits through $PM_{2.5}$ range from \$5.54/10^3 mi, up to \$31/10^3 mi.

The benefits through NO_X for heavy-duty diesel vehicles begin at \$38/10^3 mi and increase to up to \$769/10^3 mi in Toronto, with additional benefits through $PM_{2.5}$ of between \$3/10^3 mi for a new model to approximately \$27/10^3 mi for vehicles older than 14 years. Estimates for the PVHB through NO_X are larger in Quebec City, beginning with \$113/10^3 mi and increasing to \$2285/10^3 mi for vehicles of 14 years or more. Accounting for health benefits through $PM_{2.5}$ control results in additional benefits of \$2.90/10^3 mi for new models and up to \$25/10^3mi for vehicles over 14 years.

5.3.3 Per-Vehicle Health Benefit (\$/vehicle-year): Canada

We show the spatial distribution in PVHBs through NO_X on an annual, per-vehicle basis for light-duty gasoline vehicles in Canada in Figure 5-8. We find PVHB through NO_X for LDGV ranged from \$24/vehicle-year for a new model to \$418/vehicle-year for a model aged 14 or more years in Toronto and between \$62-\$1167/vehicle-year around Quebec City.

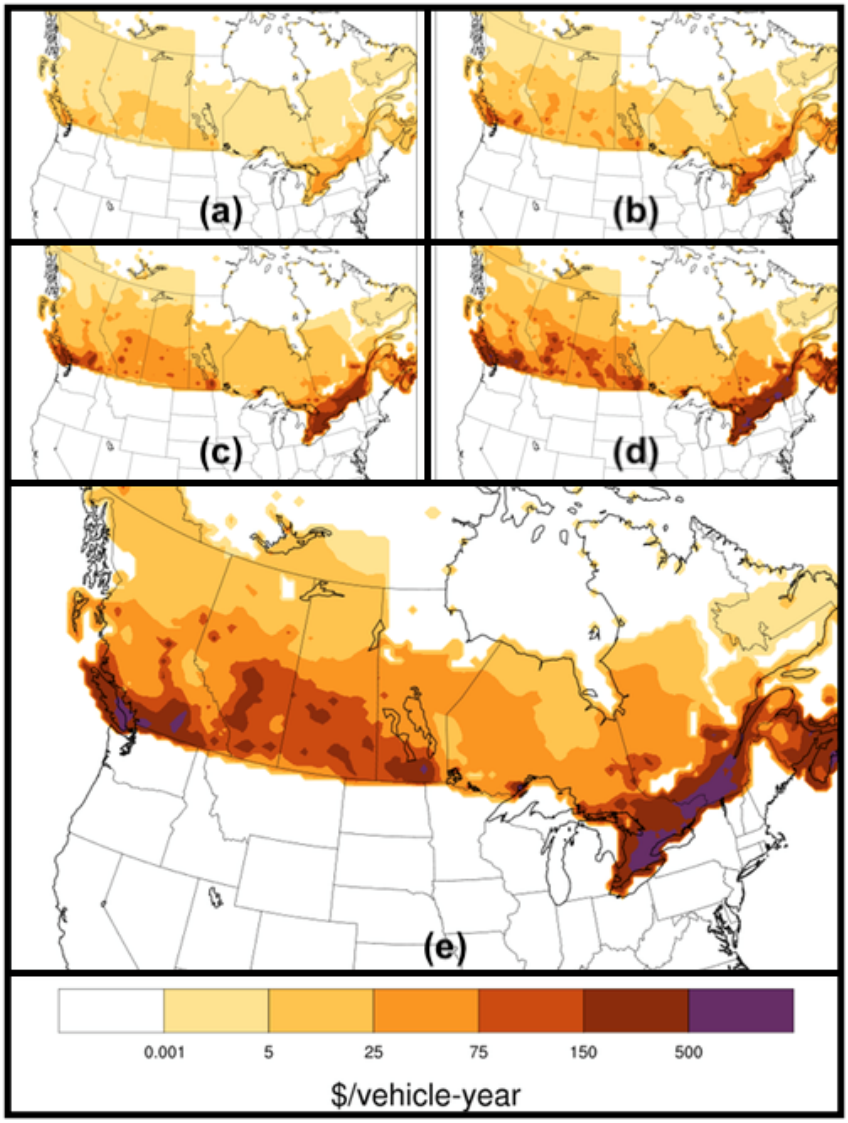

Figure 5-8: The per-vehicle health benefit results, PVHB(NO_X), are shown over five age bins, showing the results for light-duty gasoline vehicles in Canada in terms of dollar per vehicle per year. The plots depict vehicles grouped by age (a) 0-2 years, (b) 3-5 years, (c) 6-9 years, (d) 10-13 years, and (e) 14 or more years.

The total damage to public health through $PM_{2.5}$ is shown in Figure 5-9 on an annual, per-vehicle basis for light-duty gasoline vehicles in Canada. The societal health benefits of $PM_{2.5}$ range from between $27-$128/vehicle-year in Toronto and between $25-$120/vehicle-year in Quebec City, for age bins 1-5 respectively.

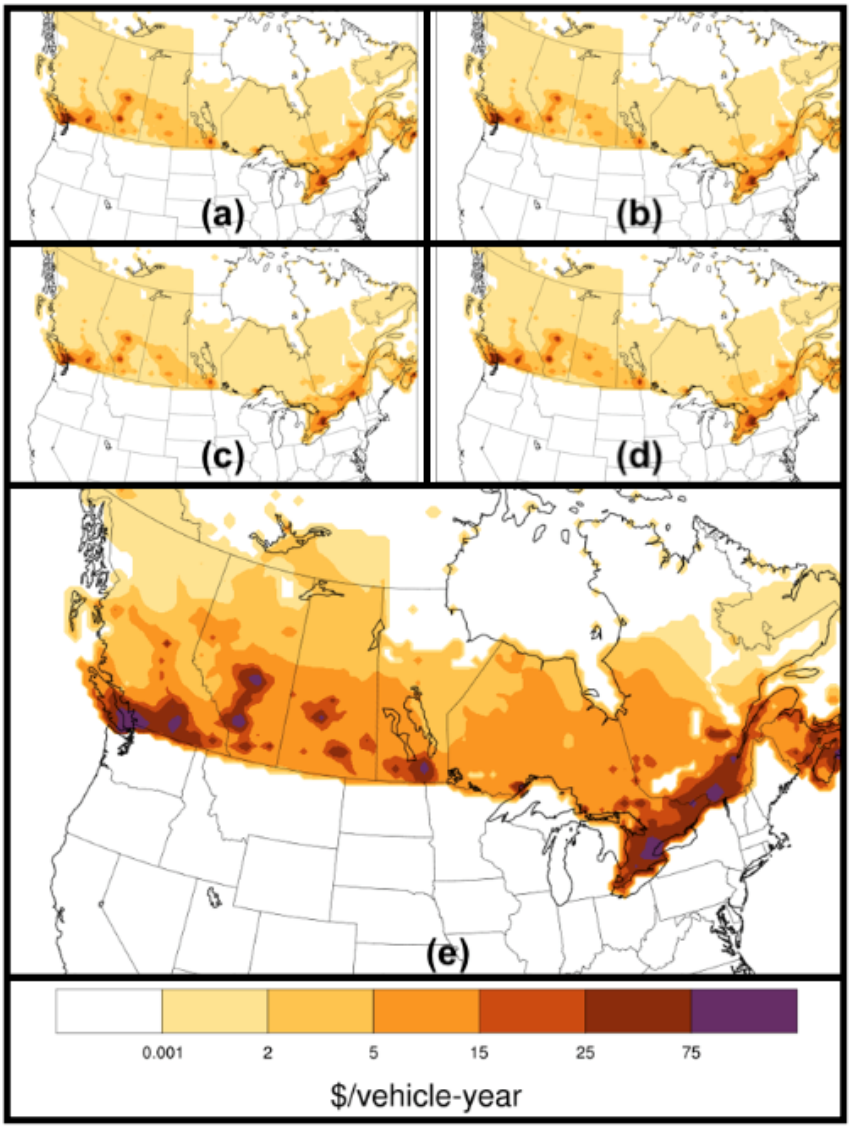

Figure 5-9: The per-vehicle health benefit results, PVHB($PM_{2.5}$), are shown over five age bins, showing the results for light-duty gasoline vehicles in Canada in terms of dollar per vehicle per year. The plots depict vehicles grouped by age (a) 0-2 years, (b) 3-5 years, (c) 6-9 years, (d) 10-13 years, and (e) 14 or more years.

For LDDV in Toronto, we find that benefits through NO_X (Figure 5-10) increase from $20 for new models to $164/vehicle-year for vehicles aged 14-16 years. In Quebec City, the PVHB ranges from $60/vehicle-year for a new model to $487/vehicle-year for a vehicle in age bin 5.

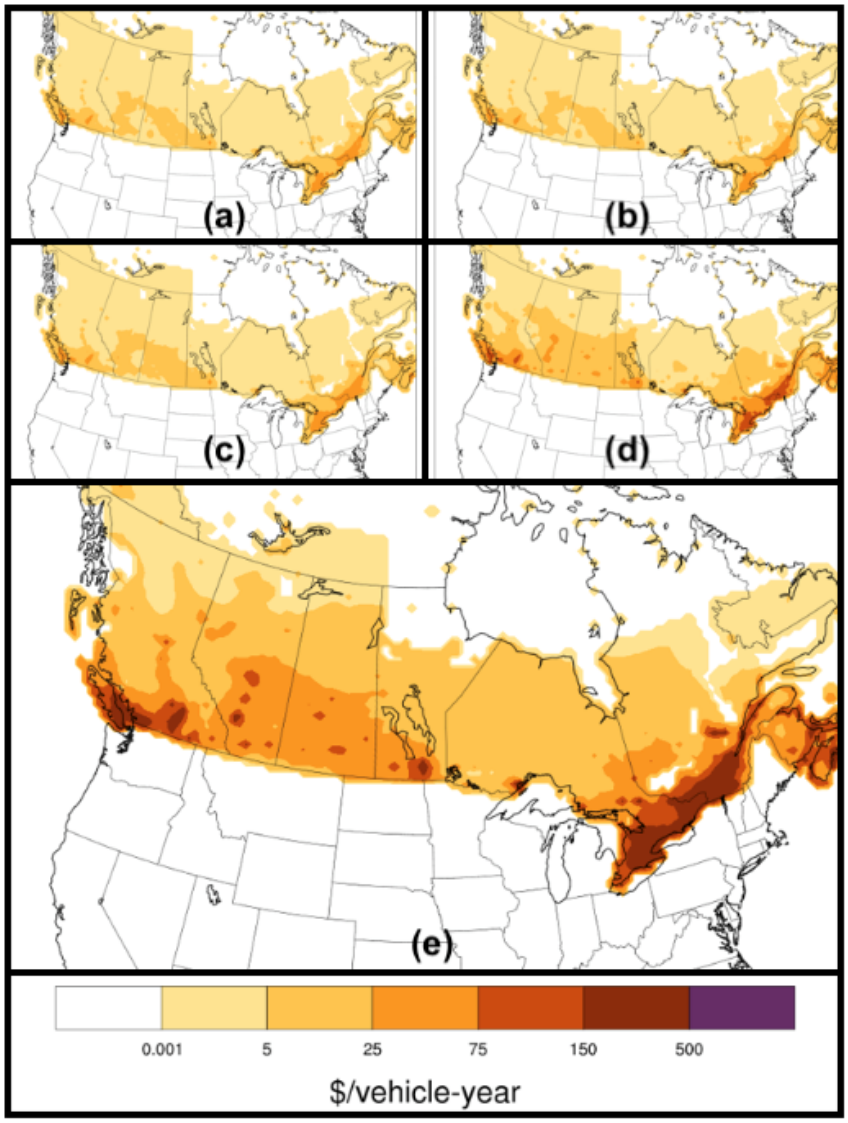

Figure 5-10: The per-vehicle health benefit results, PVHB(NO_X), are shown over five age bins, showing the results for light-duty diesel vehicles in Canada in terms of dollar per vehicle per year. The plots depict vehicles grouped by age (a) 0-2 years, (b) 3-5 years, (c) 6-9 years, (d) 10-13 years, and (e) 14 or more years.

Additional health benefits through $PM_{2.5}$ are presented in Figure 5-11. The PVHB through $PM_{2.5}$ results in estimates of between $54-$101/vehicle-year in Toronto and between $51 to $95/vehicle-year in Quebec City.

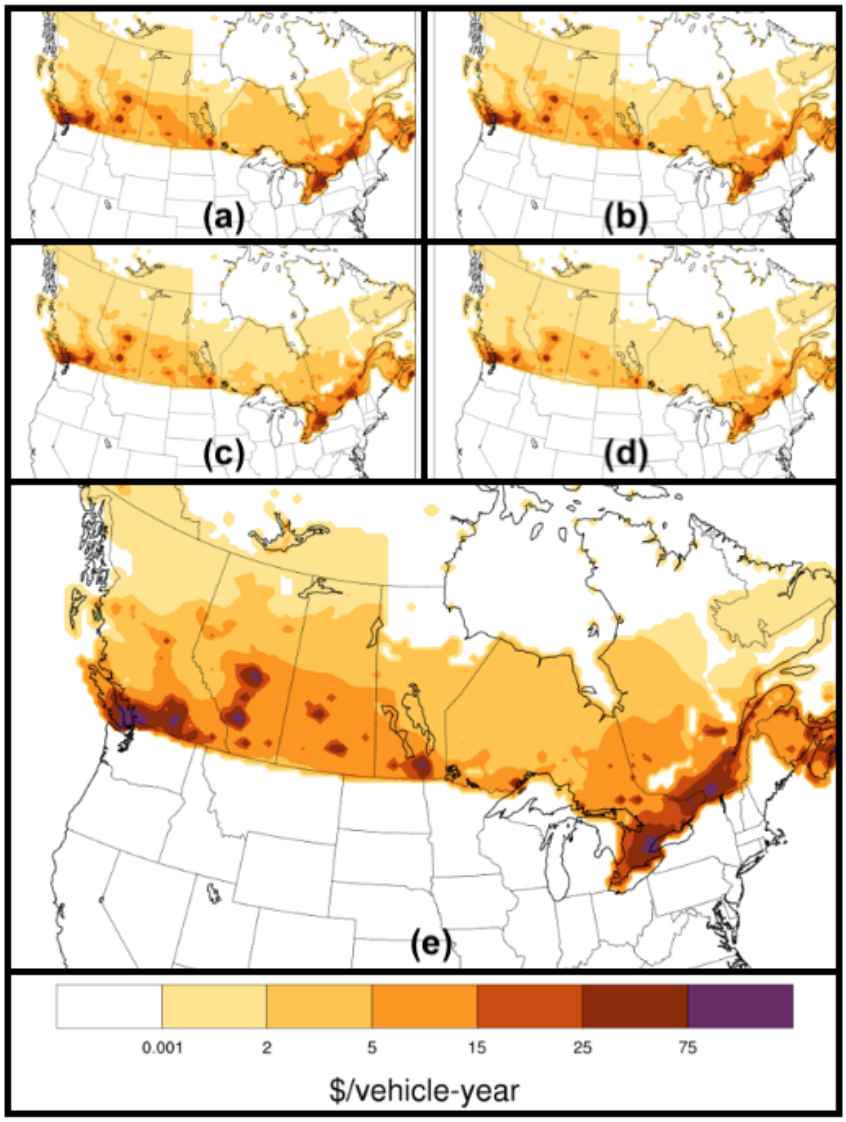

Figure 5-11: The per-vehicle health benefit results, PVHB($PM_{2.5}$), are shown over five age bins, showing the results for light-duty diesel vehicles in Canada in terms of dollar per vehicle per year. The plots depict vehicles grouped by age (a) 0-2 years, (b) 3-5 years, (c) 6-9 years, (d) 10-13 years, and (e) 14 or more years.

5.3.4 Emission Factors: United States

Average emission factors for LDGV in the United States were estimated to range from 0.1202 gNO_X/mi for models aged 0-2 years, up to 0.5608 gNO_X/mi for vehicles of 14 years or more. Estimates for NO_X emissions from LDDV range from 0.2333 gNO_X/mi for new models and 1.4916 gNO_X/mi for models within the last age bin. Particulate matter emission factors from LDGV ranged from 0.0070 to 0.0074 $gPM_{2.5}$/mi for the first and last

age bin, respectively. Average emissions from LDDV ranged from 0.0049 to 0.1217 $gPM_{2.5}$/mi for vehicles in age bins 1 and 5, respectively.

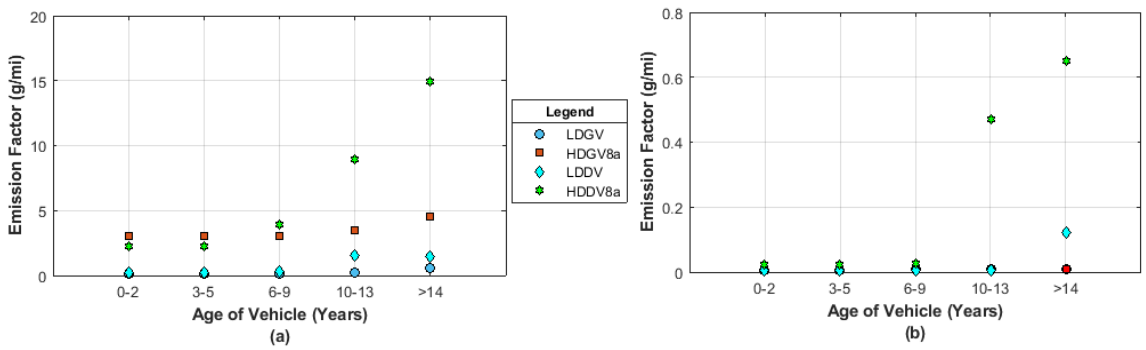

Figure 5-12: Emission factors of (a) NO_X and (b) $PM_{2.5}$ are shown over five age bins, showing the results for light-duty gasoline (LDGV), light-duty diesel (LDDV), heavy-duty gasoline (HDGV) and heavy-duty diesel (HDDV) vehicles in the United States.

Emission estimates for HDGV8a and HDGV8b were found to be 3.0497 gNO_X/mi for new models, up to 4.5693 gNO_X/mi for models older than 14 years. For HDDV, emission factors are lower for newer models (2.2452 gNO_X/mi), but increase to 14.9585 for older models (14-16 years). Emission factors of $PM_{2.5}$ are estimated for bins 1 and 5 to be 0.0077 $gPM_{2.5}$/mi and 0.0082 $gPM_{2.5}$/mi. For HDDV8a and HDDV8b, these estimates are 0.0238 – 0.6487 $gPM_{2.5}$/mi.

5.3.5 Per-Vehicle Health Benefit ($/$10^3$ miles): United States

Estimates of the per-vehicle health benefit are consistent with the spatial trends addressed in the Canadian analysis, with local maxima appearing in urban cores. The results of the PVHB through NO_X are shown in dollar per thousand vehicle-miles travelled for LDGV within age bins 1-5 in Figure 5-13. Results in Houston for LDGV range from

$2.60/10^3 mi to $12/10^3 mi for age bins 1 and 5, respectively. In Atlanta, these estimates start at $8.90/10^3 mi for new models and can be up to $42/10^3 mi for vehicles above 14 years.

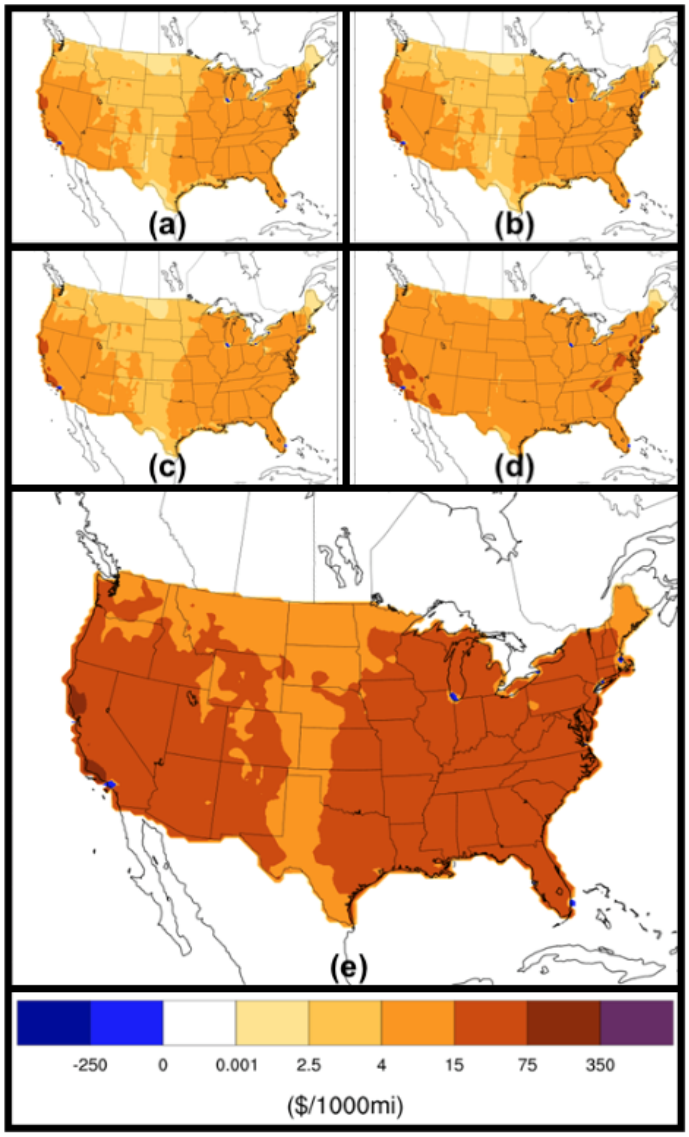

Figure 5-13: The spatial variation of the per-vehicle health benefit results, PVHB(NO_X), is shown for light-duty gasoline vehicles in the US in terms of dollar per thousand vehicle-miles travelled. The plots depict vehicles grouped into age bins of (a) 0-2, (b) 3-5, (c) 6-9, (d) 10-13, and (e) >14 years.

The results of the PVHB through $PM_{2.5}$ for LDGV are reported in dollar per thousand vehicle-miles travelled in Figure 5-14. Additional benefits through PM2.5 are shown for

LDGV within age bins 1-5 in Figure 5-14 and vary between \$1.91/10^3 mi to \$2.01/10^3 mi in Houston and between \$1.65 to \$1.74/10^3 mi in Atlanta.

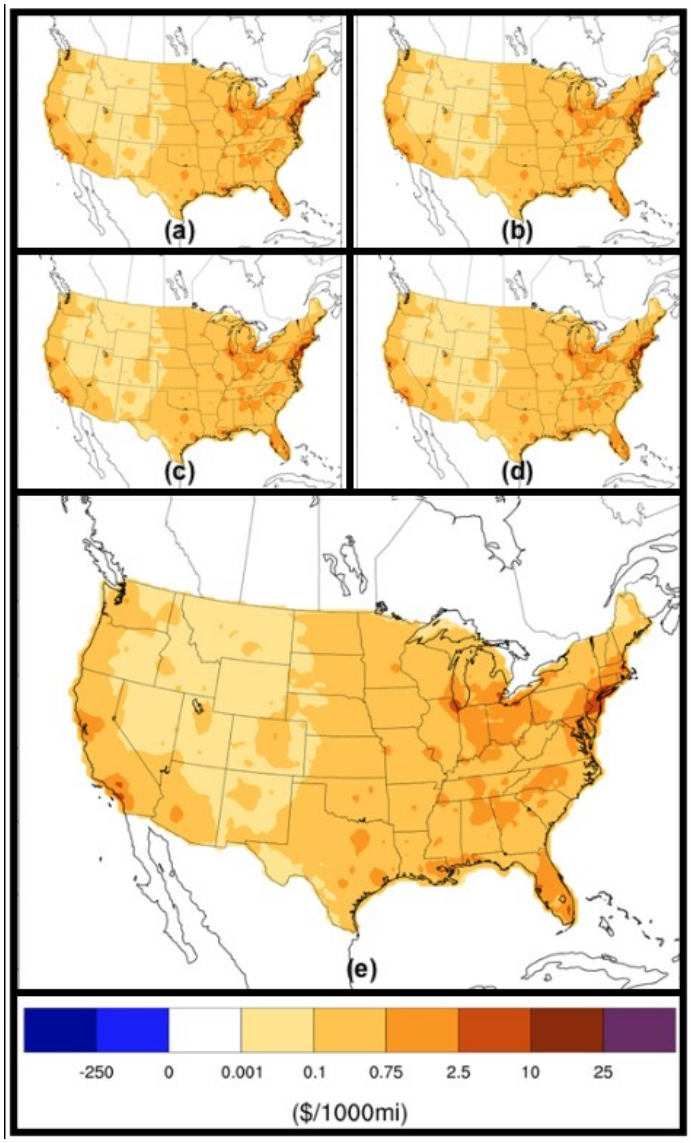

Figure 5-14: The spatial variation of the per-vehicle health benefit results through $PM_{2.5}$ is shown for light-duty gasoline vehicles in the US in terms of dollar per thousand vehicle-miles travelled. The plots depict vehicles grouped into age bins of (a) 0-2, (b) 3-5, (c) 6-9, (d) 10-13, and (e) >14 years.

For comparison, we also show the spatial distribution of health benefits for LDDV, HDGV, and HDDV in Figure 5-15 (showing benefits through NO_X) and Figure 5-16 (showing benefits through $PM_{2.5}$). In all cases, the results show that the PVHB expressed in terms of dollar per thousand vehicle-miles travelled consistently increases with vehicle age. The

PVHB through NO_X ranges from \$5.05 - \$32/10^3 mi In Houston and \$17 - \$111/10^3 mi in Atlanta for vehicles in the newest and oldest age bins, respectively. The PVHB through $PM_{2.5}$ increases these values by \$1.30 - \$33/10^3 mi In Houston and \$1.20 - \$29/10^3 mi in Atlanta for vehicles in the newest and oldest age bins, respectively.

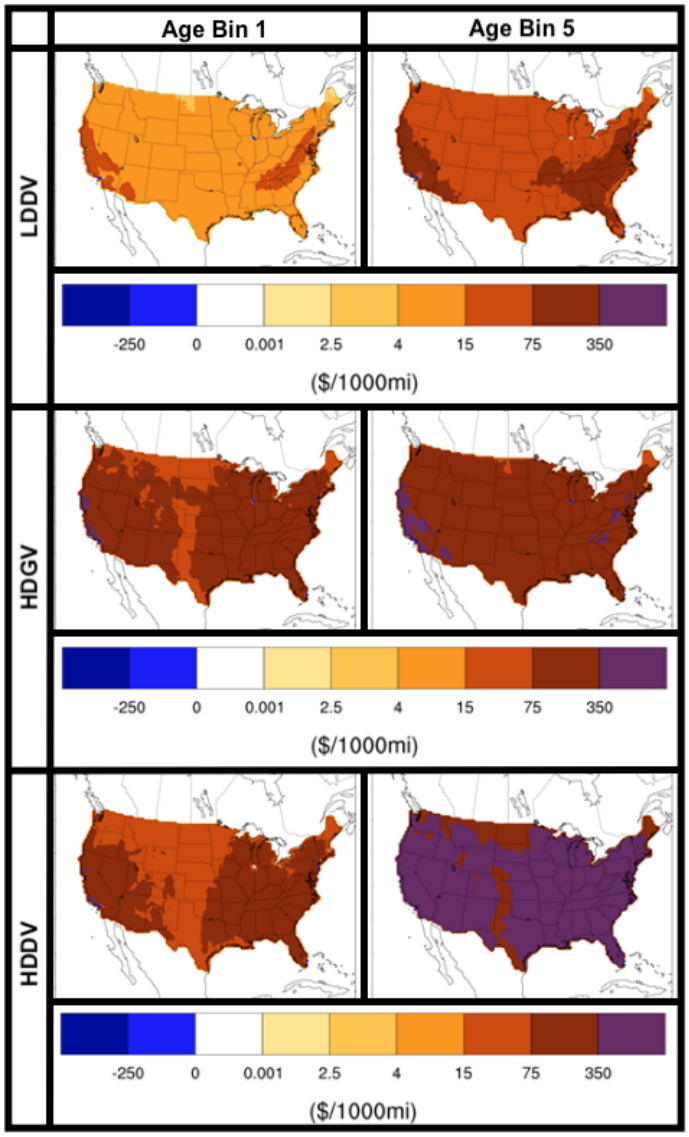

Figure 5-15: The per-vehicle health benefit results, PVHB(NO_X), are shown for light-duty diesel (LDDV), heavy-duty gasoline (HDGV), and heavy-duty diesel (HDDV) vehicles in the US. The plots depict the differences between spatial variation of PVHB estimates for vehicles in age bin 1 (0-2 years) and age bin 5 (>14 years).

The PVHB through NO_X for HDGV vary from \$66 - \$99/10^3 mi in Houston and \$227 to \$340/10^3 mi in Atlanta. The PVHB through NO_X for HDDV is significantly higher for older

vehicles than new models, ranging from \$49-\$323/10^3 mi in Houston and between \$167 - \$1112/10^3 mi in Atlanta.

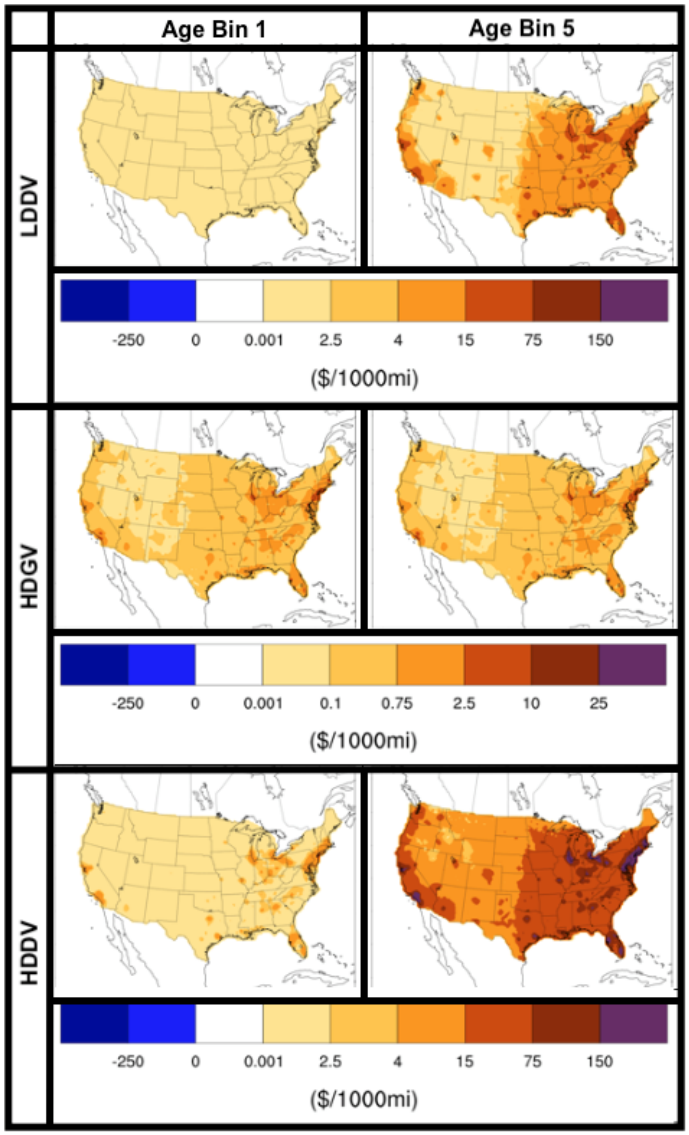

Figure 5-16: The per-vehicle health benefit results, PVHB($PM_{2.5}$), are shown for light-duty diesel (LDDV), heavy-duty gasoline (HDGV), and heavy-duty diesel (HDDV) vehicles in the US. The plots show the spatial variation of PVHB estimates for vehicles in age bin 1 (0-2 years) and age bin 5 (>14 years).

Supplemental health benefits though $PM_{2.5}$ range from \$2.11-\$2.22/10^3 mi in Houston and \$1.83-\$1.93/10^3 mi in Atlanta for HDGV. The benefits for HDDV increase significantly, with \$6.50 - \$177/10^3 mi in Houston and between \$5.60-\$153/10^3mi in Atlanta.

5.3.6 Per-Vehicle Health Benefit ($/vehicle-year): United States

We find PVHB through NO_X (Figure 5-17) for LDGV ranged from $36/vehicle-year for a new model to $90/vehicle-year for a model aged 14 or more years in Houston and between $124-$308/vehicle-year around Atlanta.

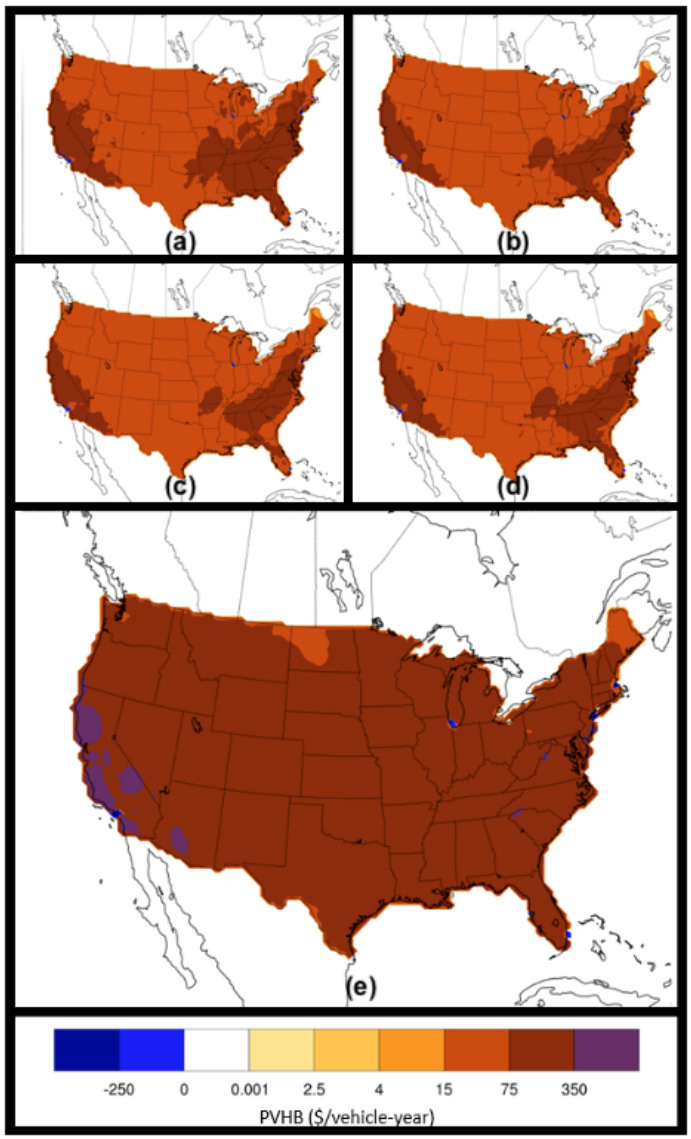

Figure 5-17: The per-vehicle health benefit results, PVHB(NO_X), are shown over five age bins, showing the results for light-duty gasoline vehicles in the US in terms of dollar per vehicle per year. The plots depict vehicles grouped by age (a) 0-2 years, (b) 3-5 years, (c) 6-9 years, (d) 10-13 years, and (e) 14 or more years.

In terms of the PVHB through $PM_{2.5}$ (Figure 5-18), we find PVHBs ranging from between \$26-\$15/vehicle-year in Houston and between \$23-\$13/vehicle-year in Atlanta for the newest and oldest model years, respectively.

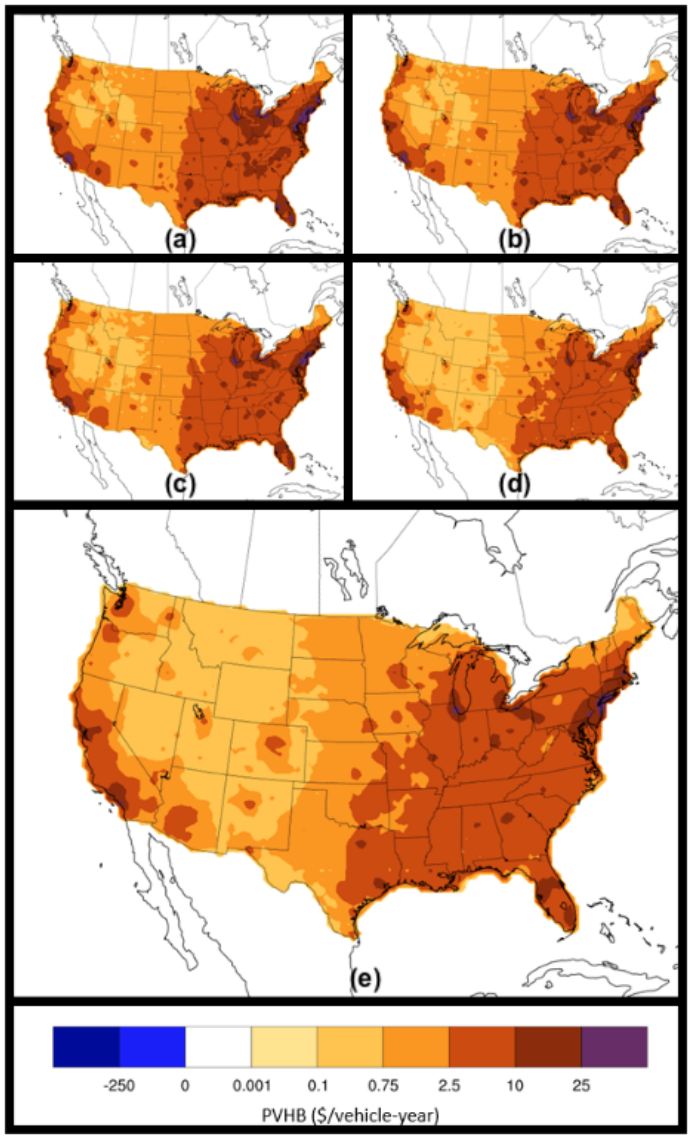

Figure 5-18: The per-vehicle health benefit results, PVHB($PM_{2.5}$), are shown over five age bins, showing the results for light-duty gasoline vehicles in the US in terms of dollar per vehicle per year. The plots depict vehicles grouped by age (a) 0-2 years, (b) 3-5 years, (c) 6-9 years, (d) 10-13 years, and (e) 14 or more years.

Estimates of the PVHB through NO_X are shown for light-duty diesel vehicles in the US in Figure 5-19. The PVHBs increase from \$240/vehicle-year for new models (0-2 years)

up to $820/vehicle-year for vehicles older than 14 years in Atlanta. Estimates for the PVHB in Houston ranges from $70 for a new vehicle (0-2 years), up to $239/vehicle-year for models above 14 years.

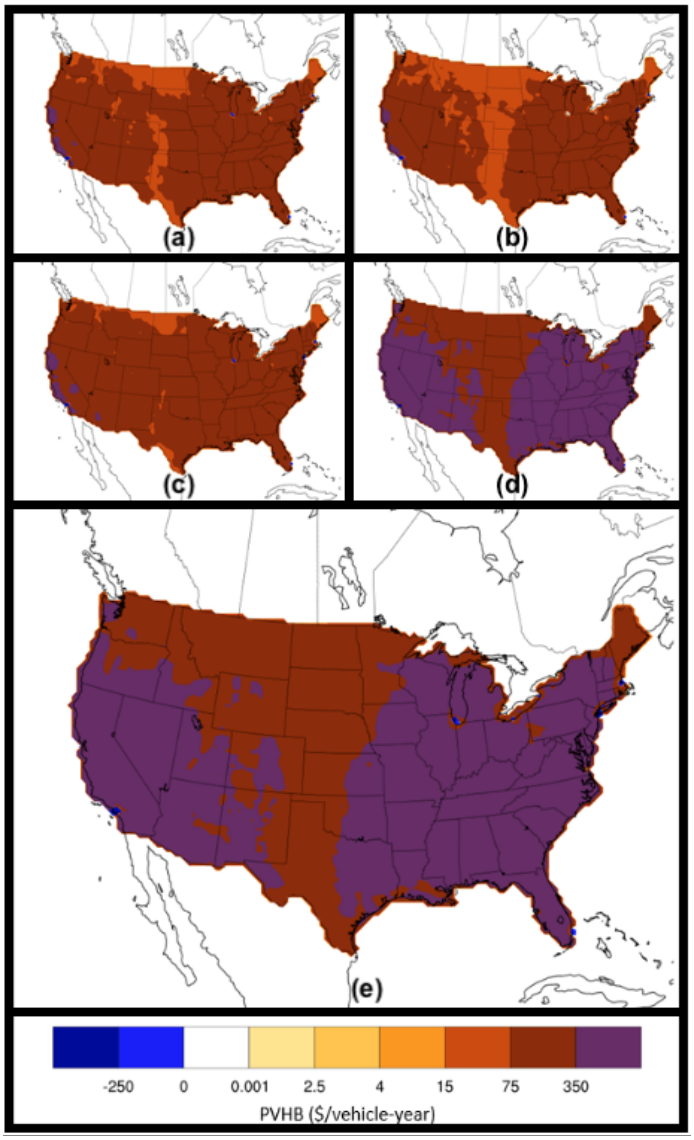

Figure 5-19: The per-vehicle health benefit results, PVHB(NO_X), are shown over five age bins, showing the results for light-duty diesel vehicles in the US in terms of dollar per vehicle per year. The plots depict vehicles grouped by age (a) 0-2 years, (b) 3-5 years, (c) 6-9 years, (d) 10-13 years, and (e) 14 or more years.

Estimates of the PVHB through $PM_{2.5}$ are estimated to increase with vehicle age across the domain. In Atlanta, these benefits begin at $16/vehicle-year for a new model

(0-2 years) and increase over all age bins, maximizing at $213/vehicle-year for models greater than 14 years old. In Houston, the PVHB estimates are larger than those in Atlanta, starting at $18/vehicle-year for new models (<2 years old) and increase to $245/vehicle-year for the oldest models in this study (14 years or more). The spatial variation in the $PM_{2.5}$ per-vehicle health benefits is depicted in Figure 5-20.

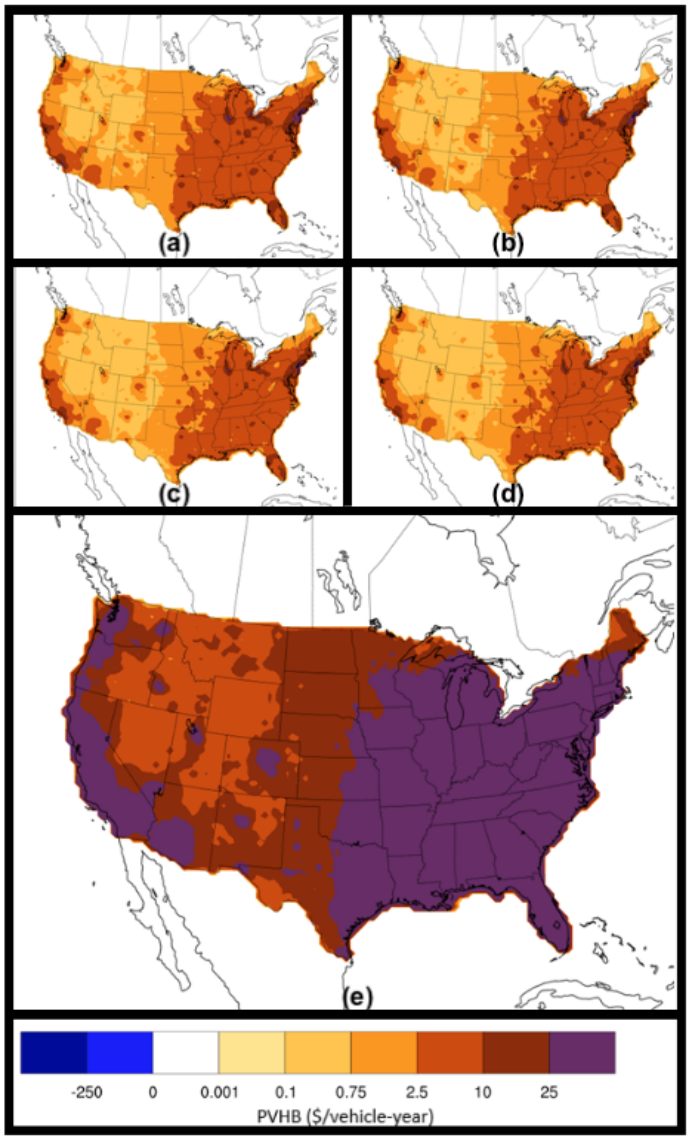

Figure 5-20: The per-vehicle health benefit results, PVHB($PM_{2.5}$), are shown over five age bins, showing the results for light-duty diesel vehicles in the US in terms of dollar per vehicle per year. The plots depict vehicles grouped by age (a) 0-2 years, (b) 3-5 years, (c) 6-9 years, (d) 10-13 years, and (e) 14 or more years.

What is unique to this analysis for the United States is the existence of negative health benefits in localized, urban areas and downtown cores (apparent in Figure 5-13). For example, the PVHB through NO_X emissions from LDGV in New York, increased from \$-3.40/10^3 mi for new vehicle models to \$-16/10^3 mi for vehicle above 14 years old. Additional benefits through $PM_{2.5}$ range from \$7.49/10^3 mi to \$7.88/10^3 mi for bins 1 and 5. Even more striking is the PVHB through NO_X emissions from LDGV in Los Angeles, California, which ranged from \$-22/10^3 mi to \$-103/10^3 mi for age bins 1 and 5, respectively. Additional benefits through $PM_{2.5}$ range from \$2.19/10^3 mi to \$2.30/10^3 mi for bins 1 and 5.

The health sensitivities are estimated to assess the impact of NO_X emissions on Canadian public health, through emissions of NO_X as NO and NO_2, but the model also accounts for the health damage (through chronic 1-hour ozone) that is caused by NO_2. It has been well established in the literature that reducing emissions in urban areas with high NO_X concentrations results in increased ozone due to ozone titration, worsening health for exposed populations locally. However, ozone titration generates NO_2, which is associated with mortality in Canada, and therefore, such negative PVHBs are not seen in Canada.

In New York and Los Angeles, reducing the tailpipe emissions of NO_X from a vehicle will initially increase the contribution from transportation on domain-wide mortality; however, any reduction in emissions of $PM_{2.5}$ positively impacts the surrounding environment. Therefore, we investigate if the $PM_{2.5}$ reductions can compensate the

damage from NO_X. We find that for older vehicles in New York, this is possible. The PVHB through NO_X for vehicles older than 14 years is $-309/vehicle-year; however, benefits through $PM_{2.5}$reductions increase to $963/vehicle-year, resulting in a scenario where overall, even in NO_X-rich areas, targets set on reducing emissions from these older vehicles will positively affect the environment.

The MOBILE6.2C model has not been updated since 2003 and as such, has its own limitations. The distribution in NO_X and $PM_{2.5}$ emission factors is more pronounced as vehicles age for Canada, as the emissions are under-predicted for newer models, and over-predicted for the oldest models. For example, NO_X emission factors for light-duty gasoline vehicles in Canada ranged from 0.0251 gNO_X/mi (0-2 years old) up to 0.7464 gNO_X/mi (14 or more years old), whereas emission factors in the United States had a smaller range of between 0.1202 gNO_X/mi to 0.5608 gNO_X/mi, for vehicles of 0-2 and 14 years or more, respectively. Although the emission factor estimates for Canada are still reasonable, this is largely due to technological advances in emission control systems and new calculation methods for vehicle deterioration that are not accounted for in the MOBILE6.2C model (MOBILE6.2C assumes deterioration with is linear; however, MOVES has an updated methodology) [139].

The difference in the trend of PVHB results expressed in dollar per vehicle (per year) for the United States and Canada is due primarily to the emission factors used for each domain, as well as the vehicle mileage (VMT) used to estimate the PVHB. For the United

States, the emission factor estimates do not change for the first and second age bin (0.1202 g/mi) and do not have as large a range as those for Canada. As vehicle mileage decreases with vehicle age (13851 mi for age bin 1, followed by 12042 mi), the estimates of PVHB for the first age bin are larger than the next, but eventually maximize for age bins 3-5. In Canada, the PVHB increases consistently with vehicle age, since there is a distinct increase in emissions (0.0247 g/mi for the first age bin and 0.0932 for the following bin). When combined with the VMT estimates for each age bin (12564 mi for the first, followed by 11481 mi).

The PVHB estimates cannot be directly compared due to the fundamental difference in national averages for vehicle emissions, driving patterns (Americans drive 13851 mi per year; Canadians drive 12564 mi per year), and marginal benefit estimations (Benefits for the US are through O_3 and $PM_{2.5}$, but in Canada NO_2 is also included). The results up to this point demonstrate that there are benefits to reducing mobile emissions in both Canada and the United States, regardless of vehicle type or age. The most significant benefits are shown for heavy polluters, including heavy-duty vehicles (both gasoline and diesel) and older vehicle models. These per-vehicle health benefit in dollar per thousand miles could help to create incentives to inform policy for heavily polluted regions, to reduce the distance travelled by vehicles, or impose restrictions on vehicle driving times/days.

We stand now where two roads diverge. But unlike the roads in Robert Frost's familiar poem, they are not equally fair. The road we have long been travelling is deceptively easy, a smooth superhighway on which we progress with great speed, but at its end lies disaster. The other fork of the road –the one less traveled by – offers our last, our only chance to reach a destination that assures the preservation of the earth.

Rachel Carson

Silent Spring

6 CONCLUSIONS

The two studies presented in chapters 4 and 5 of this thesis contribute novel information to the field of air quality, and serve as further evidence that adjoint estimation methods are effective tools in simulating and investigating the effect of policy options and related impacts.

In Chapter 4, an investigation is performed to answer a unique question for policy makers: "How can local (versus national) emission reduction and controls prove to be favourable policy options?". To address this, estimates of the marginal benefits due to chronic exposure to 1-hour ozone on a per-ton basis over emission abatement levels of 0 to 80% (in intervals of 10% emission abatement) in two heavily polluted urban areas in the United States: Los Angeles, California (LA) and New York, New York (NY). Three main research goals were identified for the study in Chapter 1, including identifying the Tipping Point of NO_X control for both LA and NY and providing estimates of the Break-Even Point at which disbenefits are fully compensated in both LA and NY. At the conclusion of this paper, a comparison of the results of local and national emission reduction policy scenarios are compared (with their respective regional and national societal health damage estimates) to demonstrate how emission control strategies can be effectively applied in heavily polluted areas.

Two emission reduction scenarios (regional and national abatement) as well as the effect of the emission control scenario on both local and national (domain-wide) mortality

estimates are investigated. The results validate that of previous studies, showing that each additional tonne of NO_X control ensues a larger benefit than the previous tonne and that compounding benefits of NO_X control will compensate initial disbenefits caused by titration within the city core. Our findings suggest that substantial benefits may be obtained with focus on NO_X control within urban cores of heavily polluted cities, where emission reductions appear unfavourable in the short term. Although both national and regional abatement scenarios provided different monetary benefit results, both scenarios resulted in similar tipping point estimates for both the national health benefits (50-52% abatement in LA; 7-8% abatement in NY) and the local health benefits (54-57% in LA; 64-62% in NY). For example, if we consider either emission control policy on the national health benefits of reducing emissions in New York, it is suggested that after a mere 7-8% emission abatement, the environment will begin to accrue benefits of further emission controls; however, when we consider the local health impact, these benefits do not exist until 62-64% emission abatement. This also applies to the break-even point, nationally, cumulative benefits will compensate immediate disbenefits in LA at 85-86% and in NY at 14-16%; however, locally, these accrued disbenefits will not be compensated until 93-94% emission abatement in LA and 100% in NY.

As the transportation sector is the dominant source responsible for NO_X emissions and remains a significant source of primary $PM_{2.5}$ emissions, the second paper presented in Chapter 5 may serve as a seamless extension of the research presented in Chapter 4. It is intended to support policies created to reduce emissions in urban cores or NO_X-rich

areas by focusing on emissions from on-road vehicles, the dominant source of NO_X emissions in both Canada and the US. In order to meet the TP in NY and LA, new policies targeting older vehicles may be enforced – reducing the societal health burden overall.

With the advances in vehicle technology and the implementation of more stringent emission standards, the Chapter 5 addresses the question of how a complex CTM such as CMAQ can be used in conjunction with vehicle emissions modelling software to estimate the contribution to the overall societal public health burden from on-road vehicles in Canada and the United States. A distribution of health impacts is also investigated by grouping vehicles into five age bins, in an effort to demonstrate the effect of age on domain-wide mortality estimates.

Emission factor estimates produced by MOBILE6.2C for Canada as well as those predicted by MOVES2010b for the United States are found to increase consistently with vehicle age for all vehicle types in both countries. The PVHB estimates were analyzed for both domains in terms of (1) dollars per thousand miles traveled ($\$/10^3$ mi) as well as (2) dollars per vehicle per year. The results in both cases ($\$/10^3$ mi) and ($/vehicle-year) were found to be lowest for new models and increase continuously over subsequent age bins, and maximize for the last bin (containing vehicles older than 14 years) in Canada. In the United States, this trend is apparent in the PVHB results expressed in $\$/10^3$ mi; however, the distributions in terms of $/vehicle-year are found to decrease initially from the first age bin to the third bin, but increase and maximize from bins 3 to 5. This difference in the

results for Canada and the United States is attributed to the differences in VMT between model years (new models described in the first age bin (0-2 years) produce higher estimates on a dollar per vehicle basis than those in the next age bin (3-5 years) as the distance driven year decreases with vehicle age).

We acknowledge that there are limitations associated with the models and results presented in this thesis. The CMAQ-adjoint sensitivities are modeled for both papers by estimating the health impacts of chronic exposure to NO_X, $PM_{2.5}$, and O_3. We recognize that the analyses in Chapters 4 and 5 are both limited by uncertainties within the data used to conduct our simulations, including changes in population demographics, epidemiology, atmospheric modelling and emission inventories. We analyze the changes in the marginal benefit for each location; however, we use a coarse modelling domain of a 36-km resolution, which does not allow for an account of how the distribution of pollution changes within the 36-km box, or how the sensitivities differ across the city. Performing this analysis at a higher resolution would allow for a more refined spatial distribution of these results. We also assume a constant value of statistical life for the entire population; however, the willingness to pay for an individual may vary across the country. In modeling health impacts for the United States, we incorporate estimates for mortality and population information based on epidemiology in 2007 (and we apply the assumption that the epidemiology has not changed drastically since then).

In Chapter 4, tipping point and break-even point estimates are given for NY and LA based on baseline 2007 emissions. The amount of NO_X emissions nation-wide in the United States have declined by 42% between 2007-2016 [34]. Therefore, we can compare this with the MB results for nation-wide abatement of 40% (Case 2), and we find that any further reduction in emissions will result in more substantial health benefits as we move closer towards the tipping point (a less polluted, more pristine environment). This 42% decrease in emissions since 2007 leaves us at a critical position for policy making – and provides direction and incentive for policy options that favour local emission reduction controls. In future work, it would be interesting to re-run all simulations for local health impacts over New York or Los Angeles with high-resolution model inputs with a 1-km horizontal resolution instead of the current coarse grid resolution of 36-km.

In Chapter 5, although the emission factor estimates produced by the MOBILE6.2C are approximated for the Canadian vehicle fleet, a limitation exists as the model itself is now outdated and has not been updated since the last update in 2003. With advances in technology and emission controls, limitations within the model exist when attempting to model calendar years after 2009 [139]. Because of this, emission factors for new models are under-predicted and are over-predicted for the oldest models. Environment Canada has shifted to the MOVES model to estimate exhaust emissions from mobile sources; however, default input data that is representative of the Canadian fleet is not publicly available and therefore emission factors could not be modelled with MOVES. Therefore, future work should involve obtaining up-to-date input data (representative of the current

vehicle fleet) for the MOVES model and updating the emission factor estimates, and the resulting PVHB distributions for Canada with updated values from the MOVES model.

It’s a collective endeavor, it’s collective accountability and it may not be too late.

Christine Lagarde

On Climate Change

Managing Director, IMF

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8 APPENDIX A: Supplementary Material for Chapter 5

8.1 Calculations: Emission factor grouping

8.1.1 Canada

Emission factors were grouped into age bins outlined in the 2009 Canadian Vehicle Survey and previous transportation in Canada Reports [129]. To do this, the emission factors for each model year and vehicle class were averaged according to the vehicle-miles travelled (VMT) fraction on each of the 5 facility types.

$$EF_0 = (EF_0)(VMT_{0,FAC1}) + (EF_0)(VMT_{0,FAC2}) + (EF_0)(VMT_{0,FAC3}) + (EF_0)(VMT_{0,FAC4}) + (EF_0)(VMT_{0,FAC5}) \quad \text{Eq. 8-1}$$

Then, emission factors were grouped by vehicle age into age bins of 0-2 years, 3-5 years, 6-9 years, 10-13 years, and 14+ years. For instance,

$$EF_{0-2} = (EF_0)(VMT_0) + (EF_1)(VMT_1) + (EF_2)(VMT_2) \quad \text{Eq. 8-2}$$

8.1.2 United States

Emission factors were disaggregated into the same 28 vehicle classes as defined by MOBILE6.2C [123]. To do this, the emission factors for each model year and vehicle class were averaged according to percentages listed in Table 5-3.

$$EF_{LDGV} = (EF_0)(\%LDGV) \quad \text{Eq. 8-3}$$

$$EF_{0-2} = (EF_0)(VMT_0) + (EF_1)(VMT_1) + (EF_2)(VMT_2) \quad \text{Eq. 8-4}$$

8.2 Supplemental Tables

8.2.1 Emission factors – Canada

Table 8-1: Nitrogen oxide (NO_X) emission factors for 8 cities in Canada, segregated into age bins 1-5 (0-2 years, 3-5 years, 6-9 years, 10-13 years, and >14 years).

					NO_X (g/mi)				
Veh ID	AB	VN	CG	QC	TO	HX	FM	WH	YK
1	1	0.0247	0.0266	0.0284	0.0248	0.0234	0.0262	0.0236	0.0234
	2	0.0932	0.0940	0.0990	0.0969	0.0901	0.0950	0.0887	0.0859
	3	0.2110	0.2085	0.2185	0.2221	0.2052	0.2126	0.2008	0.1931
	4	0.4827	0.4660	0.4857	0.5139	0.4725	0.4800	0.4590	0.4377
	5	0.7591	0.7320	0.7626	0.8092	0.7435	0.7542	0.7223	0.6887
2	1	0.0362	0.0358	0.0375	0.0381	0.0352	0.0365	0.0344	0.0330
	2	0.1458	0.1401	0.1459	0.1556	0.1430	0.1447	0.1386	0.1318
	3	0.3072	0.2927	0.3042	0.3292	0.3018	0.3033	0.2919	0.2771
	4	0.6411	0.6048	0.6268	0.6903	0.6315	0.6296	0.6089	0.5760
	5	0.9369	0.8801	0.9138	1.0222	0.9235	0.9179	0.8921	0.8414
3	1	0.0728	0.0691	0.0717	0.0782	0.0716	0.0718	0.0691	0.0655
	2	0.2049	0.1941	0.2014	0.2204	0.2018	0.2017	0.1950	0.1849
	3	0.3789	0.3585	0.3716	0.4077	0.3732	0.3725	0.3607	0.3418
	4	0.7392	0.6947	0.7191	0.7978	0.7294	0.7242	0.7034	0.6651
	5	1.3607	1.2735	1.3209	1.4876	1.3428	1.3302	1.2961	1.2214
4	1	0.0859	0.0821	0.0854	0.0919	0.0843	0.0850	0.0817	0.0776
	2	0.2371	0.2257	0.2343	0.2544	0.2333	0.2339	0.2258	0.2145
	3	0.4280	0.4064	0.4216	0.4598	0.4213	0.4216	0.4077	0.3869
	4	0.8953	0.8417	0.8712	0.9663	0.8834	0.8772	0.8521	0.8058
	5	2.1305	2.0010	2.0762	2.3264	2.1020	2.0860	2.0329	1.9191
5	1	0.1429	0.1361	0.1413	0.1533	0.1406	0.1410	0.1362	0.1293
	2	0.3311	0.3149	0.3266	0.3557	0.3261	0.3263	0.3162	0.3004
	3	0.5535	0.5243	0.5437	0.5951	0.5450	0.5445	0.5269	0.4998
	4	1.1493	1.0778	1.1153	1.2415	1.1344	1.1248	1.0928	1.0321
	5	2.8839	2.7058	2.8054	3.1528	2.8486	2.8209	2.7579	2.6048
6	1	0.1380	0.1353	0.1315	0.1419	0.1382	0.1364	0.1385	0.1379
	2	0.1494	0.1464	0.1422	0.1536	0.1495	0.1476	0.1499	0.1492
	3	0.2345	0.2298	0.2234	0.2411	0.2347	0.2318	0.2354	0.2343
	4	1.6154	1.5834	1.5386	1.6611	1.6171	1.5966	1.6216	1.6140

	5	4.2157	4.1320	4.0153	4.3348	4.2201	4.1667	4.2317	4.2120
7	1	0.1843	0.1807	0.1755	0.1895	0.1845	0.1822	0.1850	0.1841
	2	0.1956	0.1918	0.1863	0.2012	0.1958	0.1934	0.1964	0.1955
	3	0.2780	0.2725	0.2648	0.2859	0.2783	0.2748	0.2791	0.2778
	4	1.6834	1.6500	1.6034	1.7309	1.6851	1.6638	1.6897	1.6819
	5	4.4220	4.3342	4.2118	4.5469	4.4265	4.3705	4.4387	4.4181
8	1	0.1647	0.1615	0.1569	0.1694	0.1649	0.1629	0.1654	0.1646
	2	0.1764	0.1729	0.1680	0.1814	0.1766	0.1744	0.1771	0.1762
	3	0.2583	0.2532	0.2460	0.2656	0.2585	0.2553	0.2592	0.2581
	4	1.6456	1.6129	1.5674	1.6921	1.6473	1.6265	1.6518	1.6441
	5	4.3149	4.2293	4.1098	4.4368	4.3194	4.2647	4.3313	4.3111
9	1	0.1912	0.1874	0.1821	0.1966	0.1914	0.1890	0.1919	0.1910
	2	0.2028	0.1988	0.1932	0.2086	0.2031	0.2005	0.2036	0.2027
	3	0.2988	0.2929	0.2846	0.3073	0.2991	0.2953	0.2999	0.2986
	4	1.9159	1.8778	1.8248	1.9700	1.9179	1.8936	1.9231	1.9142
	5	5.0333	4.9333	4.7940	5.1754	5.0385	4.9747	5.0523	5.0288
10	1	0.1890	0.1852	0.1800	0.1944	0.1892	0.1868	0.1897	0.1889
	2	0.2007	0.1967	0.1911	0.2063	0.2009	0.1983	0.2014	0.2005
	3	0.2961	0.2902	0.2820	0.3044	0.2964	0.2926	0.2972	0.2958
	4	1.9012	1.8634	1.8108	1.9549	1.9031	1.8790	1.9083	1.8995
	5	4.9942	4.8951	4.7568	5.1353	4.9994	4.9361	5.0132	4.9898
11	1	0.2102	0.2060	0.2002	0.2162	0.2104	0.2078	0.2110	0.2100
	2	0.2277	0.2232	0.2169	0.2341	0.2279	0.2250	0.2285	0.2275
	3	0.3317	0.3251	0.3159	0.3411	0.3321	0.3278	0.3330	0.3314
	4	2.0965	2.0549	1.9969	2.1557	2.0987	2.0721	2.1045	2.0947
	5	5.5057	5.3964	5.2440	5.6613	5.5114	5.4416	5.5266	5.5008
12	1	0.2234	0.2190	0.2128	0.2298	0.2237	0.2209	0.2243	0.2233
	2	0.2409	0.2361	0.2295	0.2477	0.2412	0.2381	0.2418	0.2407
	3	0.3538	0.3468	0.3370	0.3638	0.3542	0.3497	0.3551	0.3535
	4	2.2398	2.1954	2.1334	2.3031	2.2421	2.2138	2.2483	2.2379
	5	5.8736	5.7570	5.5944	6.0395	5.8797	5.8052	5.8958	5.8684
13	1	0.0000							
	2	0.0000							
	3	0.0000							
	4	0.0000							
	5	0.0000							
14	1	0.0226							
	2	0.0254							
	3	0.0317							
	4	0.1051							
	5	0.2875							
15	1	0.0000							

	2	0.0000
	3	0.0000
	4	0.0000
	5	0.0000
16	1	0.2092
	2	0.2092
	3	0.7762
	4	2.1930
	5	3.8163
17	1	0.2401
	2	0.2401
	3	0.8896
	4	2.5123
	5	4.3738
18	1	0.2795
	2	0.2795
	3	1.0369
	4	2.9312
	5	5.1039
19	1	0.3020
	2	0.3020
	3	1.1290
	4	3.1869
	5	5.5362
20	1	0.3735
	2	0.3735
	3	1.3846
	4	3.9144
	5	7.2250
21	1	0.4632
	2	0.4632
	3	1.7117
	4	4.8414
	5	8.9450
22	1	0.5303
	2	0.5303
	3	1.9931
	4	5.6571
	5	10.7413
23	1	0.5826
	2	0.5826
	3	2.2505

	4	6.4790
	5	12.1929
24	1	0.3461
	2	0.3461
	3	0.4665
	4	0.2935
	5	0.0000
25	1	0.2559
	2	0.2655
	3	0.3841
	4	2.4616
	5	6.5029
26	1	0.7736
	2	0.7736
	3	3.1092
	4	8.4900
	5	15.3574
27	1	0.5741
	2	0.5741
	3	2.1044
	4	5.9652
	5	10.7833
28	1	0.0984
	2	0.1152
	3	0.1305
	4	0.3308
	5	1.6607

Table 8-2: Particulate matter ($PM_{2.5}$) emission factors for in Canada, segregated into age bins 1-5 (0-2 years, 3-5 years, 6-9 years, 10-13 years, and >14 years).

Vehicle ID	Age Bin	Emission Factor (g/mi)
1	1	0.0121
	2	0.0121
	3	0.0121
	4	0.0209
	5	0.0913
2	1	0.0038
	2	0.0038
	3	0.0038
	4	0.0038
	5	0.0042

3	1	0.0091
	2	0.0091
	3	0.0091
	4	0.0610
	5	0.0900
4	1	0.0091
	2	0.0091
	3	0.0091
	4	0.0598
	5	0.0883
5	1	0.0091
	2	0.0091
	3	0.0091
	4	0.0578
	5	0.0858
6	1	0.0142
	2	0.0141
	3	0.0140
	4	0.0918
	5	0.1288
7	1	0.0193
	2	0.0192
	3	0.0191
	4	0.1473
	5	0.1898
8	1	0.0191
	2	0.0190
	3	0.0189
	4	0.1439
	5	0.1838
9	1	0.0240
	2	0.0238
	3	0.0237
	4	0.1821
	5	0.2331
10	1	0.0239
	2	0.0237
	3	0.0236
	4	0.1811
	5	0.2318
11	1	0.0325
	2	0.0323

	3	0.0322
	4	0.1377
	5	0.1711
12	1	0.0335
	2	0.0334
	3	0.0332
	4	0.1483
	5	0.1849
13	1	0.0083
	2	0.0083
	3	0.0083
	4	0.0171
	5	0.0913
14	1	0.0245
	2	0.0245
	3	0.0245
	4	0.0242
	5	0.0721
15	1	0.0164
	2	0.0164
	3	0.0163
	4	0.0413
	5	0.0485
16	1	0.0065
	2	0.0065
	3	0.0065
	4	0.0319
	5	0.0463
17	1	0.0066
	2	0.0066
	3	0.0066
	4	0.0314
	5	0.0456
18	1	0.0067
	2	0.0067
	3	0.0067
	4	0.0306
	5	0.0445
19	1	0.0105
	2	0.0105
	3	0.0105
	4	0.0638

	5	0.0871
20	1	0.0140
	2	0.0140
	3	0.0140
	4	0.0988
	5	0.1271
21	1	0.0139
	2	0.0139
	3	0.0139
	4	0.0968
	5	0.1230
22	1	0.0175
	2	0.0175
	3	0.0174
	4	0.1184
	5	0.1507
23	1	0.0201
	2	0.0201
	3	0.0200
	4	0.1465
	5	0.1873
24	1	0.0218
	2	0.0218
	3	0.0217
	4	0.0928
	5	0.1150
25	1	0.0243
	2	0.0243
	3	0.0241
	4	0.1001
	5	0.1236
26	1	0.0049
	2	0.0049
	3	0.0049
	4	0.0085
	5	0.0389
27	1	0.0189
	2	0.0189
	3	0.0189
	4	0.0118
	5	0.0049
28	1	0.0167

	2	0.0167
	3	0.0166
	4	0.0416
	5	0.0488

8.2.2 Emission factors – US

Table 8-3: Nitrogen oxide (NO_x) emission factors (in g/mi) for the United States, segregated by model year.

Age	MY	LDGV	LDGT1,2,3,4	HDGV2b,3	HDGV4,5	HDGV6,7	HDGV8a,b	LDDV	LDDT12,34	HDDV2b,3	HDDV4,5
0	2017	0.1201	0.3346	0.3519	0.4180	3.1138	3.0501	0.2329	0.9599	0.9597	0.9689
1	2016	0.1202	0.3350	0.3523	0.4184	3.1132	3.0495	0.2333	0.9617	0.9615	0.9707
2	2015	0.1202	0.3354	0.3527	0.4188	3.1131	3.0494	0.2336	0.9636	0.9633	0.9726
3	2014	0.1202	0.3358	0.3531	0.4192	3.1133	3.0496	0.2338	0.9655	0.9652	0.9746
4	2013	0.1202	0.3362	0.3536	0.4196	3.1138	3.0502	0.2339	0.9675	0.9672	0.9766
5	2012	0.1201	0.3366	0.3540	0.4201	3.1147	3.0513	0.2339	0.9695	0.9692	0.9786
6	2011	0.1203	0.3375	0.3550	0.4211	3.1240	3.0619	0.2339	0.9716	0.9714	0.9808
7	2010	0.1205	0.3385	0.3560	0.4223	3.1344	3.0735	0.2339	0.9740	0.9737	0.9832
8	2009	0.1300	0.3518	0.3692	0.4354	3.1447	3.0849	0.4502	1.9625	1.9621	1.9758
9	2008	0.1405	0.3671	0.3846	0.4507	3.1635	3.1062	0.4504	1.9655	1.9651	1.9788
10	2007	0.1515	0.4180	0.4395	0.5216	3.4546	3.3881	0.4508	1.9696	1.9692	1.9830
11	2006	0.1877	0.4577	0.4795	0.5623	3.5194	3.4571	0.4509	3.8823	3.8816	3.9063
12	2005	0.1984	0.5594	0.5807	0.6616	3.5328	3.4725	2.6390	5.0797	5.0800	5.0706
13	2004	0.2524	0.7138	0.7378	0.8288	3.7854	3.7212	2.6393	5.0818	5.0821	5.0727
14	2003	0.4914	1.3412	1.3644	1.4522	4.4355	4.4074	2.6403	5.0835	5.0838	5.0744
15	2002	0.5610	1.3135	1.3380	1.4308	4.5872	4.5744	0.9162	4.6382	4.6378	4.6532
16	2001	0.6300	1.3922	1.4206	1.5282	4.7272	4.7261	0.9182	4.5364	4.5410	4.3887

Age	HDDV6,7	HDDV8a,b	MC	HDGB	HDDBT	HDDBS
0	1.1052	2.2440	0.6770	2.7712	1.2624	1.1149
1	1.1053	2.2452	0.6769	2.7708	1.2621	1.1142
2	1.1055	2.2464	0.6767	2.7707	1.2622	1.1137

3	1.1057	2.2477	0.6764	2.7709	1.2624	1.1133
4	1.1064	2.2494	0.6760	2.7712	1.2639	1.1142
5	1.1917	2.3683	0.6753	2.7720	1.3718	1.1985
6	1.1943	2.3725	0.6840	2.7768	1.3738	1.1998
7	1.1986	2.3782	0.6877	2.7828	1.3773	1.2027
8	3.5602	5.4762	0.6901	2.7891	4.3771	3.0086
9	3.5602	5.4764	0.7101	2.7982	4.3770	3.0083
10	3.5618	5.4792	0.7752	3.7843	4.3789	3.0092
11	6.9592	10.0775	0.8318	3.8348	8.6818	5.6977
12	6.9586	10.0774	0.8303	3.8423	8.6812	5.6957
13	6.9576	10.0771	0.8811	4.6171	8.6805	5.6934
14	6.9568	10.0768	1.6218	5.2302	8.6795	5.6901
15	9.7078	17.3186	1.7772	5.3356	15.7215	9.0021
16	9.6484	17.4801	1.9070	5.4360	15.7201	8.9976

Table 8-4: Particulate matter ($PM_{2.5}$) emission factors (in g/mi) for the United States, segregated by model year.

Age	MY	LDGV	LDGT1,2,3,4	HDGV2b,3	HDGV4,5	HDGV6,7	HDGV8a,b	LDDV	LDDT12,34	HDDV2b,3	HDDV4,5
0	2017	0.007	0.0121	0.0121	0.0119	0.0085	0.0077	0.0049	0.0098	0.0098	0.0098
1	2016	0.007	0.0121	0.0121	0.0119	0.0085	0.0077	0.0049	0.0098	0.0098	0.0098
2	2015	0.007	0.0122	0.0121	0.0120	0.0085	0.0077	0.0049	0.0098	0.0098	0.0098
3	2014	0.007	0.0122	0.0121	0.0120	0.0085	0.0077	0.0049	0.0098	0.0098	0.0098
4	2013	0.007	0.0122	0.0121	0.0120	0.0085	0.0077	0.0049	0.0098	0.0098	0.0098
5	2012	0.007	0.0123	0.0122	0.0121	0.0085	0.0077	0.0049	0.0106	0.0106	0.0106
6	2011	0.007	0.0124	0.0123	0.0122	0.0086	0.0078	0.0049	0.0106	0.0106	0.0106
7	2010	0.0071	0.0125	0.0124	0.0123	0.0087	0.0079	0.0049	0.0107	0.0107	0.0107
8	2009	0.0071	0.0125	0.0125	0.0123	0.0088	0.0080	0.0049	0.0111	0.0111	0.0111
9	2008	0.0071	0.0126	0.0125	0.0124	0.0088	0.0080	0.0050	0.0112	0.0112	0.0112

10	2007	0.0071	0.0126	0.0125	0.0124	0.0088	0.0080	0.0060	0.0135	0.0135	0.0135
11	2006	0.0071	0.0127	0.0126	0.0125	0.0088	0.0080	0.0062	0.2111	0.2111	0.2124
12	2005	0.0071	0.0127	0.0126	0.0125	0.0087	0.0079	0.0068	0.2121	0.2120	0.2133
13	2004	0.0071	0.0127	0.0126	0.0125	0.0087	0.0079	0.0073	0.2128	0.2127	0.2140
14	2003	0.0069	0.0132	0.0131	0.0128	0.0077	0.0070	0.1210	0.2499	0.2499	0.2502
15	2002	0.0071	0.0126	0.0126	0.0123	0.0076	0.0070	0.1217	0.2779	0.2779	0.2784
16	2001	0.0081	0.0146	0.0145	0.0140	0.0114	0.0106	0.1224	0.2733	0.2736	0.2665

Age	HDDV6,7	HDDV8a,b	MC	HDGB	HDDBT	HDDBS
0	0.0154	0.0238	0.0352	0.0145	0.0220	0.0151
1	0.0154	0.0238	0.0352	0.0145	0.0220	0.0151
2	0.0154	0.0238	0.0352	0.0144	0.0220	0.0151
3	0.0154	0.0238	0.0352	0.0144	0.0220	0.0151
4	0.0154	0.0239	0.0352	0.0144	0.0220	0.0152
5	0.0169	0.0261	0.0352	0.0146	0.0242	0.0169
6	0.0169	0.0262	0.0352	0.0148	0.0242	0.0169
7	0.0171	0.0264	0.0351	0.0150	0.0243	0.0169
8	0.0180	0.0278	0.0351	0.0152	0.0255	0.0179
9	0.0182	0.0280	0.0351	0.0152	0.0256	0.0180
10	0.0217	0.0335	0.0350	0.0152	0.0306	0.0214
11	0.4274	0.6139	0.0350	0.0152	0.6256	0.4117
12	0.4279	0.6148	0.0351	0.0151	0.6256	0.4115
13	0.4288	0.6163	0.0351	0.0150	0.6260	0.4117
14	0.4298	0.6176	0.0352	0.0138	0.6258	0.4117
15	0.4570	0.6578	0.0352	0.0136	0.6811	0.4458
16	0.4577	0.6708	0.0353	0.0164	0.6798	0.4456

Table 8-5: Nitrogen oxide (NO_X) emission factors (in g/mi) for all vehicle types in the United States, segregated into age bins 1-5 (0-2 years, 3-5 years, 6-9 years, 10-13 years, and >14 years).

Age Group	1	2	3	4	5
LDGV	0.1202	0.1202	0.1278	0.1975	0.5608
LDGT1,2,3,4	0.335	0.3362	0.3487	0.5372	1.349
HDGV2b,3	0.3523	0.3536	0.3662	0.5594	1.3743
HDGV4,5	0.4184	0.4196	0.4324	0.6436	1.4704
HDGV6,7	3.1134	3.1139	3.1417	3.573	4.5833
HDGV8a,b	3.0497	3.0503	3.0816	3.5097	4.5693
LDDV	0.2333	0.2339	0.3421	1.545	1.4916
LDDT12	0.9617	0.9675	1.4684	4.0034	4.7527
HDDV2b	0.9615	0.9672	1.4681	4.0032	4.7542
HDDV4	0.9707	0.9766	1.4797	4.0082	4.7055
HDDV6	1.1053	1.1346	2.3783	6.1093	8.771
HDDV8a	2.2452	2.2885	3.9258	8.9278	14.9585
MC	0.6769	0.6759	0.693	0.8296	1.7687
HDGB	2.7709	2.7714	2.7867	4.0196	5.3339
HDDBT	1.2622	1.2993	2.8763	7.6056	13.3737
HDDBS	1.1143	1.142	2.1049	5.024	7.8966

Table 8-6: Nitrogen oxide ($PM_{2.5}$) emission factors (in g/mi) for all vehicle types in the United States, segregated into age bins 1-5 (0-2 years, 3-5 years, 6-9 years, 10-13 years, and >14 years).

Age Group	1	2	3	4	5
LDGV	0.007	0.007	0.0071	0.0071	0.0074
LDGT1,2,3,4	0.0121	0.0122	0.0125	0.0126	0.0135
HDGV2b,3	0.0121	0.0122	0.0124	0.0126	0.0134
HDGV4,5	0.0119	0.012	0.0123	0.0125	0.0131
HDGV6,7	0.0085	0.0085	0.0087	0.0088	0.0089
HDGV8a,b	0.0077	0.0077	0.008	0.008	0.0082
LDDV	0.0049	0.0049	0.0049	0.0066	0.1217
LDDT12	0.0098	0.0101	0.0109	0.1624	0.267
HDDV2b	0.0098	0.0101	0.0109	0.1623	0.2671
HDDV4	0.0098	0.0101	0.0109	0.1633	0.265
HDDV6	0.0154	0.0159	0.0175	0.3265	0.4482
HDDV8a	0.0238	0.0246	0.0271	0.4696	0.6487
MC	0.0352	0.0352	0.0351	0.0351	0.0352
HDGB	0.0145	0.0145	0.0151	0.0151	0.0146
HDDBT	0.022	0.0227	0.0249	0.477	0.6622

HDDBS	0.0151	0.0157	0.0174	0.3141	0.4344

8.2.3 PVHB output – Canada

Table 8-7: Estimates of the Per-Vehicle Health Benefit (through NO_X) for light-duty gasoline (LDGV) and light-duty diesel (LDDV) vehicles in Canada, segregated by city and age bins 1-5 (0-2 years, 3-5 years, 6-9 years, 10-13 years, and >14 years).

Vehicle ID	Age Bin	VMT (mi/vehicle-year)	PVHB - NO_X ($/vehicle-year)							
			VN	CG	QC	TO	HX	FM	WH	YK
1	1	12564.00	22	58	6	7	5	24	51	62
	2	11481.00	74	188	18	25	19	78	174	210
	3	10197.00	150	370	35	52	38	155	349	419
	4	7968.00	268	647	60	93	68	274	624	742
	5	7968.00	421	1015	94	147	107	430	982	1167
14	1	12564.00	20	49	4	6	5	20	48	60
	2	11481.00	20	51	5	7	5	21	50	62
	3	10197.00	22	56	5	7	6	23	55	69
	4	7968.00	58	146	13	19	15	60	143	178
	5	7968.00	159	399	35	52	41	164	391	487

Table 8-8: Estimates of the Per-Vehicle Health Benefit (through $PM_{2.5}$) for light-duty gasoline (LDGV) and light-duty diesel (LDDV) vehicles in Canada, segregated by city and age bins 1-5 (0-2 years, 3-5 years, 6-9 years, 10-13 years, and >14 years).

Vehicle ID	Age Bin	VMT (mi/vehicle-year)	PVHB – $PM_{2.5}$ ($/vehicle-year)							
			VN	CG	QC	TO	HX	FM	WH	YK
1	1	12564.00	73	36	3	4	3	27	26	25
	2	11481.00	67	33	2	4	3	24	24	23
	3	10197.00	59	29	2	3	3	22	21	20
	4	7968.00	80	40	3	5	4	29	28	28
	5	7968.00	350	173	13	20	16	128	123	120
14	1	12564.00	148	73	5	8	7	54	52	51
	2	11481.00	135	67	5	8	6	49	48	46
	3	10197.00	120	59	4	7	5	44	42	41
	4	7968.00	93	46	3	5	4	34	33	32
	5	7968.00	277	137	10	16	12	101	97	95

8.2.4 PVHB output – US

Table 8-9: Estimates of the Per-Vehicle Health Benefit (through NO_X) for light-duty gasoline (LDGV) and light-duty diesel (LDDV) vehicles in the United States, segregated by city and age bins 1-5 (0-2 years, 3-5 years, 6-9 years, 10-13 years, and >14 years).

City	Age Bin	VMT (mi/vehicle-year)	PVHB – NO_X ($/vehicle-year)	
			LDGV	LDDV
Houston	1	13851	36	70
	2	12042	31	61
	3	10741	30	79
	4	7401	32	247
	5	7401	90	239
Atlanta	1	13851	124	240
	2	12042	108	209
	3	10741	102	273
	4	7401	109	850
	5	7401	308	820
Los Angeles	1	13851	-306	-593
	2	12042	-266	-517
	3	10741	-252	-674
	4	7401	-268	-2099
	5	7401	-762	-2026
New York	1	13851	-47	-90
	2	12042	-40	-79
	3	10741	-38	-103
	4	7401	-41	-320
	5	7401	-116	-309

Table 8-10: Estimates of the Per-Vehicle Health Benefit (through $PM_{2.5}$) for light-duty gasoline (LDGV) and light-duty diesel (LDDV) vehicles in the United States, segregated by city and age bins 1-5 (0-2 years, 3-5 years, 6-9 years, 10-13 years, and >14 years).

City	Age Bin	VMT (mi/vehicle-year)	PVHB – $PM_{2.5}$ ($/vehicle-year)	
			LDGV	LDDV
Houston	1	13851	26	18
	2	12042	23	16
	3	10741	21	14
	4	7401	14	13
	5	7401	15	245
Atlanta	1	13851	23	16
	2	12042	20	14
	3	10741	18	12

	4	7401	12	11
	5	7401	13	213
Los Angeles	1	13851	30	21
	2	12042	26	18
	3	10741	24	17
	4	7401	16	15
	5	7401	17	282
New York	1	13851	104	73
	2	12042	90	63
	3	10741	81	57
	4	7401	56	52
	5	7401	58	964

8.3 Computer Codes

MOBILE6.2C Input file

```
                                        DIES_15
MOBILE6 INPUT FILE
*Each space (" ") acts as its own column and is important
*234567890123456789012345678901234567890123456789012345678901234567890
*
*This is a 2nd attempt
*This run will test the difference between data for Vancouver & T.O.
**********      Header Section  **********
POLLUTANTS          : NOX
PARTICULATES        : SO4 OCARBON ECARBON GASPM LEAD SO2 NH3
REPORT FILE         : DIES15.txt
SPREADSHEET         :
DATABASE OUTPUT     :
WITH FIELDNAMES     :
DAILY OUTPUT        :
DATABASE AGES       : 0,16
RUN DATA            :


**********      Run Section     **********
EXPAND BUS EFS      :
EXPAND EXHAUST      :

**********   Scenario Record    **********
SCENARIO RECORD     : Vancouver
PARTICULATE EF      : PMGZML.CSV PMGDR1.CSV PMGDR2.CSV PMDZML.CSV PMDDR1.CSV
PMDDR2.CSV
PARTICLE SIZE       : 2.5
FUEL RVP            : 6.53
*RVP (Vancouver=6.53 ; Toronto = 7.034)
DIESEL SULFUR       : 15.00
CALENDAR YEAR       : 2017
EVALUATION MONTH    : 7
HOURLY TEMPERATURES: 58.82 61.34 63.32 65.48 67.28 69.62 71.78 73.22 73.40 74.12
74.12 71.24
                     69.44 66.20 65.12 64.40 63.50 62.42 61.16 60.08 60.80 60.80
60.26 60.62
RELATIVE HUMIDITY   : 79. 70. 62. 55. 52. 47. 49. 43. 44. 44. 47. 50.
                     54. 70. 66. 53. 56. 59. 63. 67. 64. 65. 65. 68.
BAROMETRIC PRES     : 30.14
********** End of This Run      **********
END OF RUN          :


**********      Run Section     **********
EXPAND BUS EFS      :
EXPAND EXHAUST      :
```

Page 1

```
**********   Scenario Record    **********
SCENARIO RECORD    : Toronto
PARTICULATE EF     : PMGZML.CSV PMGDR1.CSV PMGDR2.CSV PMDZML.CSV PMDDR1.CSV
PMDDR2.CSV
PARTICLE SIZE      : 2.5
FUEL RVP           : 7.034
*RVP (Vancouver=6.53 ; Toronto = 7.034)
DIESEL SULFUR      : 15.00
CALENDAR YEAR      : 2017
EVALUATION MONTH   : 7
HOURLY TEMPERATURES: 65.48 66.92 69.08 70.70 73.58 76.10 78.08 78.62 80.60 78.98
81.68 80.60
                     79.88 77.72 74.30 69.98 68.00 67.46 65.66 66.38 63.68 64.58
63.32 63.68
RELATIVE HUMIDITY  : 78. 73. 67. 63. 57. 53. 49. 47. 49. 46. 44. 45.
                     47. 44. 40. 43. 50. 51. 54. 54. 72. 76. 84. 87.
BAROMETRIC PRES    : 29.36
********** End of This Run      **********
END OF RUN         :



**********      Run Section     **********
EXPAND BUS EFS     :
EXPAND EXHAUST     :

**********   Scenario Record    **********
SCENARIO RECORD    : Fort Mac
PARTICULATE EF     : PMGZML.CSV PMGDR1.CSV PMGDR2.CSV PMDZML.CSV PMDDR1.CSV
PMDDR2.CSV
PARTICLE SIZE      : 2.5
FUEL RVP           : 8.775
DIESEL SULFUR      : 15.00
CALENDAR YEAR      : 2017
EVALUATION MONTH   : 7
HOURLY TEMPERATURES: 53.42 55.76 60.98 67.28 70.16 73.94 75.20 79.34 81.14 84.02
87.44 89.24
                     90.14 88.70 85.10 80.78 75.56 73.58 72.32 70.16 68.54 69.44
65.48 63.32
RELATIVE HUMIDITY  : 94. 93. 83. 71. 69. 54. 50. 47. 44. 41. 36. 31.
                     26. 28. 34. 42. 53. 59. 62. 66. 70. 56. 66. 73.
BAROMETRIC PRES    : 28.55
********** End of This Run      **********
END OF RUN         :
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**********      Run Section     **********
EXPAND BUS EFS     :
EXPAND EXHAUST     :

**********   Scenario Record    **********
SCENARIO RECORD    : Whitehorse
PARTICULATE EF     : PMGZML.CSV PMGDR1.CSV PMGDR2.CSV PMDZML.CSV PMDDR1.CSV
PMDDR2.CSV
PARTICLE SIZE      : 2.5
FUEL RVP           : 7.76
DIESEL SULFUR      : 15.00
CALENDAR YEAR      : 2017
EVALUATION MONTH   : 7
HOURLY TEMPERATURES: 49.82 51.98 52.34 53.96 56.12 56.66 56.48 57.02 57.02 55.04
54.68 53.42
                     52.70 52.16 51.44 50.72 50.36 49.82 49.46 49.46 48.92 48.92
48.38 48.20
RELATIVE HUMIDITY  : 78. 75. 74. 72. 67. 68. 67. 71. 71. 79. 85. 90.
                     92. 93. 94. 94. 94. 93. 93. 92. 95. 93. 95. 94.
BAROMETRIC PRES    : 27.46
********** End of This Run      **********
END OF RUN         :



**********      Run Section     **********
EXPAND BUS EFS     :
EXPAND EXHAUST     :

**********   Scenario Record    **********
SCENARIO RECORD    : Yellowknife
PARTICULATE EF     : PMGZML.CSV PMGDR1.CSV PMGDR2.CSV PMDZML.CSV PMDDR1.CSV
PMDDR2.CSV
PARTICLE SIZE      : 2.5
FUEL RVP           : 7.76
DIESEL SULFUR      : 15.00
CALENDAR YEAR      : 2017
EVALUATION MONTH   : 7
HOURLY TEMPERATURES: 61.70 65.48 67.64 69.80 70.52 70.88 70.88 70.70 67.64 67.46
66.38 68.36
                     71.78 70.88 68.00 63.86 60.26 58.10 57.02 57.02 55.76 54.86
55.22 58.46
RELATIVE HUMIDITY  : 63. 56. 53. 46. 43. 45. 43. 44. 59. 62. 67. 58.
                     49. 50. 61. 74. 83. 89. 92. 93. 95. 98. 96. 91.
BAROMETRIC PRES    : 29.01
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********** End of This Run      **********
END OF RUN         :


**********      Run Section     **********
EXPAND BUS EFS     :
EXPAND EXHAUST     :

**********   Scenario Record    **********
SCENARIO RECORD    : Calgary
PARTICULATE EF     : PMGZML.CSV PMGDR1.CSV PMGDR2.CSV PMDZML.CSV PMDDR1.CSV
PMDDR2.CSV
PARTICLE SIZE      : 2.5
FUEL RVP           : 7.76
DIESEL SULFUR      : 15.00
CALENDAR YEAR      : 2017
EVALUATION MONTH   : 7
HOURLY TEMPERATURES: 58.10 61.34 64.76 67.82 70.52 71.42 73.76 75.20 77.00 77.90
77.90 79.88
                     80.42 77.72 74.30 72.32 70.16 66.92 64.22 62.42 61.88 57.56
60.44 56.30
RELATIVE HUMIDITY  : 71. 65. 52. 43. 44. 43. 42. 43. 42. 41. 34. 32.
                     33. 41. 49. 49. 51. 56. 61. 65. 67. 78. 71. 78.
BAROMETRIC PRES    : 26.44
********** End of This Run      **********
END OF RUN         :



**********      Run Section     **********
EXPAND BUS EFS     :
EXPAND EXHAUST     :

**********   Scenario Record    **********
SCENARIO RECORD    : Halifax
PARTICULATE EF     : PMGZML.CSV PMGDR1.CSV PMGDR2.CSV PMDZML.CSV PMDDR1.CSV
PMDDR2.CSV
PARTICLE SIZE      : 2.5
FUEL RVP           : 7.76
DIESEL SULFUR      : 15.00
CALENDAR YEAR      : 2017
EVALUATION MONTH   : 7
HOURLY TEMPERATURES: 55.40 56.66 57.38 61.70 66.38 69.08 72.50 72.68 73.22 72.32
72.14 70.16
                     68.90 64.58 61.88 61.16 60.80 60.62 60.62 60.62 60.44 60.62
60.98 61.16
RELATIVE HUMIDITY  : 85. 88. 88. 84. 72. 66. 57. 59. 65. 67. 67. 72.
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```
                    75. 85. 92. 96. 97. 98. 99. 100. 100. 100. 100. 100.
BAROMETRIC PRES    : 29.39
********** End of This Run      **********
END OF RUN         :



**********      Run Section     **********
EXPAND BUS EFS     :
EXPAND EXHAUST     :

**********   Scenario Record    **********
SCENARIO RECORD    : Quebec City
PARTICULATE EF     : PMGZML.CSV PMGDR1.CSV PMGDR2.CSV PMDZML.CSV PMDDR1.CSV
PMDDR2.CSV
PARTICLE SIZE      : 2.5
FUEL RVP           : 7.76
DIESEL SULFUR      : 15.00
CALENDAR YEAR      : 2017
EVALUATION MONTH   : 7
HOURLY TEMPERATURES: 62.06 61.70 61.52 64.04 66.20 67.46 70.34 72.68 74.66 75.20
76.10 76.10
                    75.20 72.68 70.16 67.64 67.28 65.48 64.22 62.96 60.26 61.34
64.04 64.58
RELATIVE HUMIDITY  : 89. 93. 95. 90. 85. 82. 80. 74. 69. 67. 64. 66.
                    69. 73. 80. 87. 87. 92. 93. 94. 95. 96. 95. 92.
BAROMETRIC PRES    : 29.55
********** End of This Run      **********
END OF RUN         :
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