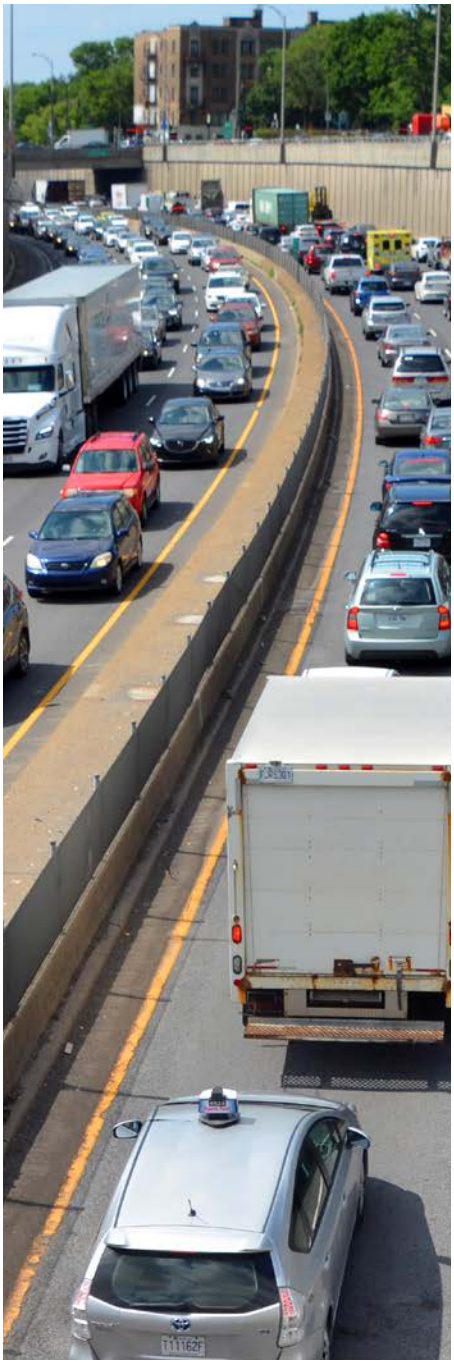

HEALTH IMPACTS OF TRAFFIC-RELATED AIR POLLUTION IN CANADA

Health Canada Santé Canada

Canada

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To obtain additional information, please contact:

Health Canada
Address Locator 0900C2
Ottawa, ON K1A 0K9
Tel.: 613-957-2991
Toll free: 1-866-225-0709
Fax: 613-941-5366
TTY: 1-800-465-7735
E-mail: publications-publications@hc-sc.gc.ca

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LIST OF ABBREVIATIONS

ADOM	Acid Deposition Oxidant Model
APEI	Air Pollutant Emissions Inventory
AQBAT	Air Quality Benefits Assessment Tool
AURAMS	A Unified Regional Air Quality Modelling System
CAD	Canadian dollar
CD	Census division
CM	Crustal material
CO	Carbon monoxide
COPD	Chronic obstructive pulmonary disease
CRF	Concentration response function
CTM	Chemical transport model
EC	Elemental carbon
ECCC	Environment and Climate Change Canada
GEM	Global Environmental Multiscale (weather forecast model)
GEM-MACH	Global Environmental Multiscale–Modelling Air Quality & Chemistry
GIS	Geographic information system
GTHA	Greater Toronto and Hamilton Area
HDV	Heavy-duty vehicle
HDV8	Class 8 heavy-duty vehicle
HEI	Health Effects Institute
HPC	High Performance Computing
IHD	Ischemic heart disease
IHME	Institute for Health Metrics and Evaluation
LDDT	Light-duty diesel truck
LDT	Light-duty truck
LDV	Light-duty vehicle
µg/m^3	Micrograms per cubic metre
MBE	Mean bias error
MOVES	Motor Vehicle Emission Simulator
NH_3	Ammonia
NO_2	Nitrogen dioxide
NO_3	Ammonium

O_3	Ozone
$PM_{2.5}$	Fine particulate matter
POA	Primary organic aerosol
ppb	Part per billion by volume
ppm	Part per million
RMSE	Root mean square error
SMOKE	Sparse Matrix Operator Kernel Emissions
SO_2	Sulphur dioxide
SOA	Secondary organic aerosol
TRAP	Traffic-related air pollution
TWBL	Tire wear and brake lining
US EPA	United States Environmental Protection Agency
VAQUM	Verification of Air Quality Models
VOC	Volatile organic compound
WHO	World Health Organization
WTP	Willingness to pay

EXECUTIVE SUMMARY

A large body of scientific evidence has accumulated over the past 25 years attributing a wide range of adverse health effects to ambient (outdoor) air pollution exposure. These effects range in severity from respiratory symptoms to the development of disease and premature death. For example, exposure to airborne particles, a component of smog, increases the risk of premature mortality from heart disease, stroke and lung cancer.

On-road vehicles contribute to air pollution through fuel combustion and evaporative emissions as well as emissions from tire and brake wear. Canadians are regularly exposed to traffic-related air pollution (TRAP), most notably in high-traffic areas such as near highways and urban centres. TRAP consists of a complex and variable mixture of particulate and gaseous pollutants that contribute to smog, including fine particulate matter ($PM_{2.5}$) and ozone (O_3).

The objective of this report is to present modelled estimates of population health impacts and socio-economic costs associated with exposure to TRAP in Canada for the year 2015, specifically the contribution from Canadian on-road vehicle emissions to $PM_{2.5}$, NO_2 and O_3 ambient concentrations in Canada. The year 2015 was selected for modelling based on data availability and quality considerations. Results at the national, provincial and territorial, and census division level are presented and discussed. The report is intended to inform Canadian authorities on the air quality and health impacts associated with on-road vehicle activity.

This report estimates that TRAP was associated with over 1,200 premature deaths in Canada in 2015. Of these, it was estimated that exposure to $PM_{2.5}$, NO_2 and O_3 contributed to 800, 340 and 85 premature deaths, respectively. Non-fatal health outcomes included 2.7 million acute respiratory symptom days, 1.1 million restricted activity days and 210,000 asthma symptom days per year. The total annual monetary value of the health burden was estimated at $9.5 billion (CAD 2015), with $9 billion being associated with premature deaths. Analysis also found that light-duty vehicles (e.g., passenger vehicles) contributed to approximately 37% of premature deaths, while heavy-duty vehicles (e.g., commercial trucks and buses) contributed to approximately 63% of premature deaths. In terms of the geographic distribution of the air pollution burden, the results indicated greater adverse health impacts in more populous provinces and census divisions (CDs): 500 premature deaths were estimated in Ontario, 410 in Quebec, 170 in British Columbia and 82 in Alberta. At the CD level, 170 premature deaths were estimated in Toronto, 150 in Montreal, and 110 in Vancouver. The results from this analysis are available for all CDs and can be obtained upon request from Health Canada.

The air quality modelling for this analysis was conducted at a grid resolution of 10 km, which is a reasonably high resolution for a national assessment. However, 10-km grid cells are too coarse to capture the local variability and magnitude in exposures to TRAP, such as those experienced by populations near high-traffic roadways. This limitation likely leads to an underestimation of population exposure to TRAP and health impacts, so that the values herein possibly represent a lower range estimate.

This analysis contributes to our understanding of the health burden of exposure to TRAP in Canada. It also complements recent and ongoing activities on TRAP at Health Canada, including health risk assessments and an evaluation of exposure to TRAP in Canada. Together, these analyses are intended to provide a comprehensive national evaluation of TRAP in Canada.

CHAPTER 1: **INTRODUCTION**

A large body of scientific evidence has accumulated over the past 25 years attributing a wide range of adverse health effects to ambient (outdoor) air pollution exposure. These effects range in severity from respiratory symptoms to the development of disease and premature death. For example, exposure to airborne particles, a component of smog, increases the risk of premature mortality from heart disease, stroke and lung cancer. In Canada and internationally, health impact assessments identify air pollution as one of the top risk factors for premature mortality and several non-fatal outcomes (WHO 2016). Exposure to air pollution is currently considered the fifth leading mortality health risk in the world and, in 2017, was responsible for 4.9 million premature mortalities, or 8.7% of all deaths globally (IHME and HEI 2019). Burnett et al. (2018) attributed 8.9 million deaths in 2015 to exposure to ambient fine particulate matter ($PM_{2.5}$) alone.

Health Canada estimated that 15,300 premature deaths in Canada were associated with ambient air pollution exposure in 2016 (Health Canada 2021). For their analysis, Health Canada defined air pollution as the above-background increment of ambient $PM_{2.5}$, nitrogen dioxide (NO_2) and ozone (O_3) concentrations that can be attributed to anthropogenic North American sources of emissions (e.g., fuel combustion, industry, etc.), as well as emissions from natural events such as wildfires. Non-fatal outcomes were also estimated, including 2.7 million asthma symptom days per year and 35 million acute respiratory symptom days, with the total economic cost of all health impacts attributable to air pollution in 2016 Canadian dollars (CAD 2016) being $120 billion (Health Canada 2021).

The contribution from on-road vehicles to ambient air pollution is referred to as traffic-related air pollution (TRAP). Most Canadians experience exposure to TRAP on a daily basis, particularly in high-traffic areas such as near highways and in urban centres (Brauer et al. 2012, 2013; Matz et al. 2018). TRAP consists of a mixture of particulate and gaseous pollutants associated with combustion (i.e., vehicle exhaust) and non-combustion (e.g., fuel evaporation, abrasion wear) processes. TRAP can influence ambient concentrations of several air pollutants, including $PM_{2.5}$ and O_3 that are responsible for smog. Owing to its complex and variable composition, the TRAP mixture cannot be measured directly. It is generally estimated using a limited number of measurable surrogates, such as NO_2 or ultrafine particulate matter, which involves some uncertainties.

The objective of this report is to present estimates of population health impacts and costs associated with exposure to TRAP in Canada. The results are intended to inform Canadians and provincial, territorial and regional stakeholders of the air quality and health impacts associated with on-road vehicle activity. This analysis focuses on the contribution of Canadian on-road vehicle emissions to ambient concentrations of $PM_{2.5}$, NO_2, O_3 and sulphur dioxide (SO_2) for the year 2015, and the associated health impacts for $PM_{2.5}$, NO_2 and O_3, including premature death and morbidity, in Canada. Traffic is defined as on-road transportation and corresponds to the category *on-road vehicles* in Canada's Air Pollutant Emissions Inventory (APEI). These include light-duty vehicles and trucks, motorcycles, buses, and medium-duty and heavy-duty vehicles. Vehicles used for off-road applications, such as agricultural and mining equipment, are excluded from the on-road category.

Health Canada collaborated with Environment and Climate Change Canada (ECCC) to estimate the ambient air pollution attributable to TRAP across Canada, interpreted herein as the exposure to TRAP. To estimate exposure, the Global Environmental Multiscale–Modelling Air Quality and Chemistry (GEM-MACH) model was used. The air quality modelling for this analysis was conducted at a grid resolution of 10 km. The air quality modelling results were then averaged over Canadian census divisions (CDs), which vary in size. The modelling approach and regional resolution were selected for this national assessment. It does not capture the local variability and magnitude in exposures to TRAP, such as those experienced by populations near high-traffic roadways.

Health Canada used the output from the air quality modelling to estimate TRAP population health impacts as well as the monetary value of the related health burden across Canada using the Air Quality Benefits Assessment Tool (AQBAT). This modelling-based analysis considered exposure to ambient levels of $PM_{2.5}$, O_3 and NO_2. Air quality and health impacts associated with other toxic air pollutants in TRAP, including SO_2, carbon monoxide (CO), organic compounds (e.g., volatile organic compounds such as formaldehyde and benzene, polycyclic aromatic compounds), ultrafine particulate matter (UFP) or specific $PM_{2.5}$-associated compounds (e.g., metals, polycyclic aromatic compounds), were not included. The health impact assessment provides results at the national, provincial, territorial and census division levels. As such, the health impact estimates reflect average regional air pollution exposures rather than local or high pollution microenvironment exposures, such as in near-road environments.

The current analysis complements and updates previous health impact analyses for diesel- and gasoline-related air pollution published by Health Canada (2016a, 2017). It uses updated data and modelling tools to investigate the health impacts of all on-road sources and fuel types. The analysis also complements risk assessment activities at Health Canada conducted to inform and support programs and policies designed to mitigate exposure to, and health impacts of, air pollution in Canada. This includes a report evaluating the role of exposure to TRAP in asthma, allergy, and lung function using a weight of evidence approach (Health Canada 2020). Health Canada is also assessing exposure to TRAP in Canada based on a literature review and an analysis of population proximity to roadways. Together, the Health Canada assessments of health risks, population health burden, exposures associated with TRAP in Canada are intended to provide a comprehensive national evaluation.

This report summarizes the methodological approach and the results for each step of the health impact modelling framework, including the air pollutant emissions inventory, the air quality (i.e., ambient air concentrations) modelling simulations, the health impact assessment, and the estimate of socio-economic costs (valuation). Results at the national, provincial and territorial, and census division level are presented and discussed.

CHAPTER 2: **METHODOLOGY**

The air pollution and health burden estimates are based on a modelling framework designed for the analysis of source sectors, including on-road vehicles. The modelling framework for this analysis utilizes three main steps and tools that are discussed in sections 2.1 to 2.3:

1. Air pollutant emissions inventory;
2. Chemical transport modelling of air pollution scenarios; and
3. Health impact assessment and valuation.

The general approach was to develop an emissions inventory for the year 2015 (step 1), which included detailed data for traffic and other source sectors of air pollution. In step 2, ambient air pollutant concentrations were estimated using a chemical transport model (CTM). Concentrations attributable to TRAP were determined by comparing the air quality estimates from the CTM for a reference emissions scenario and for a scenario involving complete removal of on-road vehicle emissions. The difference between the two scenarios corresponded to the estimate of the contribution of TRAP to ambient air pollution concentrations. In step 3, TRAP concentrations were input into the Air Quality Benefits Assessment Tool (AQBAT) to estimate the population health burden attributable to TRAP in Canada, including mortality and morbidity outcomes, as well as socio-economic costs. Additional methodological details of the modelling framework are presented in Appendix A.

2.1. EMISSIONS INVENTORY

ECCC developed a detailed emissions inventory for the year 2015, based on the 2017 Air Pollutant Emissions Inventory, to be used as input to the chemical transport model.[1] It was the most recent and accurate version available when the modelling analysis was initiated. The emissions inventory used for the United States was the 2017 projection emissions inventory based on the United States Environmental Protection Agency (US EPA) 2011 National Emissions Inventory (NEI).[2] The 2008 inventory used for Mexico (Inventario Nacional de Emisiones, INEM)[3] was also from the US EPA 2011 NEI. The emission estimates were chemically speciated, spatially distributed, and temporally resolved. ECCC used different approaches and data to develop the emissions inventory, including industrial pollutant release data, fuel use data, source-specific activity data and emission factors (ECCC 2015, 2016). Primary emissions were estimated for both anthropogenic and biogenic source categories (except emissions from wildfires). Additional methodological details of emissions modelling are presented in Appendix A.

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[1] The emissions inventory represents air pollutant emissions in 2015, based on the version of the Air Pollutant Emissions Inventory published in 2017.

[2] www.epa.gov/air-emissions-inventories/2011-national-emissions-inventory-nei-data

[3] www.gob.mx/semarnat/documentos/documentos-del-inventario-nacional-de-emisiones

On-road vehicle emissions were estimated using the Motor Vehicle Emission Simulator (MOVES) version 2014b, developed by the US EPA. The model has been modified to reflect Canadian conditions, such as vehicle population and age distribution, as well as vehicle emission standards. MOVES provides emission rates for a series of pollutants, including CO, ammonia (NH_3), nitrogen oxides (NO_X), PM, SO_2 and volatile organic compounds (VOCs), which are then combined with vehicle activity data. Traffic source emissions modelled by MOVES and considered in this analysis included exhaust, evaporative and tire wear and brake lining (TWBL) emissions from gasoline and diesel on-road vehicles, as well as on-road vehicles powered by other fuels (e.g., compressed natural gas). Evaporative emissions from stationary sites, such as those from refuelling stations or fuel storage facilities, were not included as a component of TRAP; they are associated with air pollutant emissions from commercial, institutional, or industrial sources. Dust emissions from paved and unpaved roads were also excluded as a component of TRAP; they are associated with open sources[4] in the APEI and are not allocated to specific segments of the vehicle fleet. On-road transportation emissions were compiled as monthly values for different vehicle types and aggregated at the provincial level, except for British Columbia and Ontario. Sub-provincial vehicle information was available for these two provinces (e.g., inspection and maintenance programs), allowing the creation of sub-regions (two for British Columbia and four for Ontario). Five different road classes were also used to improve the spatial allocation of on-road vehicle emissions: rural restricted and unrestricted access, urban restricted and unrestricted access, and off-network. For on-road transportation emissions specifically, Canadian engine emission standards and fuel regulations effective in 2015 were considered.[5]

The 2015 inventory database was further processed by ECCC to generate a dataset that can be used in the Sparse Matrix Operator Kernel Emissions (SMOKE) model (Sassi et al. 2016). Spatial surrogates from SMOKE version 3.7 (in ASCII format), generated from geographic information system (GIS) shapefiles, were used to allocate emissions to a national grid surface. Ancillary and updated shapefiles were also included to distribute emissions across Canada. The spatially allocated emissions became input to the chemical transport model.

2.2. AIR QUALITY MODELLING

2.2.1. Chemical transport model

The chemical transport model used for this assessment was ECCC's Global Environmental Multiscale–Modelling Air Quality and Chemistry (GEM-MACH) model, version 2.3.1. GEM-MACH, a source-oriented Eulerian model, is a prognostic tool that integrates meteorological data and specific algorithms to simulate the diffusion, transport and chemical transformation of gases and particles in the atmosphere (Makar et al. 2018; Moran et al. 2010; Whaley et al. 2018, 2020). GEM-MACH is used by ECCC to simulate hourly concentrations of air pollutants including O_3, $PM_{2.5}$, NO_2, and SO_2 over a North American domain, for operational air quality forecasting and in support of air quality regulations and management decisions for Canada.

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[4] Open sources include emissions from agriculture activities (tilling, wind erosion, fertilizer application, animals), construction operations, dust from paved and unpaved roads, mine tailings, and waste treatment or management (landfills, open burning, water and sewage treatment, energy from waste)

[5] Gasoline fuels with low sulphur (25 ppm) and benzene (less than 1%) content and no ethanol content were selected for mobile source emissions modelling. The sulphur content of ultra-low sulphur diesel was set at 10 ppm, and no biodiesel content was included.

The general air quality modelling approach included simulations conducted over a continental domain with a horizontal grid spacing of 10 km by 10 km, and 80 hybrid vertical levels extending from the surface (1.5 m) to 0.1 hectopascal (approximately 30 km). GEM-MACH simulations were done for a complete year, computing 24 hours of forecast for each day. The model's time step is 300 seconds, such that meteorological and atmospheric conditions are computed every five minutes. Hourly output values (i.e., pollutant concentrations) are simulated. Simulations were conducted using 2017 meteorology that corresponds to forecasts produced operationally on a daily basis by ECCC in 2017. It includes data assimilation of observations.

The GEM-MACH model was executed on ECCC's High Performance Computing (HPC) system. The simulation was divided into four three-month segments, each with a 21-day spin-up period. The segments correspond to January–March, April–June, July–September and October–December. All segments are computed in parallel to decrease the execution time. A task sequencer is used to manage and submit on the HPC the different steps required to perform an uninterrupted simulation.

The post-processing of GEM-MACH results consists of computing different statistics by using an automated suite of tools, such as Kornshell and Tcl scripts, as well as C/C++ and Fortran programs. The statistics include, for example, averages, rolling averages, maximum values and differences between scenarios, for different periods (monthly, annual and seasonal). Statistics are computed for the model grid and can be interpolated on different geographical areas, such as census divisions (CDs) as used in AQBAT. The final products are available in formats such as ASCII, binary, GIS, and graphical format, depending on the research objectives.

From the gridded modelled results, ambient air pollution estimates were generated at the CD level for NO_2, O_3, $PM_{2.5}$ and SO_2 for linkage with AQBAT. The results represent regional air pollution conditions and do not reflect local effects and microenvironments that are unresolved by the horizontal grid size, such as roadways and street canyons, where air pollution levels may be higher. Appendix A presents additional methodological details of the GEM-MACH model. Methodological considerations are discussed in section 4.3.

2.2.2. Simulation approach

In order to isolate the contribution of TRAP to ambient pollutant concentrations in Canada, a brute force approach was used wherein two air quality scenarios were modelled with GEM-MACH: (1) a reference scenario with the full Canadian 2015 emissions inventory and (2) a TRAP scenario with on-road vehicle emissions removed from the inventory. The TRAP scenario required zeroing-out all Canadian air pollutant emissions in Canada from motorcycles, light-duty vehicles and heavy-duty vehicles for the calendar year 2015. The differences in air quality between the full emissions inventory scenario and the scenario with Canadian on-road vehicle emissions removed are assumed to represent the contribution to ambient pollutant levels from on-road vehicles in Canada. This increment is also used to estimate the population exposure to TRAP.

2.2.3. Model output

Ground-level concentrations were estimated for each model grid cell and Canadian CD for the following pollutants and concentration metrics:

- Annual average based on:
  - hourly data for NO_2, $PM_{2.5}$, and SO_2
  - daily maximum of hourly data for O_3
- Summer (i.e., May–September) average based on daily maximum of hourly data for O_3

The units for NO_2, O_3 and SO_2 are in parts per billion by volume (ppbv), while $PM_{2.5}$ values are in micrograms per cubic metre ($\mu g/m^3$). For the current analysis, estimates of ground-level concentrations were not available for all air pollutants in the emissions inventory, including CO and PM_{10}.

Concentration estimates are reported for each province and territory as well as for some individual CDs (section 3.2). For the current analysis, the CDs were based on the Standard Geographical Classification 2011, which divides Canada into 293 CDs.[6] The CD air pollutant concentrations are area-weighted estimates from the original GEM-MACH grid. Area-weighted concentrations for each CD were determined by summing the product of a grid cell concentration and the area of that grid cell occupied by the CD, for all grid cells intersecting with the CD, and then dividing the sum by the area of the CD. For example, if three grid cells intersect with a CD, the following formula would apply to determine its concentration (C_d):

$$C_d = (A_{d1} \times C_{g1} + A_{d2} \times C_{g2} + A_{d3} \times C_{g3}) \div A_d$$

where A_{dx} is the area of overlap between the model grid cells and the CD, C_{gx} is the concentration of the grid cell g*x*, and A_d is the area of the CD.

The reported national, provincial and territorial concentrations correspond to population-weighted estimates. The population-weighting method estimates the average exposure concentration for an individual within a geographic unit. When averaged across larger geographic units, CDs with high populations have more influence or weight than CDs with low populations. Compared to an area-weighted measure, population-weighting provides a better representation of the average pollutant concentration to which Canadians are exposed. This is especially important in Canada as populations are generally limited to urban areas in the southern portion of the country, compared with the vast rural and low population northern areas.

[6] www.statcan.gc.ca/eng/subjects/standard/sgc/2011/sgc-intro#a5-1

Population-weighted concentrations for all provinces and territories were determined by summing the product of a CD concentration and the population of that CD, for all CDs in a province, and then dividing the sum by the provincial population. For example, if a province includes three CDs, the population-weighted concentration (C_{pw}) is determined by:

$$C_{pw} = (CD_{d1} \times CD_{pop1} + CD_{d2} \times CD_{pop2} + CD_{d3} \times CD_{pop3}) \div PT_{pop}$$

where CD_{dx} is the concentration of CD_x, CD_{popx} is the population of CD_x, and PT_{pop} is the population of the province. The same method was used to estimate the population-weighted national average.

2.3. HEALTH IMPACT ASSESSMENT

2.3.1. Calculating population health impacts due to air pollution

The health outcomes attributable to air pollution from the on-road sector were estimated using Health Canada's Air Quality Benefits Assessment Tool (AQBAT) (Judek et al. 2019), version 3.0.[7] AQBAT estimates the number of premature deaths and other adverse health outcomes in Canada associated with a specified change in ambient concentrations of NO_2, O_3 and $PM_{2.5}$. In AQBAT, health effect information for the three air pollutants is included in the form of concentration response functions (CRFs), which describe the association between exposure to an air pollutant and a health response. A CRF represents the excess health risk for a given endpoint (e.g., asthma symptoms, chronic bronchitis, and acute exposure mortality) associated with exposure to a unit increase in ambient pollutant concentration. For example, an increase in $PM_{2.5}$ chronic exposure of 10 μg/m^3 leads to a corresponding 10% increase in the risk of premature mortality from non-accidental causes. CRFs in AQBAT are statistically derived estimates from a single study or a meta-analysis of multiple studies.[8]

Health endpoints related to acute or chronic exposure, the associated CRFs, and the applicable population groups (e.g., age-specific groups) are predefined within AQBAT and represent Health Canada-endorsed values drawn from the peer-reviewed health science literature. AQBAT also includes a CRF for mortality associated with short-term exposure to SO_2. However, it was not included in the current analysis to reflect the Health Canada risk assessment for SO_2, which concluded that data are only suggestive of a causal relationship between short-term SO_2 exposure and all-cause and cardiopulmonary mortality at ambient concentrations (Health Canada 2016b). Health Canada only considers causal and likely causal relationships for quantitative analyses. In the context of this analysis, short-term exposure contributes to effects that occur within a few days of an increase in ambient air pollution (acute health effects), while long-term exposure refers to exposures averaged over the years preceding the development of disease or death (chronic health effects). CRFs pertaining to acute exposure were derived from studies examining effects of air pollutants in the days before health outcomes, whereas CRFs pertaining to chronic exposure were derived from studies of air pollutants averaged over the years prior to health outcomes. The pollutants and their associated health effects considered in this analysis are provided in Table A1, Appendix A. Previous

[7] Guoliang Xi and Dave Stieb, personal communications, Health Canada, 2019

[8] The version of AQBAT used for the current analyses used linear CRFs. However, linear and non-linear CRFs have been reported in the health science literature and can be used in AQBAT.

studies (Crouse et al. 2012; Judek et al. 2019; Shin et al. 2013; Stieb et al. 2015) contain background information on the CRF estimates used in this analysis (i.e., references to the scientific literature upon which the risk estimates are based) and the analysis undertaken to produce the estimates within AQBAT. Health outcomes were considered to have no threshold for effect (i.e., effects were assumed to occur at all levels of exposure).

CRFs can be input as a distribution function in AQBAT, accounting for inherent uncertainty in the CRF estimates. Monte Carlo simulations employing 10,000 iterations were used to propagate this uncertainty in the CRFs. The model generates a central estimate of the most likely health impacts equal to the median of the output distribution, as well as low- and high-end estimates equal to the 2.5 and 97.5 percentiles of the output distribution (interpreted as the 95% confidence intervals).

Baseline incidence rates in the Canadian population for each health endpoint (e.g., risk of death due to cardiovascular disease) are needed to estimate counts of health outcomes in a target population. General population and age-specific baseline incidence rates for a target population are included in AQBAT. For example, the *Restricted Activity Days* endpoint is assigned to 100% of all adults (20 years of age and older) and 85.7% of children aged 5 to 19 years (non-asthmatic). For each morbidity and mortality health endpoint in AQBAT, the baseline incidence rates are represented by a data file containing estimated annual events per million specified population, for every geographic area, age group, scenario year and population projection. Baseline incidence rates are estimated by Health Canada from detection, observation and reporting through formal means (e.g., death certificates, hospital admission records), from data provided by Statistics Canada, the Canadian Institute for Health Information, or from epidemiological studies. Incidence rates are generally associated with many factors, such as age, gender, race, education, income, environmental factors and lifestyle habits. Exposure to pollutants typically has a minor influence on the baseline incidence rates. Additional details and references on the process of deriving baseline rates have previously been published (Judek et al. 2019; Stieb et al. 2015).

2.3.2. Calculating the economic value of health outcomes due to air pollution

Estimating the economic value (or valuation) of air pollution-related health impacts monetizes health outcomes, allowing impacts to be expressed in monetary units. In doing so, the potential social, economic and public welfare consequences of a health endpoint are considered, including medical costs, reduced workplace productivity, pain and suffering, and the other effects of increased health risks. Expressing impacts in monetary terms provides a common metric across health endpoints to estimate the overall benefits or damages in order to inform air quality management strategies. The sum provides an indication of the relative benefits or damages to society resulting from reduced or increased risks to health.

In AQBAT, each health endpoint is assigned a monetary value, typically derived from survey, accounting, economic or actuarial data. The endpoint values have inherent uncertainties, which are captured by a distribution of possible values with corresponding parameters (i.e., valuation estimates are entered as a distribution in AQBAT). The valuation estimates used in the model and references to the studies from which they are derived are provided in Table A2 in Appendix A. Endpoint valuations are expressed in Canadian dollars and can be temporally adjusted from the source years of the underlying studies based on the consumer price index, as defined by Statistics Canada (Judek et al. 2019; Statistics Canada, annual). In the current analysis, data are reported in 2015 Canadian dollars (CAD 2015).

The monetary value of mortality is considerably higher than that of any other health endpoint. For the purpose of policy analysis, the recommended central estimate of an avoided premature death is $6.5 million (CAD 2007) based on a review of Canadian studies by Chestnut and De Civita (2009). It relies on analyses indicating that an average Canadian would be willing to pay approximately $65 in order to reduce the risk of premature death by 1 out of 100,000. The aggregate willingness to pay (WTP) of 100,000 Canadians ($65 each) equals the value of the one avoided death. The uncertainty in this estimate is addressed by a recommended low value of $3.5 million and a high value of $9.5 million. These values represent a reasonable range for a primary analysis but should not be considered as lower and upper bounds (Chestnut and De Civita 2009). The above values are not equivalent to the economic worth of an identified person's life, but rather an aggregation of individuals' willingness to pay for small changes in risk.[9] Following adjustments based on the consumer price index, the value of an avoided premature death in 2015 CAD is $7.4 million.

[9] Empirical studies of willingness to pay (WTP) for mortality risk reductions estimate average monetary amounts that individuals are willing to pay for small reductions in premature mortality. The valuation context or an individual's circumstances influence his WTP values—that is, they may vary for the same amount of risk reduction in different contexts and for different individuals. WTP reflects all the reasons individuals put a value on to reduce their own risk of death. Therefore, it can exceed the value of the financial impact to an individual associated with the change in risk.

CHAPTER 3:
RESULTS

3.1. EMISSIONS ASSOCIATED WITH ON-ROAD TRANSPORTATION

Table 1 presents the Canadian emissions inventory for the year 2015.[10] The major source categories are included as well as a more detailed listing of transportation classes (i.e., air, marine, off-road, on-road and rail) and sub-classes for on-road transportation (e.g., class 8 heavy-duty vehicles (HDV8), light-duty trucks (LDT), and light-duty vehicles (LDV)) and open sources, which include agricultural and construction activities as well as emissions from paved and unpaved roads. This inventory was used for the reference scenario during the chemical transport model simulations (step 2). The TRAP scenario used the same inventory but excluded on-road transportation emissions.

Overall, the emissions inventory data show that:

- The transportation sector emits large quantities of air pollutants, particularly CO and NO_X.
- On-road vehicles are the largest contributors to CO emissions among the transportation classes, followed by off-road vehicles.
- On-road vehicles are the largest contributors to NO_X emissions among the transportation classes, followed by marine vessels.
- Open sources are the largest contributors to NH_3 and PM emissions.
- Industrial and non-industrial (e.g., electricity generating units) facilities release most of the SO_2 emissions, while industrial facilities also notably contribute to VOC emissions.

Within the transportation category, on-road vehicles release more NH_3, NO_X and PM_{10} than other classes. The large number of vehicles in operation on Canadian roads explains the higher emissions for some pollutants (e.g., NO_X). For example, in 2015, almost 24 million road motor vehicles were registered in Canada, compared with 2.1 million off-road engines and vehicles.[11] Off-road engines and vehicles are associated with the highest contributions to $PM_{2.5}$ and VOC emissions. The less stringent off-road engine emission standards and the characteristics of the off-road engine population (e.g., older model year engines, engines without after-treatment devices) are drivers of high $PM_{2.5}$ and VOC emissions from this class. Marine transportation emits the largest share of SO_2 among transportation classes, with the high SO_2 marine emissions being linked to the much higher sulphur content of marine diesel fuel. In 2015, the maximum allowable sulphur content of marine diesel fuel was 1,000 ppm, compared to 15 ppm for on-road, off-road and rail transport.[12]

[10] 2015 emissions inventory developed by ECCC; internal reference BASE 2015 ref17 (November 2019).

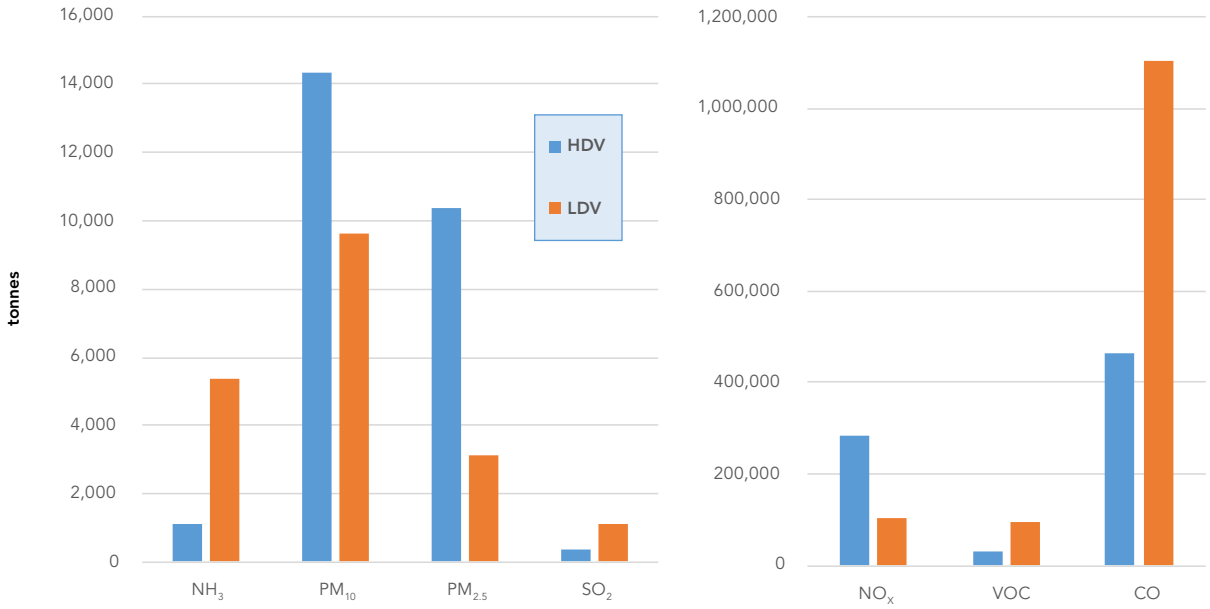
[11] Statistics Canada, annual. Vehicle registrations, by type of vehicle. Table 23-10-0067-01. www150.statcan.gc.ca/t1/tbl1/en/tv.action?pid=2310006701

[12] Sulphur in Diesel Fuel Regulations (SOR/2002-254). https://laws-lois.justice.gc.ca/eng/regulations/sor-2002-254/index.html

Within the on-road transportation class, light-duty vehicles and trucks are the largest emitters for CO, NH_3, SO_2 and VOCs (Table 1). This reflects the large number of LDVs and LDTs as well as the dominant use of spark-ignition engines to power light-duty vehicles in Canada. The smaller segment of the HDV fleet (HDV2B-3) and the larger class 8 heavy-duty vehicles (HDV8) dominate NO_X, $PM_{2.5}$ and PM_{10} emissions. The dominant use of compression-ignition engines in HDVs is partly responsible for the higher NO_X and particulate emissions compared to LDVs. Figure 1 shows emissions by pollutant for LDVs and HDVs at the national level. It also demonstrates the difference in magnitude of emissions among pollutants. Table 2 also presents the relative contribution (in percentage) from on-road transport to the 2015 Canadian emissions inventory.

For the light-duty segments specifically, LDT emissions exceed LDV emissions, with the exception of NH_3 (Table 1). Canadian vehicle registration data show that vehicles less than 4,500 kg (LDVs and LDTs) make up most of the on-road fleet, with 22 million out of 24 million registrations in 2015.[13] Canadian vehicle sales data show that more units of LDTs (trucks, light trucks and vans, minivans, sport utility vehicles, etc.) have been sold than LDVs (passenger cars) since at least 2010.[14] The heavier LDTs generally have higher fuel consumption rates than LDVs, and are prone to release more friction-based particulate matter emissions associated with brake and tire wear (Wahid 2018).

FIGURE 1: On-road heavy-duty (HDV) and light-duty (LDV) vehicle emissions in Canada, based on the 2015 inventory



[13] Statistics Canada, annual. Vehicle registrations, by type of vehicle. Table 23-10-0067-01. www150.statcan.gc.ca/t1/tbl1/en/tv.action?pid=2310006701

[14] Statistics Canada, annual. New motor vehicle sales, by type of vehicle. Table 20-10-0002-01. www150.statcan.gc.ca/t1/tbl1/en/tv.action?pid=2010000201

Particulate emissions consist of three components: exhaust, brake and tire emissions. Exhaust emissions refer to combustion-related emissions that are released from the vehicle tailpipe. Brake and tire emissions depend on friction mechanisms, through contact between vehicle parts and the road surface. They are also referred to as tire wear and brake lining (TWBL) emissions. Similarly, VOC emissions consist of exhaust emissions and evaporative emissions. The latter reflect the VOCs in fuel that evaporate through the fuel and engine system during operation of the vehicle or when it is parked or stored. TWBL and evaporative emissions make up non-combustion or non-exhaust emissions. Additional details on non-exhaust particulate and VOC emissions from on-road transportation are provided in Appendix B.

TABLE 1: Air pollutant emissions in Canada, in tonnes, for the year 2015[15]–Source categories and classes, and transportation sub-classes

Source category and class	CO	NH_3	NO_X	PM_{10}	$PM_{2.5}$	SO_2	VOC
Incineration	4,001	138	1,687	42	31	2,321	823
Industrial	1,186,309	15,077	611,071	151,068	61,743	759,833	639,746
Miscellaneous	9,737	712	76	18,183	16,746	88	372,284
Non-Industrial	1,261,307	2,744	231,665	175,017	171,213	257,673	232,765
Open Sources[a]	147,491	459,299	11,320	6,901,143	1,335,783	10,709	114,510
Agriculture	897	454,936	4,043	1,333,937	318,299	8,964	97,717
Construction operations	301	31	1,686	2,524,152	504,911	351	20
Dust–paved roads	0	0	0	576,114	139,186	0	0
Dust–unpaved roads	0	0	0	2,445,421	359,730	0	0
Transportation	2,967,720	7,133	958,868	51,376	39,747	12,903	307,956
Air	33,560	4	5,776	356	277	518	2,668
Marine	22,353	296	237,480	5,005	4,604	10,233	10,007
Off-Road	1,322,972	295	201,354	19,038	18,467	240	163,260
On-Road[b]	**1,570,637**	**6,482**	**388,824**	**23,966**	**13,447**	**1,460**	**125,767**
HDV2B-3	343,018	512	118,786	4,116	3,303	223	14,666
HDV4-5	44,255	65	18,748	688	525	24	2,173
HDV6-7	38,721	97	22,009	1,499	1,081	19	3,861
HDV8	40,191	413	125,283	8,019	5,461	80	8,543
LDT	618,266	2,606	60,981	5,452	1,651	617	49,269
LDV	474,416	2,753	42,425	4,166	1,437	493	44,470
MC	11,770	34	594	25	19	4	1,735
Rail	18,198	53	125,434	3,012	2,921	452	6,253
Total–all sources	**5,576,565**	**485,102**	**1,814,687**	**7,297,369**	**1,625,262**	**1,043,527**	**1,854,305**

HDV: Heavy-duty vehicle class; LDT: light-duty truck; LDV: light-duty vehicle; MC: motorcycle

[a] Not all classes of the open sources category are shown; it also includes dust from coal mining, mine tailings, prescribed burning, and waste;

[b] Gross vehicles weight rating for HDVs: 3,856 kg ≤ class 2B-3 < 6,350 kg; 6,350 kg ≤ class 4–5 < 8,845 kg; 8,845 kg ≤ class 6–7 < 14,969 kg; 14,969 kg ≤ class 8

[15] 2015 emissions inventory developed by ECCC; internal reference BASE 2015 ref17 (November 2019).

FIGURE 2: Heavy-duty vehicle emissions in the 2015 Canadian emissions inventory
Left panel: NH_3, PM_{10}, $PM_{2.5}$ and SO_2. Right panel: CO, NO_X and VOC

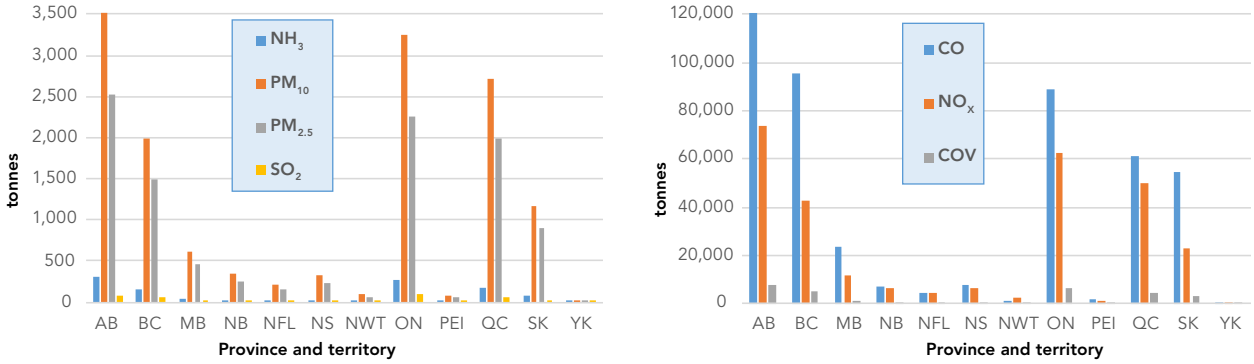
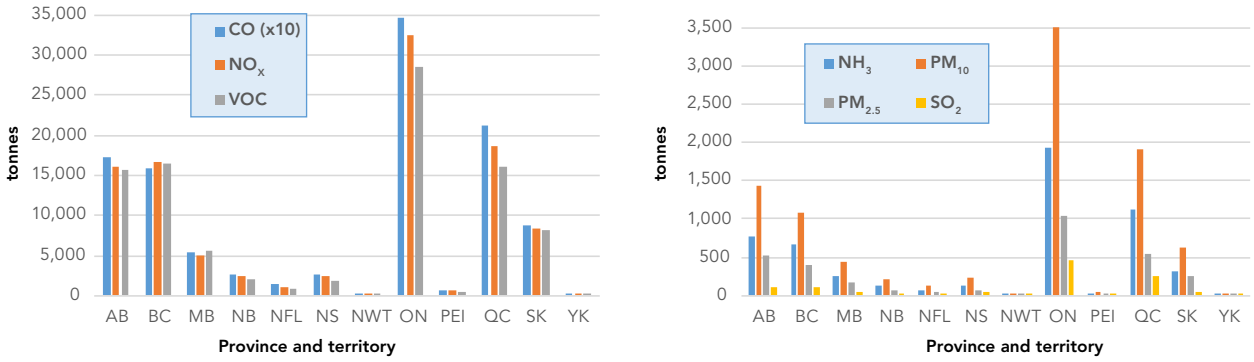


FIGURE 3: Light-duty vehicle emissions in the 2015 Canadian emissions inventory
Left panel: NH_3, PM_{10}, $PM_{2.5}$ and SO_2. Right panel: CO, NO_X and VOC. CO emissions were divided by ten to scale with values for NO_X and VOC



Figures 2 to 4 present on-road vehicle emissions by province and territory for the heavy-duty and light-duty segments of the fleet, while national estimates are shown in Figure B1 (Appendix B). The data are also presented in Table B2 in Appendix B. Although there are slight variations among regions, the inventory shows that heavy-duty vehicles emit more NO_X and PM, whereas light-duty vehicles emit larger quantities of NH_3, SO_2 and VOCs. In terms of regional results, emissions are higher in Ontario and Alberta, followed by Quebec and British Columbia. The magnitude of emissions is generally consistent with population size, except for Alberta. HDV emissions in Alberta are comparable to those in Ontario, although the population of Ontario is approximately three times that of Alberta. Vehicle registration data for 2015 indicate a higher number of HDVs registered in Alberta (112,109) than in other provinces, except Ontario (122,462).[16]

[16] Vehicles weighing 15,000 kilograms or more. Statistics Canada, annual. Vehicle registrations, by type of vehicle. Table 23-10-0067-01. www150.statcan.gc.ca/t1/tbl1/en/tv.action?pid=2310006701

Table 2 presents the relative contribution (in percentage) from on-road transport to the 2015 Canadian emissions inventory. Overall, 28% of CO emissions and 21% of NO_X emissions in Canada are from on-road transport. A modest contribution of 7% to VOC is also estimated, while less than 2% of emissions for PM_{10}, $PM_{2.5}$, NH_3 and SO_2 are from on-road transport. Variations across provinces and territories are influenced by dominant (or absent) economic sectors in each region. For example, the upstream petroleum industry releases a greater proportion of NO_X emissions in Alberta, lowering contributions from TRAP to 13%. The marine, off-road transportation, ores and mineral industry and upstream petroleum industry sectors also contribute more to NO_X emission than on-road transport in Newfoundland, where the transport sector contributes only 8% of the NO_X emissions. In Nova Scotia, the lower relative contribution to NO_X emissions from on-road transport is due to higher emissions from the coal-fired electric power generation sector. The relatively higher contributions to NH_3 emissions from on-road transport observed in the Northwest Territories and Yukon mainly reflect the absence of intensive agricultural activities in those regions, which are responsible for most of the NH_3 emissions elsewhere in Canada. Consequently, the interpretation of relative contributions to regional air pollutant emissions is context-specific and requires knowledge of key activity sectors in each province or territory.

TABLE 2: Percentage of provincial and territorial emissions from on-road transport in 2015

Province and territory	CO	NH_3	NO_X	PM_{10}	$PM_{2.5}$	SO_2	VOC
Newfoundland & Labrador	9.7	10	8.5	0.5	1.0	0.1	2.8
Nova Scotia	25	4.4	12	0.6	1.0	0.1	6.6
Prince Edward Island	26	1.3	40	0.7	1.3	3.5	8.2
New Brunswick	27	4.2	27	0.6	1.2	0.2	8.3
Quebec	18	1.9	32	0.6	1.2	0.3	7.2
Ontario	31	2.6	34	0.6	1.2	0.2	9.6
Manitoba	40	0.5	36	0.3	0.8	0	10
Saskatchewan	46	0.4	22	0.1	0.4	0.1	4.9
Alberta	28	0.8	13	0.2	0.5	0.1	3.8
British Columbia	40	4.5	23	1.0	2.3	0.3	13
Nunavut[a]	0	0	0	0	0	0	0
Northwest Territories	23	47	14	0.9	2.9	0.2	8.7
Yukon	37	52	47	0.5	1.7	0.9	7.9
Canada	**28**	**1.3**	**21**	**0.3**	**0.8**	**0.1**	**6.8**

Estimates are rounded, limited to one decimal, and given to a maximum of two significant figures.

[a] No values available for Nunavut. Estimated as nil.

Maps were generated to visualize the distribution of air pollutant emissions across Canada (Figures B2–B7, Appendix B). In general, air pollutant emission levels generally coincide with the population distribution in Canada, with higher values in urban centres and surrounding areas. In addition, regions with extensive industrial activity, including oil and gas production and mining activities as well as agricultural activities, are associated with higher air pollutant emissions. The maps for $PM_{2.5}$, NO_X and VOC emissions from all source sectors and from on-road transportation explicitly are shown and discussed in more detail in Appendix B.

3.2. MODELLED CONTRIBUTIONS FROM TRAP TO AMBIENT AIR POLLUTION

Average ambient concentrations estimated using the GEM-MACH air quality model are presented in this section for each pollutant, by province and territory. Results for individual CDs are also listed. The terms *reference*, *net* and *relative* concentrations are used for interpreting the results. Reference concentrations are the modelled results obtained from the reference simulation that included the entire emissions inventory. It is important to note that these reference concentrations are direct output from the GEM-MACH model simulations. They have not been adjusted to ambient air monitoring data. The reference concentrations should be interpreted within the context of this analysis. A performance analysis was performed for the specific combinations of the APEI, spatial surrogates and the GEM-MACH version used in the current analysis, and is presented in section 4.3.2.5 and in Appendix D. Performance of the GEM-MACH model was also reported previously in the peer-reviewed literature (Makar et al. 2014a, 2014b; Whaley et al. 2018). Air pollution maps for the reference scenario are shown in Appendix C and are the estimated ambient concentrations for 2015. The *net* and *relative* contributions are the modelled ambient air pollution concentration increments associated with Canadian on-road vehicle emissions, in absolute mass (μg/m^3 or ppbv) and relative terms (%), respectively.

3.2.1. Fine particulate matter

Table 3 shows the national and provincial $PM_{2.5}$ reference concentrations (modelled) as well as contributions from TRAP alone (in descending order of absolute net contribution). The Canadian average population-weighted reference $PM_{2.5}$ concentration is estimated to be 5.3 μg/m^3. Provincially, higher reference levels are estimated in Quebec (7.8 μg/m^3), Ontario (6.4 μg/m^3) and British Columbia (4.2 μg/m^3). Figure C1 in Appendix C shows that grid cells associated with higher area-weighted $PM_{2.5}$ concentrations (i.e., 8 μg/m^3 or more) correspond with urban centres including Vancouver, Edmonton, Toronto and Montreal. As noted previously for the geographical distribution of emissions, higher concentrations are modelled in densely populated areas and those with intensive industrial activity. Concentrations of 2 μg/m^3 or less are projected in remote and rural areas of Canada.

Population-weighted annual average estimates indicate that nationally, 0.4 μg/m^3 or 7.0% of ambient $PM_{2.5}$ concentration is attributable to Canadian on-road vehicle emissions. The relative contribution from TRAP to ambient $PM_{2.5}$ levels is highest in Manitoba (10%), followed by British Columbia (9.6%), Alberta (8.0%) and Ontario (7.1%).

TABLE 3: Contributions from Canadian on-road vehicle emissions to ambient $PM_{2.5}$ concentrations in 2015–Provincial, territorial, and national estimates–Population-weighted annual average

Region	Population[a]	Reference concentration–µg/m³	Contribution from on-road	
			Net–µg/m³	Relative–%
Ontario	13,792,052	6.4	0.5	7.1
Quebec	8,263,600	7.8	0.5	5.9
British Columbia	4,683,139	4.2	0.4	9.6
Manitoba	1,293,378	2.6	0.3	10
Alberta	4,196,457	2.7	0.2	8.0
Saskatchewan	1,133,637	1.4	0.1	6.2
Prince Edward Island	146,447	1.3	< 0.1	3.6
New Brunswick	753,871	1.2	< 0.1	2.7
Nova Scotia	943,002	1.1	< 0.1	2.3
Newfoundland and Labrador	527,756	0.5	< 0.1	1.2
Northwest Territories	44,088	0.2	< 0.1	2.0
Nunavut	36,919	0.1	0	0
Yukon	37,428	0.2	0	0
Canada	**35,851,774**	**5.3**	**0.4**	**7.0**

Net values rounded to one decimal. Relative values limited to two significant figures. Values may not correspond due to rounding. For example, the reference concentration for Ontario is 6.38 µg/m³ and the net contribution from on-road is 0.45 µg/m³ when using two decimals, giving a relative contribution of 7.1%. By contrast, values rounded to one decimal, as shown in the table, equate to 7.8%.

[a] 2015 population estimates. Source: Statistics Canada.

Area-weighted concentrations at the individual CD level (Table 4) show that the highest net contributions to $PM_{2.5}$ from Canadian on-road vehicle emissions (0.9 µg/m³) are modelled in Toronto and Laval, followed by Montreal (0.8 µg/m³), while the highest relative contributions are modelled in Greater Vancouver (9.9%), Toronto (8.4%) and York (8.4%). A combination of primary $PM_{2.5}$ emissions and secondary aerosol formation likely contribute to higher contributions in these CDs. The higher relative value in Vancouver reflects the high population density in the Vancouver area, where 50% of the British Columbia population resides, which involves a high level of personal and commercial traffic. It is also dependent on the low reference concentrations estimated for the Greater Vancouver CD. Additional factors that can influence the contribution from TRAP to ambient levels in all CDs include geographical features (e.g., valleys, mountain ranges, open land) and meteorological conditions (e.g., thermal inversions) that may favour elevated air pollution in a region. Except for the Les Moulins CD in Quebec, most CDs listed in Table 4 correspond to relatively dense urban areas that include high-volume roadways. The Les Moulins CD is located north-northeast and generally downwind of Greater Montreal, and is crossed by two primary highways and several high-volume roadways; this may partly explain the higher estimated contributions in this CD. The results from this analysis are available for all CDs and can be obtained upon request from Health Canada.

TABLE 4: Contributions from Canadian on-road vehicle emissions to ambient $PM_{2.5}$ concentrations in 2015–CDs with the highest net contributions–Area-weighted annual average

Region	Population[a]	Reference concentration–µg/m³	Contribution from on-road	
			Net–µg/m³	Relative–%
ON–Toronto (CD3520)	2,826,498	10.7	0.9	8.4
QC–Laval (CD2465)	425,225	11.9	0.9	7.2
QC–Montréal (CD2466)	1,999,795	11.9	0.8	6.5
BC–Greater Vancouver (CD5915)	2,504,363	6.5	0.6	9.9
QC–Longueuil (CD2458)	421,342	10.8	0.6	5.8
ON–Peel (CD3521)	1,438,770	7.2	0.6	7.8
QC–Les Moulins (CD2464)	159,098	11.0	0.5	4.9
QC–Québec (CD2423)	580,639	7.9	0.5	6.7
ON–York (CD3519)	1,140,024	6.3	0.5	8.4
ON–Halton (CD3524)	559,213	7.2	0.5	7.3
Canada	**35,851,774**	**5.3**	**0.4**	**7.0**

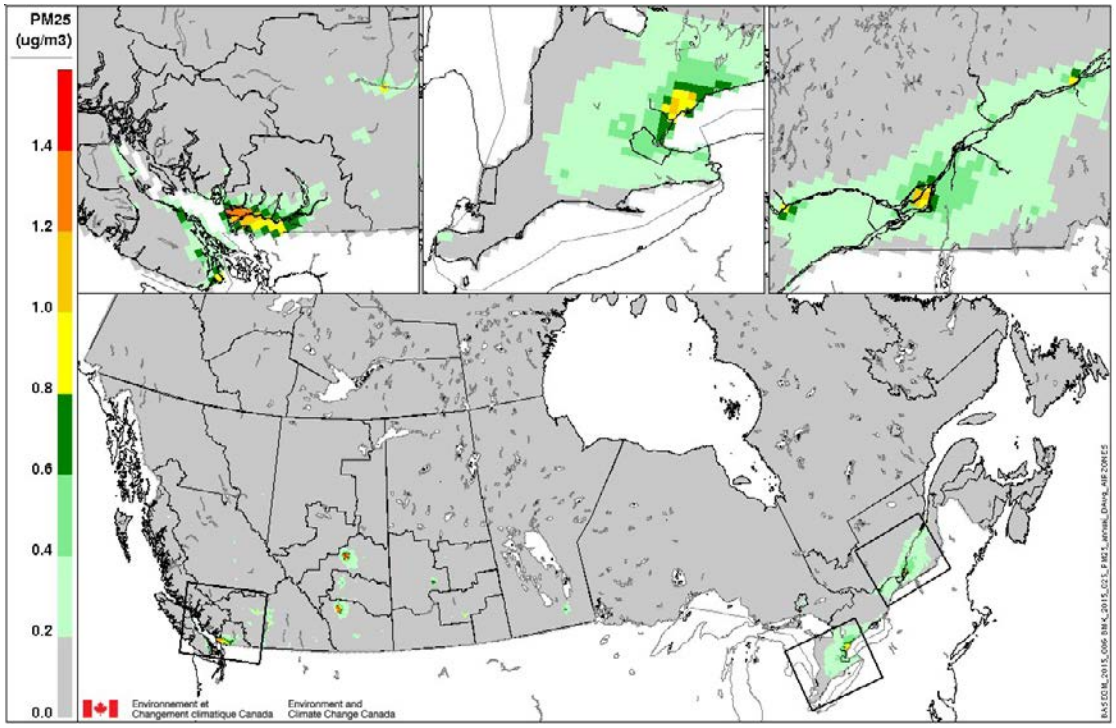
BC: British Columbia; ON: Ontario; QC: Quebec

Net values rounded to one decimal. Relative values limited to two significant figures. Values may not correspond due to rounding.

[a] 2015 population estimates. Source: Statistics Canada.

Figure 4 shows the net contributions from Canadian on-road vehicle emissions to ambient $PM_{2.5}$ concentrations across Canada in 2015. Relative contributions are shown in Figure C2, Appendix C. The maps use the gridded results from the GEM-MACH model and have a resolution of 10 km by 10 km. They do not include any population or area weighting and consequently may differ slightly from the CD-based results presented above. The scale in Figure 4 varies from 0 to approximately 2 µg/m³. Higher contributions are observed in the southwestern part of British Columbia, around Edmonton and Calgary, and along the Windsor–Québec Corridor, which correspond to areas of higher population. In and around the most populated urban centres (Vancouver, Calgary, Edmonton, Regina, Winnipeg, Toronto, and Montreal), contributions from Canadian on-road vehicle emissions are between 0.4 and 1.4 µg/m³. Higher values are estimated in Calgary and Edmonton. The individual maps for LDVs and HDVs (not shown) suggest that within urban centres, both vehicle classes contribute fairly equally to ambient $PM_{2.5}$ concentrations, combining for greater impacts in these grid cells. However, the contribution from HDVs is visible beyond the urban core, covering metropolitan and adjoining areas more broadly than contributions from LDVs.

FIGURE 4: Net contribution (µg/m^3) from Canadian on-road vehicle emissions to annual average $PM_{2.5}$ concentrations in 2015



Notes: Insets for southern British Columbia, Ontario and Quebec

3.2.2. Nitrogen dioxide

Table 5 shows the national and provincial NO_2 reference concentrations (modelled) as well as contributions from TRAP alone (in descending order of absolute net contribution). The modelled Canadian average reference population-weighted NO_2 concentration is 5.1 ppbv.[17] Higher reference levels than the national average are estimated in Ontario (6.2 ppbv), Quebec (5.6 ppbv), and Alberta (5.1 ppbv). Figure C3 in Appendix C shows that grid cells with the highest NO_2 concentrations (i.e., 10 to 20 ppbv) include the major urban centres of Vancouver, Edmonton, Calgary, Toronto and Montreal. Higher concentrations (approximately 2 to 10 ppbv) are generally associated with populated areas and those with intensive oil and gas industrial activity. Concentrations of 2 ppbv or less are estimated for remote and rural areas of Canada.

Table 5 also indicates that, across Canada, 2.0 ppbv or 38% of the annual population-weighted NO_2 concentration is attributable to Canadian on-road vehicle emissions. The highest net contribution is reported in Quebec (2.5 ppbv), followed by British Columbia (2.2 ppbv) and Ontario (2.2 ppbv). The relative contribution from Canadian on-road vehicle emissions to ambient NO_2 exceeds the national relative contribution of 38% in three provinces: Manitoba (51%), British Columbia (47%) and Quebec (45%). This demonstrates the preponderance of on-road vehicle emissions as a source of NO_2 in those regions. Lower relative contributions can reflect the presence of important NO_X sources other

[17] The emissions inventory includes data for NO_X, which represents the sum of nitric oxide (NO) and NO_2 emissions. In general, there are environmental and health concerns associated with exposure to NO_2 in ambient air, but not for exposure to NO. NO emissions are included in the modelling as it is a precursor to NO_2 in ambient air. Air quality modelling results are only available for NO_2.

than on-road vehicles in a province, including upstream petroleum industry activities, electric power generation and off-road vehicles (e.g., mining and oil sands in the case of Alberta). Further, relative contributions are dependent on provincial reference concentrations.

TABLE 5: Contributions from Canadian on-road vehicle emissions to ambient NO_2 concentrations in 2015–Provincial, territorial, and national estimates–Population-weighted annual average

Region	Population[a]	Reference concentration–ppbv	Contribution from on-road	
			Net–ppbv	Relative–%
Quebec	8,263,600	5.6	2.5	45
British Columbia	4,683,139	4.8	2.2	47
Ontario	13,792,052	6.2	2.2	35
Manitoba	1,293,378	3.7	1.9	51
Alberta	4,196,457	5.1	1.3	25
Saskatchewan	1,133,637	1.2	0.3	24
Prince Edward Island	146,447	0.5	0.2	36
Nova Scotia	943,002	0.5	0.2	30
New Brunswick	753,871	0.5	0.1	27
Newfoundland and Labrador	527,756	0.3	0.1	23
Northwest Territories	44,088	0.1	< 0.1	7.3
Yukon	37,428	< 0.1	< 0.1	25
Nunavut	36,919	< 0.1	0	0
Canada	**35,851,774**	**5.1**	**2.0**	**38**

Net values rounded to one decimal. Relative values limited to two significant figures. Values may not correspond due to rounding.

[a] 2015 population estimates. Source: Statistics Canada.

TABLE 6: Contributions from Canadian on-road vehicle emissions to ambient NO_2 concentrations in 2015–CDs with the highest net contributions–Area-weighted annual average

Region	Population[a]	Reference concentration–ppbv	Contribution from on-road	
			Net–ppbv	Relative–%
QC–Laval (CD2465)	425,225	10.5	5.6	53
ON–Toronto (CD3520)	2,826,498	14.0	4.9	35
QC–Montréal (CD2466)	1,999,795	11.1	4.7	42
QC–Longueuil (CD2458)	421,342	7.6	3.9	51
BC–Greater Vancouver (CD5915)	2,504,363	7.8	3.7	47
QC–Québec (CD2423)	580,639	6.9	3.4	50
MB–Division No. 11 (CD4611)	721,819	5.9	3.2	55
QC–Les Moulins (CD2464)	159,098	6.1	3.0	50
ON–Peel (CD3521)	1,438,770	7.5	2.9	39
ON–Halton (CD3524)	559,213	6.7	2.9	43
Canada	**35,851,774**	**5.1**	**2.0**	**38**

BC: British Columbia; MB: Manitoba; ON: Ontario; QC: Quebec

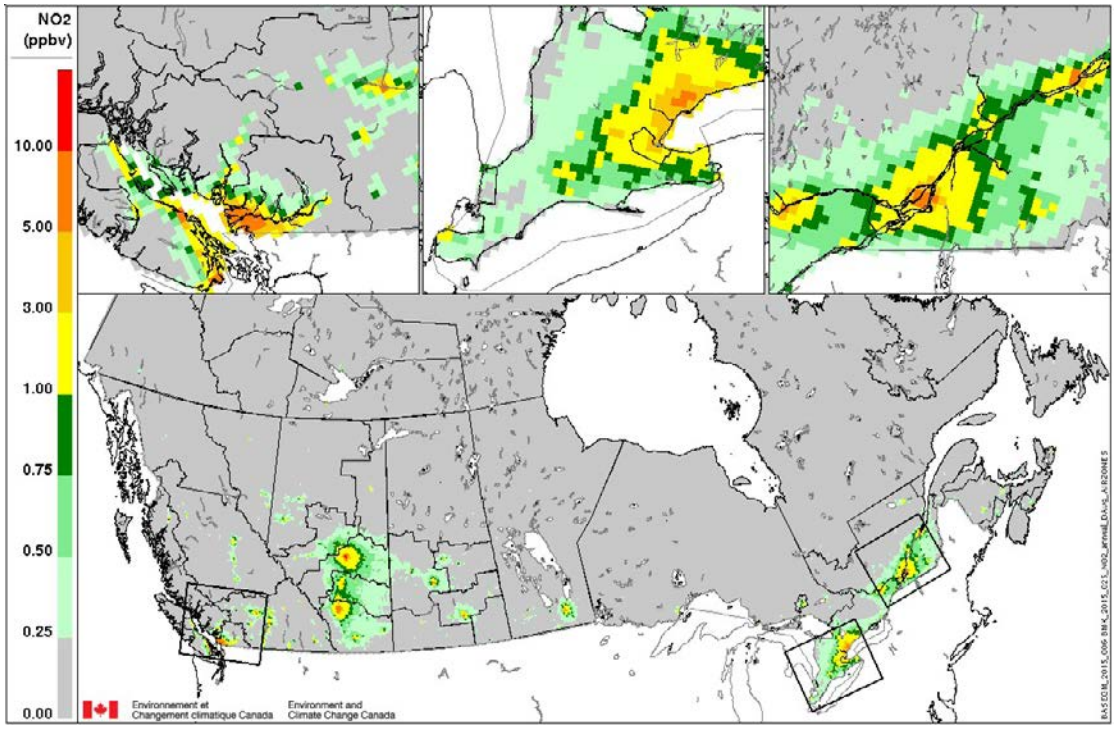
Net values rounded to one decimal. Relative values limited to two significant figures. Values may not correspond due to rounding.

[a] 2015 population estimates. Source: Statistics Canada.

The CD results (Table 6) indicate that the highest net contribution to ambient NO_2 from Canadian on-road vehicle emissions is observed in Laval (5.6 ppbv) and the highest relative contribution is observed in Division No. 11,[18] Manitoba (55%). High relative values are associated with urban areas as well as with suburban regions, indicating an influence of local emissions and regional sources through dispersion of traffic NO_X emissions released in larger urban centres, such as Montreal, Toronto and Vancouver.

Figure 5 shows the contribution from Canadian on-road vehicle emissions to ambient NO_2 concentrations at the grid cell level. The scale in Figure 5 goes up to 10 ppbv. In addition to the grid cells corresponding to CDs listed in Table 6, relatively high contributions to NO_2 concentrations are modelled in Edmonton and Calgary. The general distribution of TRAP contributions to ambient NO_2 concentrations is comparable to that of $PM_{2.5}$. Relative contributions to ambient NO_2 concentrations are shown in Figure C4 (Appendix C). Relative contributions modelled in remote regions (up to 15%), where local NO_X on-road vehicle emissions are minimal, can be associated with very low ambient concentrations (Figure C3), the absence of significant local sources, and atmospheric transport. The modelled absolute contribution from on-road vehicles is generally less than 0.25 ppbv in those regions.

FIGURE 5: Net contribution (ppbv) from Canadian on-road vehicle emissions to annual average NO_2 concentrations in 2015



Notes: Insets for southern British Columbia, Ontario and Quebec

[18] The Division No. 11 census division includes the city of Winnipeg.

3.2.3. Ozone–summer

O_3 concentrations were estimated for the summer period (May 1 to September 30), when O_3 production is generally higher owing to warmer temperatures, increased biogenic VOC emissions and longer daylight hours. Table 7 shows the national and provincial summer O_3 reference concentrations (modelled) as well as contributions from TRAP alone (in descending order of absolute net contribution). The modelled Canadian average reference summer O_3 concentration was 40.0 ppbv. This represents the average daily maximum of hourly data for O_3 for the summer period. Higher reference levels were modelled in Ontario (43.8 ppbv), Alberta (41.4 ppbv) and British Columbia (39.5 ppbv). Figure C5 in Appendix C generally shows higher summer O_3 concentrations (37 ppbv or more) in the more densely populated and developed regions of Canada, compared to concentrations of 30 ppbv or less in remote and rural areas. Particularly high summer O_3 concentrations (greater than 40 ppbv) were estimated between Sarnia and Hamilton and along the northern shore of Lake Erie, as well as in areas directly downwind of major urban centres, including Vancouver, Calgary and Montreal. Areas with sources of elevated NO_X emissions, such as high-traffic roadways and urban centres, often have lower O_3 levels compared to areas further downwind (e.g., suburban areas) owing to the scavenging (titration) effect of NO. This is linked to the photochemical reactions involving NO_X, O_3 and VOCs in ambient air that can increase or decrease O_3 concentrations. Environmental (e.g., large water bodies, valleys) and meteorological conditions can also influence ambient O_3 levels.

Modelled contributions from Canadian on-road vehicle emissions to ambient summer O_3 concentrations at the national and provincial levels were limited to less than 4%. On average across Canada, on-road vehicle emissions were responsible for 0.9 ppbv or 2.3% of ambient summer O_3. The highest net and relative provincial contributions from Canadian on-road vehicle emissions to ambient summer O_3 were reported in Alberta (1.6 ppbv, 3.7%), Quebec (1.4 ppbv, 3.8%) and Saskatchewan (1.0 ppbv, 2.8%).

TABLE 7: Contribution from Canadian on-road vehicle emissions to ambient summer O_3 concentrations in 2015–Provincial, territorial, and national estimates–Population-weighted summer average of daily maximum values

Region	Population[a]	Reference concentration–ppbv	Contribution from on-road	
			Net–ppbv	Relative–%
Alberta	4,196,457	41.4	1.6	3.7
Quebec	8,263,600	36.5	1.4	3.8
Saskatchewan	1,133,637	36.2	1.0	2.8
Prince Edward Island	146,447	34.2	0.8	2.4
New Brunswick	753,871	33.5	0.8	2.3
Manitoba	1,293,378	32.7	0.8	2.3
Ontario	13,792,052	43.8	0.7	1.5
Nova Scotia	943,002	36.2	0.7	1.8
British Columbia	4,683,139	39.5	0.6	1.5
Newfoundland and Labrador	527,756	34.8	0.3	0.8
Northwest Territories	44,088	29.2	0.2	0.5
Yukon	37,428	28.5	0.1	0.2
Nunavut	36,919	30.5	< 0.1	0.1
Canada	**35,851,774**	**40.0**	**0.9**	**2.3**

Net and relative values rounded to one decimal. Values may not correspond due to rounding.

[a] 2015 population estimates. Source: Statistics Canada.

As shown in Table 8, the highest net and relative contributions from on-road vehicle emissions to summer O_3 were estimated in Central Okanagan (5.2 ppbv; 14%) and Fraser Valley (3.3 ppbv; 8.7%), British Columbia, followed mostly by suburban CDs downwind of Montreal, Quebec. These CDs do not correspond to urban core areas, or those linked to the highest NO_2 contributions (see Table 6), but they still correspond to areas of higher NO_2 levels (see Figure 5). This likely relates to the influence of NO_X emissions transported from urban centres towards areas with relatively high VOC concentrations and limited local sources of NO_X emissions (i.e., NO_X-limited conditions).

The contribution from Canadian on-road vehicle emissions to summer O_3 concentrations was negative in the CDs of Toronto (-1.3 ppbv or -2.8%) and Greater Vancouver (-0.6 ppbv or -1.3%). On-road vehicles do not emit O_3 directly; they emit pollutants that react to increase or decrease ambient O_3 concentrations. Shifting the balance among O_3 precursors, especially NO_X, can lead to changes in ambient O_3 in both directions. Under some conditions, elevated local traffic-related NO_X emissions may titrate O_3 molecules and cause a decrease in ambient O_3 levels in urban areas (i.e., VOC-limited/NO_X-inhibited conditions). The apparent air quality benefit in Toronto and Greater Vancouver is associated with removing on-road emissions (a large source of NO_X emissions) in those urban areas. Higher contributions to summer O_3 from Canadian on-road vehicle emissions were modelled in areas adjacent to these CDs and that may be impacted by urban emissions, such as Fraser Valley, Central Okanagan, and Cowichan Valley near Greater Vancouver (Table 8). Decreases in ambient O_3 concentrations in urban areas associated with elevated NO_X emissions, and increases in suburban or rural areas downwind from urban areas, have been widely observed and discussed in previous analyses (Environment Canada and Health Canada 2011; Jhun et al. 2015).

TABLE 8: Contribution from Canadian on-road vehicle emissions to summer O_3 concentrations in 2015–CDs with the highest net contributions–Area-weighted summer average of daily maximum values

Region	Population[a]	Reference concentration–ppbv	Contribution from on-road	
			Net–ppbv	Relative–%
BC–Central Okanagan (CD5935)	197,287	37.6	5.2	14
BC–Fraser Valley (CD5909)	301,097	38.1	3.3	8.7
QC–Nicolet-Yamaska (CD2450)	22,889	38.8	3.1	7.9
QC–Francheville (CD2437)	153,691	37.5	2.9	7.6
QC–Pierre-de-Saurel (CD2453)	51,088	40.2	2.8	7.1
QC–Drummond (CD2449)	102,797	38.4	2.8	7.4
QC–Les Maskoutains (CD2454)	86,201	39.6	2.8	7.1
QC–Bécancour (CD2438)	20,346	37.0	2.8	7.5
BC–Cowichan Valley (CD5919)	82,605	35.8	2.6	7.1
QC–Acton (CD2448)	15,443	37.6	2.6	6.8
Canada	**35,851,774**	**40.1**	**0.9**	**2.3**

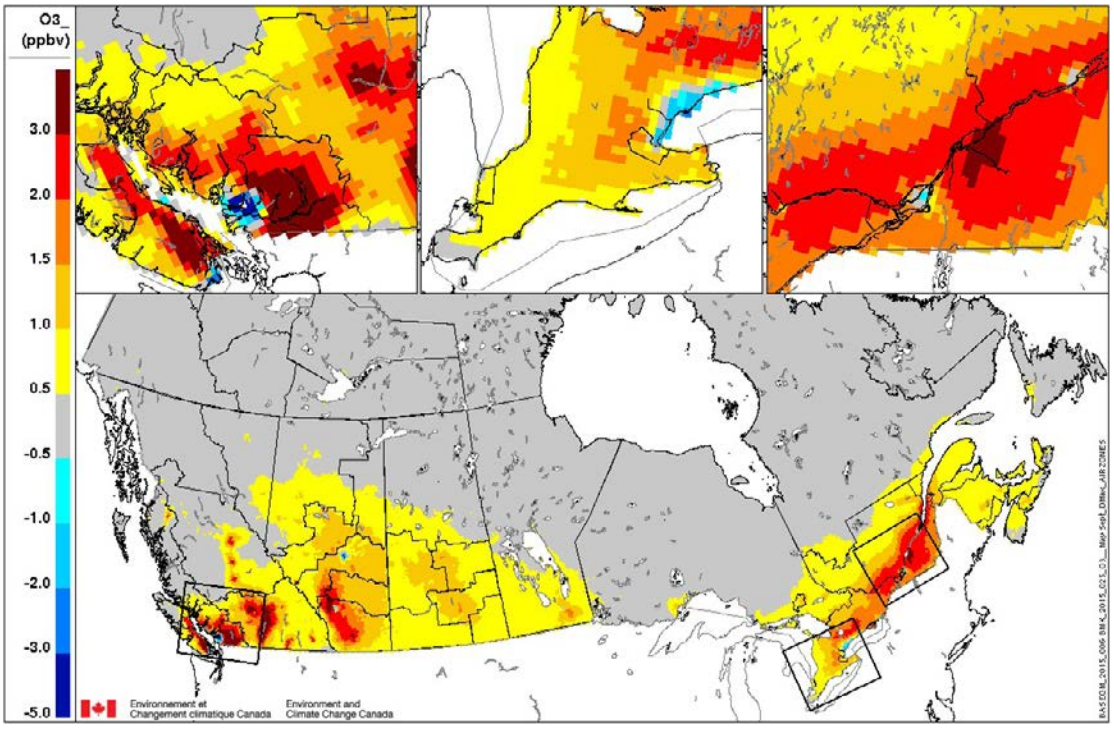
BC: British Columbia; QC: Quebec

Net values rounded to one decimal. Relative values limited to two significant figures. Total may not correspond due to rounding.

[a] 2015 population estimates. Source: Statistics Canada.

Figure 6 shows the contribution from Canadian on-road vehicle emissions to ambient summer O_3 concentrations (average daily 1-hour maximum concentrations). Contributions between 2 and 3 ppbv are modelled in western provinces and along the southern parts of Ontario and Quebec. The insets show decreases in summer O_3 levels in grid cells corresponding to densely populated areas, including Metro Vancouver, the Greater Toronto and Hamilton Area (GTHA) and downtown Montreal. The relative contribution map (Figure C6, Appendix C) provides comparable information about the impacts of TRAP on summer O_3 levels. Relative contributions across individual grid cells vary between decreases of 10% to 20% and increases of 6%.

FIGURE 6: Net contribution (ppbv) from Canadian on-road vehicle emissions to summer average daily maximum O_3 concentrations in 2015



Notes: Insets for southern British Columbia, Ontario and Quebec

3.2.4. Ozone–annual

Table 9 shows the national and provincial annual O_3 reference concentrations (modelled) as well as contributions from TRAP alone (in descending order of absolute net contribution). The modelled Canadian average annual O_3 reference concentration[19] was 36.4 ppbv. This represents the average daily maximum of hourly data for O_3 for the entire year. Provincial reference concentrations were higher in Ontario (38.6 ppbv), Alberta (37.3 ppbv), Nova Scotia (36.5 ppbv) and Newfoundland and Labrador (36.4 ppbv). Figure C7 in Appendix C indicates that the highest annual O_3 concentrations (40 ppbv or more) at the grid cell level were modelled in areas around urban centres and along the region between Sarnia and Hamilton, as well as on the northern shore of Lake Erie. In remote and rural areas of Canada, annual O_3 concentrations were 30 ppbv or less. In general, the geographic distributions of annual and summer O_3 concentrations are comparable (see figures C5 and C7). However, annual O_3 concentrations are generally lower than summer O_3, such that high annual O_3 concentration grid cells are fewer and less geographically dispersed.

National population-weighted annual average estimates indicate that Canadian on-road vehicle emissions decrease annual O_3 concentrations by 0.1 ppbv or 0.3%. As noted previously, sources of elevated NO_X emissions such as on-road vehicle traffic can lower O_3 levels owing to the scavenging effect of NO. The decrease O_3 concentrations reflects complex photochemical reactions involving NO_X, O_3 and VOCs in ambient air and the influence of meteorological conditions. The modelled small decrease in annual O_3 concentrations does not indicate that on-road vehicle emissions are beneficial

[19] Annual average daily maximum 1-h O_3 concentration

to air quality. Traffic emissions were also modelled to decrease annual O_3 concentrations by 0.3 ppbv in British Columbia, 0.2 ppbv in Ontario, 0.2 ppbv in Manitoba and 0.1 ppbv in Quebec (less than 1% of the reference concentration in each province). By contrast, the highest net and relative contributions from on-road vehicle emissions to annual O_3 are modelled in New Brunswick (0.4 ppbv, 1.1%), Prince Edward Island (0.4 ppbv, 1.1%) and Saskatchewan (0.4 ppbv, 1.1%).

TABLE 9: Contribution from Canadian on-road vehicle emissions to ambient annual O_3 concentrations in 2015–Provincial, territorial, and national estimates–Population-weighted annual average of daily maximum values

Region	Population[a]	Reference concentration–ppbv	Contribution from on-road	
			Net–ppbv	Relative–%
Prince Edward Island	146,447	35.4	0.4	1.1
Saskatchewan	1,133,637	34.5	0.4	1.1
New Brunswick	753,871	34.4	0.4	1.1
Nova Scotia	943,002	36.5	0.3	0.9
Alberta	4,196,457	37.3	0.3	0.8
Newfoundland and Labrador	527,756	36.4	0.1	0.4
Northwest Territories	44,088	30.3	0.1	0.2
Yukon	37,428	32.0	< 0.1	0.1
Nunavut	36,919	31.6	< 0.1	0.1
Quebec	8,263,600	33.9	-0.1	-0.4
Manitoba	1,293,378	31.3	-0.2	-0.6
Ontario	13,792,052	38.6	-0.2	-0.6
British Columbia	4,683,139	35.4	-0.3	-0.9
Canada	**35,851,774**	**36.4**	**-0.1**	**-0.3**

Net and relative values rounded to one decimal. Total may not correspond due to rounding.

[a] 2015 population estimates. Source: Statistics Canada.

Table 10 shows that contributions to annual O_3 are limited to 2.2 ppbv or less at the CD level, with the maximum value being in Central Okanagan, British Columbia. As outlined above for summer O_3, higher contributions to annual O_3 correspond to CDs that are likely influenced by on-road transport emissions released upwind, generally in major urban centres. Most CDs in Table 10 can be described as suburban or even rural regions, and they are located downwind of Vancouver, Montreal and the National Capital Region. Table 11 shows that negative values were modelled in 11 CDs accounting for 39% of the Canadian population. Six of these CDs have populations over 1,000,000 and correspond to urban areas (i.e., high road traffic activity and higher NO_X emissions leading to O_3 scavenging). The reduction in ambient annual O_3 concentration reaches 1.8 ppbv (4.9%) in Toronto. Considerations for interpreting the negative values are discussed in section 4.1.

TABLE 10: Contribution from Canadian on-road vehicle emissions to ambient annual O_3 concentrations in 2015–CDs with the highest net contributions–Area-weighted annual average of daily maximum values

Region	Population[a]	Reference concentration–ppbv	Contribution from on-road	
			Net–ppbv	Relative–%
BC–Central Okanagan (CD5935)	197,287	34.9	2.2	6.3
BC–Fraser Valley (CD5909)	301,097	36.0	1.4	4.0
BC–North Okanagan (CD5937)	84,798	32.3	1.2	3.8
QC–Nicolet-Yamaska (CD2450)	22,889	35.6	1.1	3.0
QC–Bécancour (CD2438)	20,346	34.9	1.0	3.0
QC–Drummond (CD2449)	102,797	36.1	1.0	2.8
QC–Acton (CD2448)	15,443	36.4	1.0	2.7
BC–Cowichan Valley (CD5919)	82,605	35.4	1.0	2.8
QC–Arthabaska (CD2439)	71,354	35.8	1.0	2.7
QC–Papineau (CD2480)	22,828	35.1	1.0	2.8
QC–L'Érable (CD2432)	23,486	35.3	1.0	2.8
Canada	**35,851,774**	**36.4**	**-0.10**	**0.3**

BC: British Columbia; QC: Quebec

Net and relative values rounded to one decimal. Total may not correspond due to rounding.

[a] 2015 population estimates. Source: Statistics Canada.

TABLE 11: Contribution from Canadian on-road vehicle emissions to ambient annual O_3 concentrations in 2015–CDs with negative estimates–Area-weighted annual average of daily maximum values

Region	Population[a]	Reference concentration–ppbv	Contribution from on-road	
			Net–ppbv	Relative–%
ON–Toronto (CD3520)	2,826,498	37.9	-1.8	-4.9
QC–Laval (CD2465)	425,225	33.0	-1.5	-4.4
BC–Greater Vancouver (CD5915)	2,504,363	36.3	-1.3	-3.6
QC–Montréal (CD2466)	1,999,795	33.0	-1.3	-3.9
QC–Longueuil (CD2458)	421,342	34.9	-0.6	-1.8
MB–Division No. 11 (CD4611)	721,819	30.9	-0.6	-1.9
QC–Quebec (CD2423)	580,639	32.3	-0.4	-1.3
ON–Peel (CD3521)	1,438,770	38.9	-0.4	-1.0
ON–Halton (CD3524)	559,213	39.7	-0.3	-0.8
AB–Division No. 11 (CD4811)	1,404,432	36.3	-0.1	-0.2
ON–York (CD3519)	1,140,024	39.0	< 0.1	-0.1
Canada	**35,851,774**	**36.4**	**-0.1**	**0.3**

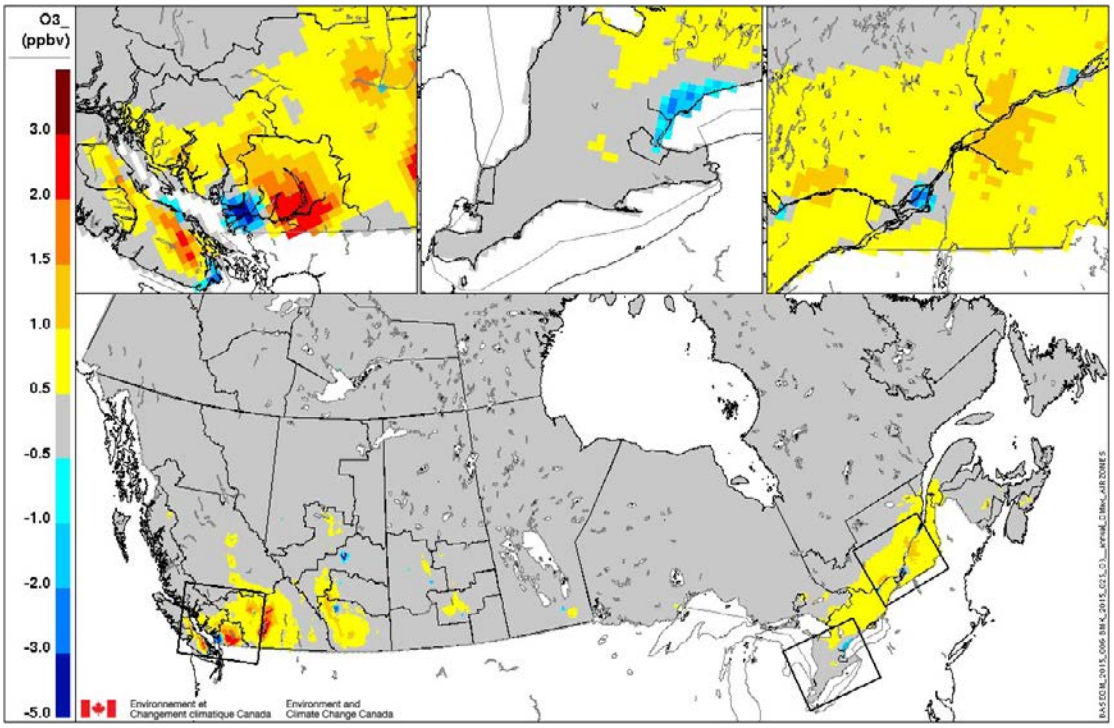
AB: Alberta; BC: British Columbia; MB: Manitoba; ON: Ontario; QC: Quebec

Net and relative values rounded to one decimal. Total may not correspond due to rounding.

[a] 2015 population estimates. Source: Statistics Canada.

Figure 7 shows the modelled contribution from Canadian on-road vehicle emissions to annual O_3 concentrations across Canada in 2015. Grid cell estimates concur with values presented in the previous tables: Decreases are reported in grid cells corresponding to urban centres, while increases are reported in grid cells surrounding less densely populated areas. Transition zones with no change in modelled concentrations form a visible ring around larger urban centres, including Vancouver, the GTHA and Montreal (see insets in Figure 11). Figure C8, Appendix C, displays relative contributions and provides comparable information, reiterating that negative contributions to annual O_3 concentrations affect fewer grid cells but are of greater magnitude (up to 10% to 20%) than positive contributions (up to approximately 6%).

FIGURE 7: Net contribution (ppbv) from Canadian on-road vehicle emissions to annual average daily maximum O_3 concentrations in 2015



Notes: Insets for southern British Columbia, Ontario and Quebec

3.2.5. Sulphur dioxide

Table 12 shows the national and provincial SO_2 reference concentrations (modelled) as well as the population-weighted annual average contributions from TRAP alone (in descending order of absolute net contribution). The modelled Canadian average reference SO_2 concentration was 0.7 ppbv, and the highest provincial reference values were estimated in Ontario (0.9 ppbv), Quebec (0.9 ppbv) and Alberta (0.8 ppbv).

Less than 0.1 ppbv or 1.5% of ambient SO_2 concentration was attributable to TRAP, on average, across Canada in 2015. At the provincial level, net contributions from Canadian on-road vehicle emissions were also lower than 0.1 ppbv. The highest relative contribution, 3.4%, was modelled in British Columbia. The sulphur content of fuels used in on-road vehicles in Canada is very low. Ultra-

low sulphur diesel is used in all on-road diesel vehicles (since 2006) and has a maximum sulphur content of 15 ppm.[20] Gasoline fuels in 2015 met a pool average of 30 ppm.[21] By contrast, the sulphur content in marine vessel diesel fuel had a maximum allowable content of 1,000 ppm in 2015.

At the individual CD level (results not shown), modelled net contributions were less than 0.1 ppbv, while relative contributions of 3.2% were modelled in Toronto and 2.1% in Montreal. The modelled reference concentrations for SO_2 are low in Canada, and the low incremental concentrations attributable to Canadian on-road vehicle emissions suggest that they were not a major contributor to ambient SO_2 pollution in 2015.

TABLE 12: Contributions from Canadian on-road vehicle emissions to ambient SO_2 concentrations in 2015–Provincial, territorial, and national estimates–Population-weighted annual average

Region	Population[a]	Reference concentration–ppbv	Contribution from on-road	
			Net–ppbv	Relative–%
British Columbia	4,683,139	0.3	< 0.1	3.4
Ontario	13,792,052	0.9	< 0.1	1.6
Quebec	8,263,600	0.9	< 0.1	1.4
Manitoba	1,293,378	0.3	< 0.1	2.6
Alberta	4,196,457	0.8	< 0.1	0.4
New Brunswick	753,871	0.2	< 0.1	0.2
Saskatchewan	1,133,637	0.3	< 0.1	0.1
Nova Scotia	943,002	0.2	0	0
Prince Edward Island	146,447	0.2	0	0
Newfoundland and Labrador	527,756	0.1	0	0
Northwest Territories	44,088	< 0.1	0	0
Nunavut	36,919	< 0.1	0	0
Yukon	37,428	0	0	0
Canada	**35,851,774**	**0.7**	**< 0.1**	**1.5**

Net and relative values rounded to one decimal. Total may not correspond due to rounding.

[a] 2015 population estimates. Source: Statistics Canada.

[20] Sulphur in Diesel Fuel Regulations (SOR/2002-254). https://laws-lois.justice.gc.ca/eng/regulations/sor-2002-254/index.html

[21] The sulphur content limit in gasoline fuel has been reduced to 10 ppm since January 1, 2017; Sulphur in Gasoline Regulations (SOR/99-236). https://laws-lois.justice.gc.ca/eng/regulations/SOR-99-236/page-1.html#h-1029238

3.3. MODELLED AIR POLLUTION HEALTH IMPACTS

National count and monetary value estimates for all mortality and morbidity endpoints, modelled using AQBAT, are presented in Table 13. TRAP was associated with 1,200 premature deaths in Canada in 2015, for an estimated economic value of $9 billion (CAD 2015). Each premature death is valued at $7.4 million (CAD 2015), which is considerably more than the valuation for non-fatal outcomes (see Table A2 in Appendix A). Chronic exposure to traffic-related $PM_{2.5}$ contributed 800 or 66% of total premature deaths. Exposure to NO_2 and O_3 contributed to approximately 420 premature deaths, with 330 associated with short-term exposure and 92 associated with long-term exposure.[22] CRFs for non-fatal outcomes included in the current version of AQBAT are based on exposures to $PM_{2.5}$ and summer O_3. Annual counts for some non-fatal outcomes are much higher than for premature deaths, including 2,700,000 acute respiratory symptom days, 1,100,000 restricted activity days and 210,000 asthma symptom days. Estimates for more severe non-fatal outcomes include 610 emergency room visits and 170 hospital admissions per year. The economic costs for non-fatal outcomes total $470 million per year (CAD 2015). Overall, mortality and morbidity impacts total $9.5 billion in 2015.

TABLE 13: National estimates of premature deaths and non-fatal health outcomes associated with exposure to TRAP from Canadian sources in 2015, by health endpoint–Counts and valuation

Health Endpoint	Pollutant	Count	Valuation ($1,000; CAD 2015)
		Median [95% CI]	Median [95% CI]
Premature deaths			
Acute exposure mortality	NO_2, O_3	330 [110; 540]	2,400,000 [620,000; 5,100,000]
Chronic exposure respiratory mortality	Summer O_3[b]	92 [32; 150]	680,000 [180,000; 1,400,000]
Chronic exposure mortality	$PM_{2.5}$	800 [430; 1,200]	5,900,000 [2,200,000; 11,000,000]
Total deaths[a]	**All pollutants**	**1,200 [790; 1,700]**	**9,000,000 [4,500,000; 15,000,000]**
Non-fatal outcomes			
Acute respiratory symptom days	Summer O_3, $PM_{2.5}$	2,700,000 [670,000; 4,900,000]	27,000 [0; 95,000]
Adult chronic bronchitis cases	$PM_{2.5}$	800 [6; 1,600]	340,000 [1,700; 910,000]
Asthma symptom days	Summer O_3, $PM_{2.5}$	210,000 [68,000; 360,000]	15,000 [2,800; 39,000]
Cardiac emergency room visits	$PM_{2.5}$	83 [44; 120]	510 [250; 810]
Cardiac hospital admissions	$PM_{2.5}$	63 [33; 93]	[c]
Child acute bronchitis episodes	$PM_{2.5}$	3,600 [0; 8,000]	1,600 [0; 4,500]

[22] The AQBAT estimates are presented with a maximum of two significant figures. Total may not correspond due to rounding.

Health Endpoint	Pollutant	Count	Valuation ($1,000; CAD 2015)
		Median [95% CI]	Median [95% CI]
Minor restricted activity days	Summer O_3	150,000 [0; 620,000]	4,600 [0; 23,000]
Respiratory emergency room visits	Summer O_3, $PM_{2.5}$	530 [250; 800]	1,500 [690; 2,400]
Respiratory hospital admissions	Summer O_3, $PM_{2.5}$	110 [51; 160]	[c]
Restricted activity days	$PM_{2.5}$	1,100,000 [660,000; 1,600,000]	75,000 [18,000; 150,000]
Total non-fatal outcomes[a]	**All pollutants**	-	**470,000** [23,000; 1,200,000]
All endpoints	**All pollutants**	-	**9,500,000** [5,000,000; 15,000,000]

Values represent median estimates of counts and economic value. Estimates are rounded to the nearest integer and given to a maximum of two significant figures. 95% confidence intervals (CI) represent the 2.5 and 97.5 percentiles of provincial and territorial values.

[a] Total or difference may not correspond due to rounding;

[b] May–September only;

[c] In AQBAT, no economic valuation is associated with hospital admissions. It is assumed that air pollution-related hospital admissions involve an initial emergency room visit of the same type (cardiac or respiratory) that includes a cost.

Table 14 presents modelled estimates of premature deaths associated with exposure to TRAP from Canadian sources, by province and pollutant. Results are not shown for the territories because estimated contributions were nil. Overall, greater health impacts were estimated for the more populated provinces, including Ontario (500 premature deaths) and Quebec (410), followed by British Columbia (170) and Alberta (82). Of the 1,200 total premature deaths estimated across Canada, 800 were associated with exposure to $PM_{2.5}$, 340 with exposure to NO_2 and 85 with exposure to O_3 (92 deaths for summer O_3 and -7 for annual O_3). The acute exposure mortality estimate in Table 13 (330 premature deaths) represents the total for NO_2 (340 premature deaths) and annual O_3 (-7 premature deaths) (see Table 14). Interpretation of negative estimates of premature deaths associated with exposure to O_3 is addressed in section 4.3.

The last two columns of Table 14 show the count for all pollutants combined as well as the rate of premature death per 100,000 population. The rate per 100,000 population provides a population-based normalized value, allowing for appropriate comparisons of health impact estimates among geographic regions of different population size. The results normalized by population show that the risk of premature death associated with TRAP, on average across Canada in 2015, was equivalent to 3 deaths per 100,000 population. The provincial estimates ranged from 0 deaths per 100,000 population in Newfoundland and Labrador to 5 deaths per 100,000 population in Quebec.

TABLE 14: Premature deaths associated with exposure to TRAP in 2015, by pollutant–Provincial and national estimates–Counts and rates per 100,000 population

Region	NO_2[b]	Annual O_3[b]	Summer O_3[c]	$PM_{2.5}$[d]	All[a,b,c,d]	
					Count	per 100,000
Alberta	20	5	13	43	82	2
British Columbia	49	-3	10	110	170	4
Manitoba	14	-2	3	24	38	3
New Brunswick	1	2	2	2	6	1
NFL	0	1	0	0	2	0
Nova Scotia	1	2	2	2	7	1
Ontario	130	-10	25	350	500	4
PEI	0	0	0	1	2	1
Quebec	120	-6	34	260	410	5
Saskatchewan	2	3	3	7	14	1
Canada[a] [95% CI]	**340** [120; 550]	**-7** [-9; –5]	**92** [32; 150]	**800** [430; 1,200]	**1,200** [790; 1,700]	**3**

NFL: Newfoundland and Labrador; PEI: Prince Edward Island

Values represent the median, are rounded to the nearest integer and given to a maximum of two significant figures. 95% confidence intervals (CI) represent the 2.5 and 97.5 percentiles of provincial and territorial values.

[a] Total may not correspond due to rounding;

[b] Acute exposure all-cause mortality;

[c] Chronic exposure respiratory mortality;

[d] Chronic exposure all-cause mortality.

Table 15 presents CDs with the highest modelled premature deaths. As expected, the list of CDs includes the most densely populated and urban areas of Canada, including Metro Vancouver, the GTHA, and Montreal, as well as Québec, Winnipeg, Calgary, Edmonton and Ottawa. These CDs were characterized by higher contributions from Canadian on-road vehicle emissions to ambient concentrations (see section 3.2). The Toronto CD stands out with 170 premature deaths overall, including reductions of 26 premature deaths associated with annual O_3 concentrations and 8 deaths for summer O_3. The Montreal and Greater Vancouver CDs follow with 150 and 110 premature deaths, respectively, with both CDs including a reduction of 16 premature deaths linked to annual O_3. The rates of premature deaths per 100,000 population varied from 2 in Division No. 11 (Edmonton), Alberta, to 9 in Laval, Quebec. By comparison, the national rate was 3 deaths per 100,000 population in 2015.

Of note, the Greater Vancouver CD contributed to 68% of the estimated premature deaths attributable to TRAP from Canadian sources in British Columbia (110 of 170 premature deaths). By contrast, Toronto and Montreal CDs each contributed to 36% of the health impacts in their respective provinces. The geographic distributions of non-fatal outcomes across Canadian CDs were generally comparable to the one for premature deaths.

TABLE 15: Premature deaths associated with exposure to TRAP from Canadian sources in 2015, by pollutant–Census divisions with the highest estimates–Counts and rates per 100,000 population

Census division	NO_2[b]	Annual O_3[b]	Summer O_3[c]	$PM_{2.5}$[d]	All[a]	
					Count	per 100,000
ON–Toronto (CD3520)	62	-26	-8	140	170	6
QC–Montréal (CD2466)	53	-16	3	110	150	7
BC–Greater Vancouver (CD5915)	41	-16	-3	88	110	4
ON–Peel (CD3521)	13	-2	1	31	43	3
QC–Québec (CD2423)	13	-2	2	25	38	7
ON–York (CD3519)	11	0	2	26	38	3
QC–Laval (CD2465)	14	-4	1	26	37	9
AB–Division No. 6 (CD4806)	9	3	6	17	36	2
MB–Division No. 11 (CD4611)	13	-3	1	20	32	4
AB–Division No. 11 (CD4811)	9	-1	3	18	29	2
ON–Ottawa (CD3506)	6	3	4	16	28	3
ON–Hamilton (CD3525)	8	0	2	19	28	5
QC–Longueuil (CD2458)	9	-2	1	19	28	7
ON–Durham (CD3518)	6	0	2	17	25	4
ON–Halton (CD3524)	7	-1	1	15	21	4

AB: Alberta; BC: British Columbia; MB: Manitoba; ON: Ontario; QC: Quebec

Values represent the median, are rounded to the nearest integer and given to a maximum of two significant figures.

[a] Total may not correspond due to rounding;

[b] Acute exposure all-cause mortality;

[c] Chronic exposure respiratory mortality;

[d] Chronic exposure all-cause mortality.

CHAPTER 4: **DISCUSSION**

4.1. EMISSIONS INVENTORY AND MODELLED AIR QUALITY

The 2015 emissions inventory indicated that on-road sources were a notable contributor to national emissions of CO (28%), NO_X (21%) and VOC (7%) (Table 2), but accounted for 1% or less of $PM_{2.5}$ and SO_2 emissions. These estimates correspond to primary emissions and do not consider secondary formation of pollutants, such as secondary aerosols (i.e., secondary $PM_{2.5}$) and O_3. Secondary pollutants are accounted for in the air quality modelling.

Table 16 summarizes the contributions from on-road vehicle emissions to ambient air concentrations for NO_2, O_3 and $PM_{2.5}$. Results are not included for SO_2 owing to low values (see Table 12). Although variations exist among provinces and territories for the different air pollutants, higher on-road vehicle emissions were generally estimated for the more populated provinces of British Columbia, Ontario and Quebec, as well as for Manitoba.

In contrast to the emissions inventory, the ambient air quality modelling results indicated that on-road transportation was an important contributor to population-weighted ambient $PM_{2.5}$ concentrations, reaching 7.0% on average across Canada (Table 3). The discrepancy between contributions to primary $PM_{2.5}$ emissions (0.8%; see Table 2) and ambient $PM_{2.5}$ concentrations (7.0%) is linked to secondary reactions involving precursors such as NO_X, VOCs, and NH_3. Further, the national and provincial average concentration estimates are population-weighted values accounting for population distributions within provinces at the CD level; they represent average exposure concentrations across a given population. The emissions inventory data, by comparison, represent provincial totals only, and they do not account for population weighting and the distribution of emissions across each CD. Given that the largest fraction of on-road vehicle emissions is released near populations, the contributions based on population weighting are expected to be higher than non-weighted national or provincial averages. This also explains why the contribution from Canadian on-road vehicle emissions to population-weighted NO_2 concentrations nationally (38%; see Table 5) is higher than its contribution to NO_X emissions (21%; see Table 2).

Estimates of population exposure to O_3 did not follow the same pattern as other pollutants. The analysis indicated that on-road transport contributed to 2.3% of population-weighted summer O_3 (see Table 7), whereas it contributed to an overall decrease of 0.3% in annual O_3 (see Table 9). Vehicles do not release O_3 directly (no primary emissions of O_3), but release air pollutants that can contribute both to the formation and destruction of O_3 in the atmosphere. The eventual change in O_3 concentration (increase or decrease) depends on emissions, ambient air quality and environmental conditions, and is influenced by non-linear atmospheric photochemical reactions. Further, O_3 metrics of different averaging periods (i.e., summer and annual) respond differently due to the seasonal nature of biogenic VOC emissions and photochemical production of O_3. The two O_3 metrics were selected based on epidemiological evidence indicating associations between acute health effects and exposure to O_3 throughout the year, as well as between chronic health effects and exposure to O_3 during summer months (Judek et al. 2019).

Canadian on-road vehicle emissions contributed to only 1.5% of population-weighted ambient SO_2 levels (see Table 12), which generally concurs with the emission data (0.1% contribution; see Table 2). Low SO_2 emissions were expected owing to the use of low sulphur on-road fuels in Canada.

TABLE 16: Contributions from on-road vehicle emissions to ambient concentrations of NO_2, O_3 and $PM_{2.5}$ in 2015–Provincial and territorial population-weighted averages and maximum area-weighted CD estimates

Province and territory	Population	NO_2–ppbv		O_3 summer–ppbv		O_3 annual–ppbv		$PM_{2.5}$–µg/m^3	
		PT p-w avg	CD a-w max	PT p-w avg	CD a-w max	PT p-w avg	CD a-w max	PT p-w avg	CD a-w max
Newfoundland and Labrador	527,756	0.1	0.1[e]	0.3	0.5[n]	0.1	0.2[n]	< 0.1	< 0.1[e,n,u,v]
Nova Scotia	943,002	0.2	0.2[f]	0.7	0.9[o]	0.3	0.5[o]	< 0.1	< 0.1[f,o,w]
Prince Edward Island	146,447	0.2	0.2[h]	0.8	0.8[h]	0.4	0.4[t]	0.1	0.1[h]
New Brunswick	753,871	0.1	0.2[d]	0.8	1.0[m]	0.4	0.5[m]	< 0.1	0.05[d]
Quebec	8,263,600	2.5	5.6[i]	1.4	3.1[q]	-0.1	-1.5[i]/1.1[q]	0.5	0.9[i]
Ontario	13,792,052	2.2	4.9[g]	0.7	-0.3[g]/2.5[p]	-0.2	-1.8[g]/0.9[p]	0.5	0.9[g]
Manitoba	1,293,378	1.9	3.2[c]	0.8	1.3[l]	-0.2	-0.6[c]/0.5[l]	0.3	0.4[c]
Saskatchewan	1,133,637	< 0.1	0.4[j]	1.0	1.2[j]	0.4	0.4[j]	0.1	0.1[j]
Alberta	4,196,457	1.3	1.7[a]	1.6	2.4[a]	0.3	-0.1[r]/0.6[s]	0.2	0.3[a]
British Columbia	4,683,139	2.2	3.7[b]	0.6	-0.3[b]/5.2[k]	-0.3	-1.3[b]/ 2.2[k]	0.4	0.6[b]
Nunavut	36,919	0	0	< 0.1	0.1	< 0.1	< 0.1	0	0
Northwest Territories	44,088	< 0.1	< 0.1[x]	0.2	0.2[y]	< 0.1	0.1[y]	0	< 0.1[z]
Yukon	37,428	< 0.1	N/A	0.1	N/A	< 0.1	N/A	0	N/A
Canada	**35,851,774**	**2.0**	**5.6[i]**	**0.9**	**5.2k**	**-0.1**	**-1.8[g]/2.2[k]**	**0.4**	**0.9[g]**

CD a-w max: maximum area-weighted CD estimate; µg/m^3: micrograms per cubic metre; ppbv: part per billion by volume; PT p-w avg: provincial or territorial population-weighted average; N/A: not applicable, values only available for one CD.

Values are rounded to the second decimal for NO_2, O_3, $PM_{2.5}$ and SO_2, except for nil values. For maximum values, the CD associated with the highest positive (and negative, if applicable) estimate is identified.

[a] Division No. 6 (CD4806);
[b] Greater Vancouver (CD5915);
[c] Division No. 11 (CD4611);
[d] Westmorland (CD1307);
[e] Division No. 1 (CD1001);
[f] Halifax (CD1209);
[g] Toronto (CD3520);
[h] Queens (CD1102);
[i] Laval (CD2465);
[j] Division No. 6 (CD4706);
[k] Central Okanagan (CD5935);
[l] Division No. 2 (CD4602);
[m] Madawaska (CD1313);
[n] Division No. 4 (CD1004);
[o] Hants (CD1208);
[p] Prescott and Russell (CD3502);
[q] Nicolet-Yamaska (CD2450);
[r] Division No. 11 (CD4811);
[s] Division No. 3 (CD4803);
[t] Prince (CD1103);
[u] Division No. 7 (CD1007);
[v] Division No. 9 (CD1009);
[w] Kings (CD1207), Colchester (CD1210) and Cape Breton (CD 1217);
[x] multiple CDs;
[y] Region 5 (CD6105);
[z] Region 6 (CD6106).

Table 16 also indicates the maximum area-weighted values for CDs, i.e., the average contribution from Canadian on-road vehicle emissions to ambient air pollutant levels over an entire CD area. Higher contributions to air pollution are associated with populous CDs, including Division 6 (Calgary) in Alberta and Toronto in Ontario, but more sparsely populated CDs are also affected considerably by on-road vehicle emissions, including the Nicolet-Yamaska CD in Quebec and the Madawaska CD in New Brunswick. The comparatively high impacts from on-road vehicle emissions in sparsely populated CDs reflects local emissions as well as the atmospheric transport of traffic pollution from regions further away and generally upwind.

Meng et al. (2019) investigated the contributions of North American emission source sectors to ambient $PM_{2.5}$ concentrations across Canada in 2013. They used the GEOS-Chem chemical transport model combined with satellite-derived $PM_{2.5}$ across North America, achieving a final model grid resolution of 1 km. Emissions data sources included the Canada's Air Pollutant Emissions Inventory and the US National Emissions Inventory (NEI 2011 v6.3), as well as default emissions datasets in GEOS-Chem (e.g., biogenic, wildfires). They quantified contributions using brute force sensitivity simulations targeting five individual sectors, including transportation. Transportation emissions combined on-road and off-road mobile sources, as well as dust from paved and unpaved roads. Population data from the National Aeronautics and Space Administration Socioeconomic Data and Application Center[23] were used to calculate regional and provincial population-weighted average $PM_{2.5}$ concentrations in Canada. Meng et al. (2019) estimated that the Canadian average population-weighted reference $PM_{2.5}$ concentration was 5.5 $\mu g/m^3$ for calendar year 2013, slightly higher than the 5.3 $\mu g/m^3$ annual population-weighted average for calendar year 2015 modelled in this analysis. Canadian transportation emissions accounted for 10%, or approximately 0.60 $\mu g/m^3$, of population-weighted $PM_{2.5}$ concentrations.[24] The estimated contribution from Canadian transportation emissions to ambient $PM_{2.5}$ by Meng et al. (0.60 $\mu g/m^3$) is approximately 50% higher than the Canadian on-road contribution to ambient $PM_{2.5}$ estimated in the current analysis (0.4 $\mu g/m^3$ or 7%; see Table 3). However, the results from the current analysis are not directly comparable to those from Meng et al. (2019) without adjustments. Fundamental differences were noted in the definition of transportation and the simulation scenarios. First, transportation in the current analysis targeted on-road vehicle emissions only and excluded road dust emissions. By contrast, Meng et al. included on-road and off-road mobile source emissions, as well as dust from paved and unpaved roads. As indicated in Table 1, Canadian off-road mobile sources released 18,467 tonnes of $PM_{2.5}$ emissions in 2015, and dust from paved and unpaved road contributed approximately 500,000 tonnes to $PM_{2.5}$ emissions.[25] These source emissions far exceed the 13,477 tonnes of $PM_{2.5}$ emissions associated with on-road transportation. Off-road mobile emissions are associated with a variety of applications and are geographically distributed across urban and rural areas. Based on previous Health Canada health impact assessments of Canadian gasoline and diesel mobile source emissions, off-road applications contributed approximately 28% of transportation-related population-weighted $PM_{2.5}$ concentrations (refer to Health Canada 2016a, 2017). Second, only Canadian on-road vehicle emissions were perturbed in the brute force simulations for the current analysis, whereas Meng et al. excluded those from Canada and the United States. The authors reported the fraction of population-weighted $PM_{2.5}$ concentrations attributable to Canadian transportation emissions at 10%, or approximately 0.60 $\mu g/m^3$ (63% of the estimated contribution from the transportation sector). Considering the possible contribution from off-road sources (approximately 30% of transportation-related air pollution; Health Canada 2016a, 2017), and a contribution from road dust, the estimates from this analysis and those

[23] Gridded Population of the World, v4, SEDAC. Available online at: http://sedac.ciesin.columbia.edu/data/collection/gpw-v4

[24] According to Meng et al. (2019) North American transportation emissions contributed 16%, or 0.96 $\mu g/m^3$, to ambient population-weighted $PM_{2.5}$ over Canada in 2013. This contribution was approximately equivalent to those from wildfires (1.0 $\mu g/m^3$) and residential combustion (0.91 $\mu g/m^3$). Transportation emissions originating from the United States were estimated to account for 6% of population-weighted $PM_{2.5}$ concentrations across Canada (Meng et al. 2019). The contribution from the United States was subtracted from the North American contribution to estimate the share attributable to Canadian transportation emissions.

[25] Road dust emission are generated via abrasion and re-suspension processes that differ from those influencing exhaust emissions, and the atmospheric transport of road dust emissions is expected to be different than exhaust emissions. Moreover, dust from unpaved roads may be generally released in lower population, non-urban areas.

from Meng et al. (2019) generally concur. In addition, there are meaningful differences associated with datasets (e.g., versions of the emission inventories, meteorology) and modelling tools (e.g., GEM-MACH or GEOS-Chem) that must be accounted for. Additional analyses that are beyond the scope of this assessment would be required to thoroughly compare results from these assessments.

4.2. ESTIMATES OF POPULATION HEALTH BURDEN

Exposure to TRAP was associated with 1,200 premature deaths in Canada in 2015. Non-fatal health outcomes included 2,700,000 acute respiratory symptom days, 1,100,000 restricted activity days, 210,000 asthma symptom days, 610 emergency room visits, and 170 hospital admissions. The total annual monetary value of the health burden was $9.5 billion (CAD 2015), with $9 billion being associated with premature deaths. Higher numbers of health outcomes were estimated in populous areas including Metro Vancouver, the GTHA and Greater Montreal, owing to the confluence of higher TRAP concentrations and larger exposed populations.

The air quality modelling for this analysis was conducted at a grid resolution of 10 km, which is too coarse to capture the local variability and magnitude in exposures to TRAP, such as those experienced by populations near high-traffic roadways. This limitation likely leads to an underestimation of population exposure to TRAP in urban areas. It may possibly have the opposite effect in suburban areas that share part of a grid cell with high pollution areas. It is important to consider that the CRFs in AQBAT are mostly derived from air pollution health studies based on monitoring data collected at centrally located sites. These central locations have usually been selected to reflect regional air pollution levels and the AQBAT results are valid representations of regional population-level impacts. The air quality grid resolution was more appropriate for regional exposure estimates, as represented by CDs in this analysis, and it is not necessarily a limitation for health impact analyses. The geographic resolution for AQBAT is discussed further in section 4.3.3.

Complementary simulations targeting segments of the on-road fleet showed that light-duty vehicle air pollution contributed to 420 premature deaths per year and that heavy-duty vehicle air pollution contributed to 730 premature deaths per year.[26] The small difference between the total TRAP health impacts (1,200)[27] and the sum of the impacts from light-duty and heavy-duty vehicle air pollution (1,150) is associated with the brute force approach (see section 4.3 for a discussion of methodological considerations).

Health Canada previously estimated health impacts attributable to above-background air pollution in Canada in 2015 (Health Canada 2019) using a methodological framework similar to the current assessment. The number of premature deaths attributable to air pollution was 14,600 per year, and the total economic cost of all health impacts attributable to air pollution was $114B (CAD 2015). Although the current assessment of health impacts associated with TRAP is not directly comparable, the assessment framework shares similarities including the use of AQBAT. Generally, based on a comparison of premature death estimates from the two analyses, the results suggest that TRAP contributes approximately 8% of air pollution-related health impacts in Canada. Thakrar et al. (2020) estimated mortality in the United States attributable to all domestic, anthropogenic emissions of

[26] Unpublished analysis by Health Canada.

[27] 1,220 if three significant digits are retained.

primary $PM_{2.5}$ and secondary $PM_{2.5}$ precursors. They reported that 19,700 of the 100,000 premature deaths per year were associated with transportation activities, suggesting that approximately 20% of premature death associated with anthropogenic $PM_{2.5}$ per year are attributable to transportation activities.[28] Additional analyses that are beyond the scope of this assessment would be required to compare these assessments in detail.

4.2.1. Comparison with previous assessments of air pollution health impacts linked to on-road diesel and gasoline vehicle emissions in Canada

Previous human health risk assessments by Health Canada for diesel exhaust (Health Canada 2016a) and gasoline exhaust (Health Canada 2017) included air quality modelling that used a brute force framework comparable to the current analysis. Those assessments estimated the health impacts of on-road diesel vehicle emissions and of on-road gasoline vehicle emissions for calendar year 2015.[29] The diesel exhaust and gasoline exhaust assessments were based on the same sets of data and simulations with A Unified Regional Air Quality Modelling System (AURAMS) and were directly comparable. The current TRAP assessment used updated datasets and tools (e.g., GEM-MACH) to conduct the simulations. Table 17 compares the estimated contributions to ambient $PM_{2.5}$ for the different simulations, including those for TRAP presented in this report. The modelled contributions associated with on-road gasoline vehicle emissions and on-road diesel vehicle emissions were combined to approximate the total population-weighted contribution from all on-road vehicle emissions. Table 18 provides a similar comparison for modelled NO_2 estimates. Overall, the estimates were generally similar across simulations for the national estimates. At the provincial level, the results indicate comparable estimates (less than 30% difference) for NO_X with the exception of Newfoundland and Labrador. For $PM_{2.5}$, differences exceed 30% in half of the provinces including Alberta, British Columbia, Prince-Edward Island, Quebec and Saskatchewan.

[28] Transportation activities included passenger vehicle use, truck use, light commercial vehicle use, and municipal vehicle use.

[29] The analyses targeting on-road gasoline and on-road diesel vehicle emissions used 2015 emission projections (i.e. future forecasts). Refer to the Health Canada human health risk assessments for diesel exhaust (Health Canada 2016a) and gasoline exhaust (Health Canada 2017) for more details. By contrast, the current analysis used a validated emissions inventory for the year 2015 that was finalized in 2017. They represent different versions of the 2015 emissions inventory. While the calendar years correspond, this is coincidental.

TABLE 17: Contributions from Canadian on-road gasoline vehicle emissions, on-road diesel vehicle emissions, and TRAP to ambient $PM_{2.5}$ concentrations in 2015–Provincial and national estimates–Population-weighted annual average

Province	Relative (%) contribution to ambient $PM_{2.5}$ concentrations			
	Gasoline[a]	Diesel[b]	Total	TRAP
Newfoundland & Labrador	0.6	0.3	0.9	1.2
Nova Scotia	1.2	0.6	1.8	2.3
Prince Edward Island	0.7	0.8	1.5	3.6
New Brunswick	1.3	1.1	2.4	2.7
Quebec	4.6	3.6	8.2	5.9
Ontario	6.6	2.1	8.7	7.1
Manitoba	3.0	5.2	8.2	10
Saskatchewan	2.2	2.1	4.3	6.2
Alberta	2.6	2.3	4.9	8.0
British Columbia	9.4	3.9	13	9.6
Canada	**5.7**	**2.8**	**8.5**	**7.0**

Values are rounded to one decimal and given to a maximum of two significant figures.

[a] Health Canada 2017;

[b] Health Canada 2016a

TABLE 18: Contributions from Canadian on-road gasoline vehicle emissions, on-road diesel vehicle emissions, and TRAP to ambient NO_2 concentrations in 2015–Provincial and national estimates–Population-weighted annual average

Province	Relative (%) contribution to ambient NO_2 concentrations			
	Gasoline[a]	Diesel[b]	Total	TRAP
Newfoundland & Labrador	5.3	6.7	12	23
Nova Scotia	9.9	13	23	30
Prince Edward Island	14	25	38	36
New Brunswick	11	19	30	27
Quebec	17	21	38	45
Ontario	14	14	28	35
Manitoba	22	33	55	51
Saskatchewan	11	15	26	24
Alberta	7.6	10	18	25
British Columbia	29	21	50	47
Canada	**16**	**17**	**33**	**38**

Values are rounded to one decimal and given to a maximum of two significant figures.

[a] Health Canada 2017;

[b] Health Canada 2016a

ECCC's previous chemical transport model AURAMS and GEM-MACH produced similar estimates for $PM_{2.5}$ and NO_2 in terms of the magnitude and geographic distribution of these pollutants or their precursors. The consistent results also suggest that the relative contribution of on-road vehicle emissions to the overall $PM_{2.5}$ and NO_2 inventory were similar between simulations. Large differences in model input (for example, a decrease in the magnitude of on-road emissions and an increase in emissions for other sectors from one inventory to another), as well as the difference in model spatial resolution between GEM-MACH and AURAMS, could have markedly influenced modelling results and their comparability. For ambient O_3 concentrations, direct comparisons in both magnitude and direction of change are more difficult to interpret than for $PM_{2.5}$ and NO_2. Ground-level O_3 is solely a secondary pollutant (i.e., absence of direct or primary emissions) and ambient air concentrations depend on precursors, including NO_X and VOCs, and several non-linear atmospheric chemical mechanisms and interactions. Table 19 presents the relative contributions to ambient summer average ambient 1-hour maximum O_3 concentrations. The influence of on-road traffic (gasoline, diesel or all vehicles) on air quality in large urban CDs, by contrast to national and provincial estimates, was more often associated with decreases in O_3.

TABLE 19: Contributions from on-road gasoline vehicle emissions, on-road diesel vehicle emissions and TRAP to summer average ambient 1-hour maximum O_3 concentrations in 2015–Provincial and national estimates–Population-weighted annual average

Province	Relative (%) contribution to ambient O_3 concentrations			
	Gasoline[a]	Diesel[b]	Total	TRAP
Newfoundland & Labrador	0.4	0.5	0.9	0.8
Nova Scotia	0.8	1.0	1.8	1.8
Prince Edward Island	0.8	1.1	1.9	2.4
New Brunswick	0.8	1.0	1.8	2.3
Québec	0.5	–1.4	–0.9	3.8
Ontario	0.9	0.1	1.0	1.5
Manitoba	1.3	1.2	2.5	2.3
Saskatchewan	1.4	1.6	3.0	2.8
Alberta	1.5	0.7	2.2	3.7
British Columbia	2.0	–1.1	0.9	1.5
Canada	**1.0**	**–0.2**	**0.8**	**2.3**

Values are rounded to one decimal and given to a maximum of two significant figures.

[a] Health Canada 2017;

[b] Health Canada 2016a

Different versions of datasets (e.g., APEI, NPRI) and tools (e.g., MOVES, SMOKE, GEM-MACH, AURAMS) were used for the TRAP assessment than for the gasoline and diesel assessments. For example, the gasoline and diesel assessments were based on earlier versions of the APEI and census data, as well as on AURAMS, and they applied a somewhat different set of concentration-response functions in AQBAT. The modelling year was also 2015, but emissions were projected from a different, earlier reference year. Moreover, on-road vehicle emissions were modelled for the gasoline and diesel assessments using a combination of the MOBILE6.2C model for LDVs and the MOVES 2010a model for HDVs, while vehicle emissions were simulated with MOVES 2014b for all on-road vehicle classes in the current assessment of TRAP. Spatial surrogates used by SMOKE to

allocate provincial total emissions to each grid cell may be different between these two studies. Lastly, the geographical resolution of AURAMS in the diesel and gasoline exhaust assessments was 22.5 km, compared to 10 km in this assessment. These differences in data and tools introduce some uncertainty when comparing results between analyses. Refer to section 4.3 for a more complete discussion of methodological considerations.

The previous analyses estimated a Canadian air pollution health burden of $5.4 billion (CAD 2015) with 700 premature deaths (Health Canada 2017) for on-road gasoline vehicles, and a health burden of $2.5 billion (CAD 2015) with 320 premature deaths for on-road diesel vehicles (Health Canada 2016a). The current estimate of premature deaths associated with all Canadian on-road vehicle emissions (1,200) is quite similar, at 18% higher than the total of the gasoline and diesel assessments (1,020). In addition to differences in air pollution exposures, the selection of the concentration response function (CRF) for $PM_{2.5}$ also varied. A more recent version of AQBAT (version 3.0) was used for the current assessment than for the gasoline and diesel analyses (AQBAT version 2.1) in order to account for new epidemiological data. AQBATv3.0 includes a revised risk estimate for $PM_{2.5}$ mortality that reflects the findings from Crouse et al. (2012) on associations between long-term exposure to ambient PM2.5 concentrations and non-accidental mortality in Canadian adults. Chronic exposure premature mortality associated with $PM_{2.5}$ was previously represented by four endpoints in AQBATv2.1: cerebrovascular mortality, chronic obstructive pulmonary disease mortality, ischemic heart disease (IHD) mortality, and lung cancer mortality. AQBAT estimates derived using the non-accidental $PM_{2.5}$ mortality CRF from Crouse et al. (2012) increase national $PM_{2.5}$ mortality impacts by approximately 56% compared with the aggregated (four causes) $PM_{2.5}$ mortality values published in the gasoline and diesel assessments. Total premature deaths associated with on-road gasoline and diesel emissions would increase by 270 and 140, respectively, with the all-cause mortality $PM_{2.5}$ CRF, reaching a total of 1,430 premature deaths in 2015 (Table 20). The CRFs for other pollutants in AQBAT have not been modified between versions. The estimate of 1,200 premature deaths from the current 2015 analysis is similar (within 20%) to those resulting from the application of the all-cause mortality CRF or multiple CRFs in the original gasoline and diesel analyses, suggesting that the methodologies employed by Health Canada to estimate health impacts from TRAP are consistent overall.

TABLE 20: Premature death associated with Canadian on-road gasoline vehicle emissions, on-road diesel vehicle emissions and all on-road vehicle emissions using various CRFs for $PM_{2.5}$ chronic exposure mortality–National estimates for calendar year 2015

Assessment and $PM_{2.5}$ CRF type	Total premature deaths–all pollutants		
	On-road gasoline	On-road diesel	All on-road
TRAPa–all cause mortality	n.a.	n.a.	1,200
On-road G&D[b,c]–4 causes of mortality	700	320	1,020
On-road G&D[b,c]–all cause mortality	970	460	1,430

CRF: concentration response functions; G&D: gasoline and diesel emissions; n.a.: not available

[a] Current analysis;

[b] Health Canada 2017; [c] Health Canada 2016a

Analyses using the identical modelling framework and datasets as in the current assessment were also completed to estimate impacts associated with LDV and HDV fleets individually. The simulation scenarios were defined as follows:

- LDV scenario: on-road light-duty vehicle emissions excluded from the inventory.
- HDV scenario: on-road heavy-duty vehicle emissions excluded from the inventory.

It was estimated that LDV emissions were responsible for 420 premature deaths in 2015, while HDV emissions were responsible for 730 premature deaths. The sum of these estimates (1,150) is very close to that for all on-road vehicle emissions (1,200 deaths).

Since most LDVs are gasoline-powered and most HDVs presumably use diesel fuel, and since all simulations targeted the year 2015, comparisons were also made with the previous diesel and gasoline assessments published by Health Canada. Similarities were expected between gasoline (700 deaths) and LDV results (420 deaths), and between diesel (320 deaths) and HDV results (700 deaths), but the estimates diverge considerably. Data from the APEIs used for the simulations highlight possible sources of these differences. The make-up of the LDV and HDV fleets is not directly equivalent to the gasoline and diesel fleets. For example, the APEI used in the current assessment shows that heavy-duty gasoline trucks (HDGTs) release higher emissions of NO_X (37,000 tonnes), $PM_{2.5}$ (975 tonnes) and VOCs (12,600 tonnes) than light-duty diesel vehicle and trucks (1,900 tonnes of NO_X, 23 tonnes of $PM_{2.5}$, and 1,700 tonnes of VOCs), which could account for the higher HDV health impacts compared to diesel vehicles (as a surrogate of HDVs).

Comparisons with the emissions inventory used in the previous diesel and gasoline assessment (also for the year 2015 but using different projections) show important differences in NO_X and $PM_{2.5}$ emissions (i.e., of more than 50%) for some on-road vehicle sub-classes. For example, NO_X emissions for heavy-duty diesel vehicles (HDDVs) in the current APEI are increased by 51% (8,000 tonnes), $PM_{2.5}$ emissions for HDGVs are increased by more than three-fold (750 tonnes), while those for light-duty diesel trucks (LDDTs) are reduced by more than 90% (170 tonnes) compared to the earlier estimates. Emissions inventory projections change over time in light of new data and trends associated with technological and socio-economic parameters, such as personal vehicle activity (e.g., vehicle-kilometres travelled and fuel use), commercial activity (e.g., increase freight transport and home deliveries), vehicle sales (e.g., increase in sport utility vehicles, reduction in share of diesel vehicles), overall economic activity, and updated information for emission rates under different operation conditions. Variations in emissions can have a meaningful influence on air quality and health impact estimates. Hence, while the overall values in Table 20 seem to converge, additional analyses that are beyond the scope of this assessment would be required to thoroughly compare and interpret data across different assessments. Overall, the 2015 emissions inventory used in the current analysis, published in 2017, is considered more accurate than previous ones. Notably, it relies on reported activity data for the year 2015, by contrast to projections developed prior to the year 2015. In addition, more recent emission inventories benefit from incremental and continuous improvements in methodology and data.

4.3. METHODOLOGICAL CONSIDERATIONS

Efforts were made to use the best available emissions, air quality and health impact modelling tools and data available for the current health impact assessment of TRAP. However, limitations and uncertainties exist. Uncertainty in modelling emissions, ambient air pollution concentrations, and health impacts originate from a variety of sources, including the availability and quality of Canadian data on vehicle fleets and vehicle emission factors, photochemical algorithms in the chemical transport model, meteorological data, and health data to support the selection of CRFs that are relevant to the Canadian population. Moreover, assumptions made during the early stages of the modelling framework (e.g., those made for developing the APEI) are reflected in subsequent stages of the analysis (e.g., air quality modelling) and may affect the final health estimates.

This section addresses several categories of uncertainties associated specifically with the analysis of health impacts. Generic and scenario-specific uncertainties were characterized qualitatively, and scenario-specific sensitivity analyses were included for some parameters. Uncertainties were evaluated when possible according to their potential influence on the direction and magnitude of estimated health impacts, as well as to the degree of evidence available to support the approach or assumptions selected for the current assessment. This approach borrowed from, but did not replicate, the World Health Organization (WHO) uncertainty framework (WHO 2008) and the US EPA qualitative assessment of uncertainty approach outlined in the assessment of the PM National Ambient Air Quality Standard (2012).

4.3.1. Emissions inventory

The development of air emission inventories depends on the availability and processing of data for releases to air from all sources and sectors in Canada. Emission inventories are dynamic, with new versions being regularly developed for different policy or scientific purposes. Air quality modellers select the version that best meets their needs in terms of accuracy, reliability, spatial resolution and time period, and that helps to address a specific policy or research question (Zhang et al. 2018).

For the APEI in general, overestimation or underestimation of releases to air from individual sectors could influence the air quality impact estimates for TRAP. The potential bias could be enhanced for sectors that share a common geographic distribution with on-road vehicle emissions. Further, inventories are deterministic in nature and do not provide ranges of possible values. Uncertainties associated with the version of the emissions inventory used for the current analysis were not quantified.

The spatial surrogates that are used to geographically distribute emissions from vehicles and other sources across Canada are also imperfect. Spatial surrogates are developed using Canadian or other data. For on-road vehicles, road network data, population densities and vehicle activity data can be used to distribute vehicle emissions geographically across Canada. Assumptions are required to simplify the process (e.g., vehicle activity based on road classification) and compensate for possible gaps in data. As the spatial allocation of emissions is a continually improving process, uncertainty analyses are not systematically undertaken. Uncertainties associated with spatial surrogates have not been quantified.

4.3.2. Air quality modelling

4.3.2.1. Geographic scale and grid resolution

One strength of the air quality modelling framework used in the current analysis was the provision of estimates for every census division in Canada based on the same exposure assessment approach, for all pollutants. The use of a consistent method to assess exposure to TRAP across all of Canada, although uncommon, allowed for direct comparisons across many regions. However, the national coverage comes at the cost of a lower geographic resolution (i.e., a 10-km grid) that was too coarse to estimate local-scale exposure to TRAP. Approaches allowing for higher resolution estimates are available, but they are often limited geographically to a smaller study domain and apply to a limited number of air pollutants. For example, land use regression models integrating road and traffic characteristics are generally city-specific and incorporate multiple sources, dispersion models can be limited to one or a few road segments, and source apportionment analyses, such as positive matrix factorization, are often based on observations at a single monitoring site. Each approach can provide some information in relation to traffic as a source of exposure to air pollution. However, these alternative approaches for estimating exposure to TRAP are also associated with limitations and uncertainties, and method selection will depend on study objectives and conditions. The approach selected for the current analysis was considered the most appropriate and consistent for an assessment of the health impacts associated with TRAP across Canada.

The grid resolution can impact air modelling results and introduce uncertainty in exposure estimates (e.g., Arunachalam et al. 2011; Isakov et al. 2007; Stroud et al. 2011). In general, larger grid cells (i.e., lower grid resolution) provide an average of pollutant emissions and concentrations over a wider area (smoothing), and this can affect how emissions and concentrations are allocated to local sources (Tessum et al. 2012). For example, for modelling domains with low grid resolutions, urban and rural areas may be included in the same model grid, which may artificially dilute urban emissions to the surrounding rural areas (Galarneau et al. 2014; Whaley et al. 2020). This situation can lead to population exposure misclassification by overestimating air pollution in rural areas or underestimating air pollution in urban areas. The air quality modelling for this analysis was conducted at a grid resolution of 10 km, which is a reasonably high resolution for a national assessment. However, 10-km grid cells are too coarse to capture local or road-scale exposures to TRAP (Stroud et al. 2011). An entire urban area may be represented by a few grid cells, and these will not accurately reflect the spatial variability in air pollution (Queen and Zhang 2008). This limitation likely leads to an underestimation of population exposure to TRAP in urban areas. It may possibly have the opposite effect in suburban areas that share part of a grid cell with high pollution areas. Owing to the spatial resolution of the chemical transport model simulations (i.e., 10-km grid cell), the current analysis did not specifically assess local risk levels for communities adjacent to high-traffic roadways that are directly impacted by TRAP. The grid resolution was more appropriate for regional exposure estimates, as represented by CDs in the health burden evaluation (see section 4.3.3.2). Although this uncertainty was not quantitatively evaluated, the sensitivity of pollutant concentrations to grid resolution is of importance because health outcomes and their corresponding economic evaluations are driven mainly by $PM_{2.5}$ concentrations and, to a lesser extent, NO_2 and O_3 concentrations. Mischaracterizing concentrations for these pollutants could significantly influence population risk estimates. The uncertainty associated with this misclassification of population exposure levels was not estimated in the current analysis.

Provincial, territorial and national average ambient air pollutant concentrations were calculated from the CD estimates using a population-weighting approach, which accounts for the distribution of population across geographic areas. While population-weighting provides a more accurate representation of exposures at the provincial, territorial and national levels than an area-weighted approach, it still involves uncertainties associated with the influence of smoothing.

4.3.2.2. Unaccounted-for chemical mechanisms and secondary pollutants

Photochemical modelling is a complex exercise and the current models, although highly advanced, may not fully account for secondary pollutants that are governed by the characteristics of primary emissions, photochemical processes and environmental factors. The GEM-MACH model incorporates several, but not all, indirect and heterogeneous pathways for organic (e.g., VOCs) and inorganic (e.g., NO_3^- and SO_4^{2-}) aerosols (Burr and Zhang 2011). For example, it has been suggested that modelling simulations generally under-predict organic aerosol concentrations for a number of reasons, such as unaccounted-for intermediate VOCs in the emission inventories, aerosol-phase chemistry effects and insufficient or non-representative SOA data (Gentner et al. 2017; Lu et al. 2020; Stroud et al. 2011). As secondary PM formation via aerosols released from vehicle exhaust contributes to total $PM_{2.5}$ concentrations in ambient air, under-predicting aerosol levels would underestimate the $PM_{2.5}$ effect associated with traffic-related emissions. The direction and magnitude of the potential bias were not assessed in the current assessment.

4.3.2.3. Averaging times

The ambient air concentration estimates for NO_2, $PM_{2.5}$ and SO_2 were based on annual averages. By contrast, O_3 concentration estimates were based on daily 1-hour maximum concentrations. Moreover, to account for significant seasonal variations, annual O_3 and summer O_3[30] averages were estimated. The choice of the air pollution exposure metric was dependent on the epidemiological data used to define the CRFs. For example, annual average $PM_{2.5}$ concentrations were estimated because the more robust epidemiologic data showed associations between annual $PM_{2.5}$ levels and health outcomes. However, it is recognized that pollutant concentrations vary daily and seasonally owing to changes in mobile and stationary source emissions, and atmospheric and meteorological conditions (Demerjian and Mohnen 2008; Hu et al. 2009; Zhu et al. 2006). Consequently, the annual average estimates may not truly reflect temporal variations in personal exposures to air pollutants. Notwithstanding these uncertainties, the availability and selection of CRFs determined the air quality projections and population exposure estimates that were relevant for the estimation of health impacts.

4.3.2.4. Modelling approach

The modelling results were a product of the selected modelling approach. The brute force modelling method is a practical option for assessing the effects associated with a change in emissions, or those attributed to a specific source sector, such as TRAP. It is a sensitivity (to source) analysis method. Two simulations were completed and the results compared to assess the air quality impacts from TRAP. The positive contributions from TRAP were often interpreted as source contributions, whereas negative TRAP contributions were interpreted as sensitivity of the model predictions to the on-road vehicle emissions that were excluded. However, the brute force method is not a true source

[30] Summer O_3 corresponds to values from May 1 to September 30.

apportionment method: The relative contributions from all emissions sectors, if modelled independently by zeroing-out, do not sum to 100%. This is the case if the relationship between emissions and concentrations is non-linear (Zhang et al. 2014). In other words, a brute force analysis conducted for each source sector independently would not truly represent the source contributions because the modelled perturbation inherently excludes the interactions between source emissions from different sectors.

Interpretation of the brute force results was also limited by the fact that compensating effects are indistinguishable. Upon a reduction in on-road transportation emissions, the rate of secondary pollutant formation per molecule of primary source emissions is altered. For example, a reduction in on-road NO_X emissions can lead to situations where NO_X emissions from a non-traffic source could be producing O_3 more efficiently compared with the reference scenario, subsequently underestimating the contribution from TRAP to O_3 concentrations. This was exemplified in Table 14 with the O_3-related reduction of premature deaths associated with traffic emissions. While this compensating effect should be captured in the simulation, the influence associated with TRAP explicitly is not estimated. This is one of the main reasons why the sum of the air pollution health contributions from the LDV fleet and the HDV fleet are not exactly equivalent to the TRAP results (see section 4.2). The brute force method shows only the net or incremental change in pollutants associated with variations in emissions, lowering the ability to assess the relative benefits of different emission regulations or to identify more efficient pathways for improving air quality by targeting specific sources. Different sensitivity analysis tools (e.g., decoupled direct method) could be tested and compared to brute force, and more complex modelling tools (e.g., tagged species; adjoint modelling) could be used in the future to address limitations for distinguishing the air quality effects linked to specific emission sources (Grewe et al. 2012; Koo et al. 2009; Samaali et al. 2011). However, testing different approaches is resource intensive. More importantly, it must be noted that the brute force and other methods provide answers to slightly different questions. For example, the brute force approach identifies air quality impacts associated with a change in emissions, whereas the tagged-species method determines the contributions from specific sources. The nature of these estimates is not interchangeable (Grewe et al. 2012). An assessment of the uncertainties associated with the brute force approach compared to other methods was beyond the scope of the current analysis. The direction and magnitude of the potential bias were not assessed.

4.3.2.5. Model performance analysis

Performance of the GEM-MACH model has been evaluated previously and reported in the peer-reviewed literature (Makar et al. 2014a, 2014b; Whaley et al. 2018). The performance of GEM-MACH is generally comparable or better than other CTMs, including ECCC's previous model AURAMS.

A performance analysis was conducted for the specific combinations of the APEI, spatial surrogates and the GEM-MACH version used in the current analysis. Modelled ambient air concentrations for the year 2015 were compared with observations at ground monitoring stations across Canada for the year 2017. Observation data correspond with measurements collected at monitoring stations associated with different networks, including the National Air Pollution Surveillance (NAPS) program, the Canadian Air and Precipitation Monitoring Network, and provincial or regional monitoring networks. Observations for the year 2017 (rather than 2015) were used to match the 2017

meteorological data (see section 2.2.1) owing to the influence of meteorology on ambient air pollutant concentrations. Annual, seasonal, monthly and daily comparisons were conducted for NO_2, O_3 and $PM_{2.5}$, and mean bias error (MBE), Pearson correlation coefficient, and root mean square error (RMSE) were reported.

Overall, the 2015 base-case predictions were comparable to the 2017 observations. The annual MBE and Pearson correlation coefficient showed good agreement for NO_2 and O_3, whereas larger differences for $PM_{2.5}$ were observed. Greater MBE and weaker correlations for $PM_{2.5}$ were expected as natural forest fire emissions, which significantly impact ambient $PM_{2.5}$ observations, were not included in the 2015 base case. The RMSE metric is sensitive to absolute values and outliers (each error is squared); larger differences have considerable impact on RMSE. O_3 has larger concentration fluctuations compared to NO_2, while $PM_{2.5}$ is subject to extreme values during wildfire events. Table 21 summarizes the annual performance evaluation metric values for Canada, eastern Canada, and western Canada.

Regionally, performance metrics were generally better for Canada as a whole and eastern Canada compared to western Canada. The more frequent and extensive forest fires in western Canada are partly responsible for lower model performance in that region (NRCan 2016).

TABLE 21: Annual performance evaluation statistics comparing the 2015 base case estimates in GEM-MACH and 2017 observations

Region	MBE	Correlation[a]	RMSE
NO_2			
Canada	1.1	0.6	7.3
Eastern Canada	0.9	0.7	6.3
Western Canada	1.2	0.6	8.1
O_3			
Canada	-2.7	0.7	10.2
Eastern Canada	-2.3	0.7	9.5
Western Canada	-3.2	0.6	11.2
$PM_{2.5}$			
Canada	-1.3	0.3	10.1
Eastern Canada	-0.4	0.4	8.2
Western Canada	-2.4	0.2	11.9

MBE: Mean bias error; RMSE: Root mean square error

[a] Pearson correlation coefficient

The summer statistical values are similar to the annual values for NO_2 and O_3. By contrast, lower scores are estimated for $PM_{2.5}$ owing to the influence of forest fires during the summer season. NO_2 values show better performance than O_3 and $PM_{2.5}$ for RMSE. This is partly due to lower absolute NO_2 concentrations compared to O_3 and $PM_{2.5}$; consequently, a lower RMSE is expected for NO_2.

Additional details of the model performance analysis are included in Appendix D.

4.3.3. Estimates of population health burden

Estimating the population health impacts associated with TRAP is the final step in a sequential process that follows an evaluation of on-road vehicle emissions and their impact on ambient air quality. Estimates of health impacts from TRAP are influenced by the uncertainties in each of the previous steps in the analysis, which were addressed above. Limitations specific to the health impact assessment step were mainly from the CRFs, which reflect the health risks associated with ambient concentrations of air pollutants, as well as from geographical and temporal considerations.

4.3.3.1. Concentration–response functions

The inclusion of air pollutants in quantitative benefits assessment requires the development and use of relative risks (RRs) or CRFs. The main prerequisite is the assumption that causality exists between exposure to a specific air pollutant and one or several health outcomes. Health Canada only considers causal and likely causal relationships for quantitative health impact analyses. Determination of causality includes a weight-of-evidence analysis and consideration of the Bradford-Hill criteria, including: consistency of epidemiological findings reproduced over different geographic areas, periods and study designs; coherence of the observed effects; and biological plausibility supported by clinical or toxicological studies, as well as through intervention studies (i.e., a change in exposure conditions causing a change in outcome) (Brunekreef et al. 2009; Health Canada 2016c; Hill 1965). An extensive exposure assessment is also necessary to derive RRs and CRFs, such as through long-term air monitoring data collected at geographically distributed sites. CRFs should ideally be developed for the population considered in the assessment, or based on a population with similar health and socio-economic characteristics (Barry et al. 2019).

It is important to consider the selection of CRFs when comparing health impact estimates among studies. For example, Thomson et al. (2014) estimated health impacts associated with O_3 and $PM_{2.5}$ using the various CRFs included in the US EPA's Environmental Benefits Mapping and Analysis Program (BenMAP)[31] and found that differences between the highest and lowest mean values among the CRFs could vary by up to 300% for O_3 and 150% for $PM_{2.5}$. The health impact estimates for TRAP were generally more sensitive to the CRF values than they were to the air quality modelling grid resolution, the uncertainties of which were discussed in section 4.3.2.

AQBAT includes a limited number of CRFs for NO_2, O_3, and $PM_{2.5}$ that were derived from peer-reviewed sources and are endorsed by Health Canada. CRFs include some inherent uncertainties that are reflected in the AQBAT output as the 2.5 and 97.5 percentile risk estimates (95% confidence interval). Comparisons of the 95% confidence intervals with the median estimates showed that extreme estimates were generally less than three times the central estimate (see Table 13), suggesting that the conclusions based on the central CRF values were reasonable. However, for endpoints based on exposure to summer O_3, the estimated median and 95% confidence intervals can show a wider range in extreme values (ambient summer O_3 exposure estimates can have negative and positive values).

[31] BenMAP is an open-source computer program developed by the US EPA that calculates the number and economic value of air pollution-related deaths and illnesses; https://www.epa.gov/benmap

The CRFs in AQBAT are based on robust scientific data, represent Health Canada-endorsed values, and are reviewed with each updated version of the tool. AQBAT version 3.0 includes the CRFs for which the available epidemiological and air pollution data supported their selection by Health Canada during model development. Other health outcomes associated with exposure to NO_2, O_3, or $PM_{2.5}$ (e.g., reproductive and developmental outcomes) or different CRFs for the endpoints included in AQBAT have been reported in the literature (e.g., Barry et al. 2019; Health Canada 2013, 2016c; Pinault et al. 2017; US EPA 2009). However, not all adverse health outcomes can be adequately quantified in terms of a CRF based on the available evidence, and in some cases, the weight of evidence remains too limited to conclude a causal or likely causal role for a given air pollutant. Alternative, robust CRFs that are relevant to the Canadian population may be considered in future health impact assessments as the AQBAT is further updated.

On-road transportation emissions are expected to affect ambient concentrations of pollutants (e.g., polycyclic aromatic compounds, VOCs, ultrafine particles) other than those considered herein (Whaley et al. 2020). The population health effects from exposures to other pollutants were not quantified in the current assessment owing to several factors, including (1) the lack of capacity of emissions and air quality modelling systems to address specific constituents, (2) the lack of CRFs for specific constituents, and (3) other data limitations regarding specific pollutants or pollutant characteristics. Because AQBAT does not include all potential health endpoints and air pollutants of concern, the health and economic impacts associated with TRAP were likely underestimated. However, even with the inclusion of additional pollutants and CRFs, $PM_{2.5}$, and to a lesser extent NO_2 and O_3, would likely remain the most important contributors to health impacts. Overall, there is high confidence in the selected CRFs for a Canadian assessment, and it is expected that existing uncertainties pertaining to CRFs lead to minor underestimations of TRAP health impacts.

For the current assessment, it was assumed that all $PM_{2.5}$ constituents (e.g., black carbon, organic carbon, polycyclic aromatic hydrocarbons, and ultrafine particles) were equally toxic on a mass basis and contributed to the incremental premature mortality risk from exposure to ambient $PM_{2.5}$ based on their contribution to $PM_{2.5}$ mass. This assumption was relevant because the CRF for $PM_{2.5}$ was initially derived for urban mixtures of PM species, not for a specific type of PM (Levy et al. 2012). Although the toxicity of PM species associated with TRAP likely differs from other sources (Liu et al. 2019), the data to support quantitative distinctions between PM sources and constituents are currently lacking (US EPA 2019), and the relative toxicity of traffic-related $PM_{2.5}$ components compared with other components is unknown, both in terms of direction and magnitude. In addition, the same CRFs were applied across CDs and did not consider differences among population groups, except for age. It is possible that some population groups are more or less susceptible to adverse health effects than others following exposure to air pollution (Barry et al. 2019; Stieb et al. 2019). However, data and analyses to develop population-adjusted CRFs across Canada were not available.

Estimates of mortality impacts associated with air pollutant levels below the levels observed in epidemiological studies have more uncertainty, owing to lower confidence in the shape of the concentration–response relationship in that exposure range (Evans 2016). This raises questions about whether or not a change in concentration has the same impact for populations residing in more polluted environments (e.g., non-attainment areas) compared with populations in fairly pristine regions. For example, supralinear relationships between exposure to air pollution and health effects have been reported, including for Canada (Burnett et al. 2018; Pinault et al. 2017). A supralinear shape of association indicates higher incremental risks over the lowest range of air

pollution. A large fraction of the Canadian population is expected to be exposed to air pollution levels in the lower range of exposures (e.g., less than 5 μg/m^3 for $PM_{2.5}$). While AQBAT version 3.0 includes a CRF characterized by supralinear shapes, it was not used for the current analysis. The uncertainty associated with low pollutant concentrations, which is highly relevant in Canada, could potentially bias estimates (minor magnitude, both directions). Nonetheless, there is confidence in the approach integrated into AQBAT using linear CRFs, which reflects current scientific evidence.

Overall, it is expected that uncertainties associated with CRFs in the current assessment likely underestimate population health impacts in Canada.

4.3.3.2. Geographic and temporal resolution of health burden estimates

AQBAT results are considered valid representations of regional population-level impacts (results generated for CDs and averaged for larger geographies). However, the results expressed as numbers of health outcomes can be misleading when interpreted on a per-capita basis. The potential spatial correspondence between high exposure and sensitive populations (e.g., households with lower income residing near busy roadways) within cities may also further bias estimates. These issues support the idea that AQBAT results for counts of health outcomes may be considered valid representations of the population-average burden only.

It is important to consider that, with the exception of all-cause premature mortality associated with long-term exposure to $PM_{2.5}$, the CRFs in AQBAT were derived from epidemiological studies that assessed population health effects based on monitoring data collected at centrally located sites. The epidemiological studies for those CRFs did not use high-resolution exposure data, and epidemiological studies based on high-resolution exposure estimates were not readily available. By contrast, recent epidemiological studies have applied spatially refined estimates of annual ambient concentrations in studies of population health effects (Crouse et al. 2012, 2015; Pinault et al. 2017; Stieb et al. 2019). The ability to develop higher resolution exposure surfaces and to combine these with detailed population health data may enable improved detection of associations between air pollution and health effects compared with exposure estimates based on central site data, particularly for TRAP (Brauer et al. 2019; Chen et al. 2020; Dionisio et al. 2013; Özkaynak et al. 2013; Sarnat et al. 2013). The goal of refined approaches is to reduce exposure error and its resulting bias, in order to provide more power to detect potential epidemiologic associations of interest (Baxter et al. 2013). The ability for improved exposure assessments in future air pollution epidemiology studies will reduce uncertainty in health risk assessments of ambient air pollution (Sarnat et al. 2013) and will allow for more detailed analyses of the influence of model grid resolution on population health impacts. As additional analyses assessing population exposures using alternative estimation approaches at different geographic locations and temporal scales become available, notably in Canada, regional and national health impact estimates may improve.

The exposure estimates modelled with GEM-MACH were also based on annual averages (and summer average for O_3) and did not consider short-term temporal variations (e.g., daily and weekly) in ambient pollutant concentrations. For example, the use of annual averages may underestimate high exposure events, such as weekday congestion driving (Evans et al 2019). Although unaccounted for temporal variables likely influenced risk estimates minimally, the direction and magnitude of the bias are uncertain.

CHAPTER 5: **CONCLUSION**

The objective of the current analysis was to estimate population health impacts and socio-economic costs associated with exposure to TRAP in Canada in 2015, where TRAP corresponded to Canadian on-road vehicle emissions.

Health Canada collaborated with ECCC to develop an emissions inventory, conduct national air quality modelling simulations and estimate the air pollution increment associated with Canadian on-road vehicle emissions (on-road vehicles in Canada's Air Pollutant Emissions Inventory). The emissions inventory listed releases from all source sectors and for key air pollutants including CO, NH_3, NO_X, $PM_{2.5}$, PM_{10}, SO_2 and VOCs. These included primary pollutants of concern (e.g., NO_X, $PM_{2.5}$) as well as those that contribute to the formation of secondary pollutants (e.g., NH_3, VOCs). Air pollutant concentrations were modelled for $PM_{2.5}$, O_3, NO_2 and SO_2 with the GEM-MACH chemical transport model, and they were used in Health Canada's AQBAT computer application to estimate the health burden and costs of TRAP at the CD level. The health analysis focused on three air pollutants: $PM_{2.5}$, O_3 and NO_2.

The 2015 Canadian emissions inventory showed that, on average, on-road vehicles contributed 28% to total CO emissions, 21% to NO_X, and 7% to VOCs in Canada. Contributions to total emissions were less than 1% for $PM_{2.5}$ and SO_2. The data for on-road vehicle emissions were generally consistent with previous Canadian analyses for on-road gasoline and diesel fleets, also for the year 2015.

Annual average ambient air pollutant concentrations and contributions from Canadian on-road vehicle emissions were modelled for $PM_{2.5}$, O_3 (daily 1-h maximum), NO_2 and SO_2. Summer average daily 1-h maximum concentrations were also modelled for O_3. The Canadian average reference concentrations (population-weighted) were 5.3 $\mu g/m^3$ for $PM_{2.5}$, 5.1 ppbv for NO_2, 40.0 ppbv for summer O_3, 36.4 ppbv for annual O_3, and 0.7 ppbv for SO_2. The national average population-weighted contributions from Canadian on-road vehicle emissions were:

- 0.37 $\mu g/m^3$, or 7.0% of annual average $PM_{2.5}$;
- 1.95 ppbv, or 38% of annual average NO_2;
- 0.93 ppbv, or 2.3% of summer average daily 1-h maximum O_3;
- 0.10 ppbv, or 0.3% of annual average daily 1-h maximum O_3; and
- 0.01 ppbv, or 1.5% of annual average SO_2.

Higher reference concentrations and contributions from Canadian on-road vehicle emissions were generally modelled for populous provinces such as British Columbia, Alberta, Ontario and Quebec, as well as for larger urban centres including the CDs corresponding to Vancouver, Calgary, Edmonton, Toronto and Montreal. These results were consistent with previous Canadian analyses of air pollution for on-road gasoline and diesel fleets.

The health burden analysis indicated that TRAP from Canadian sources was associated with 1,200 premature deaths in Canada in 2015. It was estimated that $PM_{2.5}$ contributed to 800 premature deaths, NO_2 to 340 premature deaths, and O_3 to 85 premature deaths. Non-fatal health outcomes included 2.7 million acute respiratory symptom days, 1.1 million restricted activity days and 210,000 asthma symptom days per year. The total annual monetary value of the health burden was estimated at $9.5 billion (CAD 2015), with $9 billion being associated with premature deaths. Greater health impacts were estimated in more populous provinces and CDs. For example, 500 premature deaths were estimated in Ontario, 410 in Quebec, 170 in British Columbia and 82 in Alberta. At the CD level, 170 premature deaths were estimated in Toronto, 150 in Montreal, 110 in Vancouver and 43 in Peel (Ontario). Additional modelling showed that the light-duty fleet contributed to approximately 37% of premature deaths, while the heavy-duty fleet contributed to approximately 63% of premature deaths.

While efforts were made in the current health assessment to use the best available air quality and health modelling tools and data for Canada, there were limitations and uncertainties. Uncertainty in modelling emissions, ambient air pollution concentrations, and health impacts originated from various sources, including: the availability and quality of Canadian data on vehicle fleets and emission factors; algorithms representing atmospheric transport and transformation; and health data to support the selection and use of relevant CRFs for the Canadian population. For example, ambient air pollutant concentrations were only available for $PM_{2.5}$, O_3, NO_2 and SO_2, and health impacts estimates were included for a limited number of adverse health outcomes associated with exposure to those pollutants. Moreover, the current analysis provided a regional evaluation of health burden, rather than an assessment of local risks for communities in direct proximity to sources of TRAP, such as high-traffic roadways. CDs represent a reasonable resolution for a national assessment, but do not reflect local or microenvironment conditions where contributions from TRAP to ambient concentrations can be higher and may even represent the main driver of air pollution. Overall, it was expected that uncertainties in the current assessment likely underestimated health impacts. The values herein possibly represent a lower range estimate.

The air pollution and health burden estimates are based on a modelling framework designed for the analysis of source sectors, including on-road vehicles. The modelled predictions for the year 2015 used specific and validated tools, data, and assumptions. They can differ from other analyses by the Government of Canada or health research organizations. The interpretation of the results accounts for methodological considerations and limitations, aiming to contextualize the current findings within the broader air pollution health literature.

The current analysis contributes to our understanding of the health risks associated with exposure to TRAP in Canada. While providing an update to previous Health Canada publications, it also complements recent and ongoing activities on TRAP at Health Canada. These include risk assessments characterizing the relationship between exposure to TRAP and the risk of adverse health outcomes using a weight of evidence approach (e.g., Health Canada 2020). Findings from risk assessment can provide the evidence needed to incorporate additional health outcomes into health impact analyses and improve future evaluations. Health Canada is also conducting an assessment of exposure to TRAP in Canada based on a literature review and an analysis of population proximity to roadways. Together, the Health Canada assessments of health risks, population health burden, and exposures associated with TRAP in Canada are intended to provide a comprehensive national evaluation.

In addition, Health Canada is currently investigating the air quality and health impacts of several source sectors across Canada based on the same modelling framework, tools and datasets used here. The consistency across simulations will provide an opportunity for comparisons of national and regional air pollution health impacts between sectors. It is intended that this information on population health burden will inform provincial, territorial and regional stakeholders, such as air zone managers,[32] and support further development of efficient and effective air quality management strategies.

[32] Canadian Council of Ministers of the Environment, Air Quality Management System, www.ccme.ca/en/resources/air/index.html

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APPENDICES

APPENDIX A: GEM-MACH AND AQBAT METHODOLOGY–ADDITIONAL INFORMATION

The Air Quality Policy-Issue Response Section of Environment and Climate Change Canada (ECCC) has developed a comprehensive air quality policy modelling platform. The platform allows for sound, traceable, reproducible, and standardized air quality simulations. It includes three main modelling components that are completed systematically:

- Emissions modelling
- Air quality modelling using the Global Environmental Multiscale–Modelling Air Quality and Chemistry model (GEM-MACH)
- Post-processing of GEM-MACH results.

The platform is used to support the development of regulatory impact analysis statements, conduct cost-benefit analyses, and to evaluate the air quality impacts of activity sectors, including industrial sources, transportation, residential emissions, and energy generation. In the current report, the platform was used to characterize the air quality and health impacts of on-road vehicle emissions in Canada.

A.1. Emissions modelling

The emissions modelling component involves the development of air pollutant emissions inventories to be used as input to the GEM-MACH model. Emission estimates need to be chemically speciated, spatially distributed, and temporally resolved. For GEM-MACH analyses, a set of hourly emission estimates, over an entire calendar year, are generated. Both anthropogenic and biogenic emissions are considered. However, wildfire emissions are excluded.

Biogenic emissions

Biogenic emissions are simulated with the US EPA Biogenic Emission Inventory System (BEIS v3.09) model[33] with modifications to the Biogenic Emissions Landuse Database (BELD3)[34] using maps from Canadian forest surveys. Biogenic emissions are separated into 15 types of land uses: crops, mixed farming; deciduous broadleaf; deciduous needle leaf; desert; dwarf trees and shrubs; evergreen broadleaf; evergreen needle leaf; grassland; ice; inland water; mixed forest; ocean; tundra; urban; and wet land with plants. The biogenic emissions are processed separately from the anthropogenic sources as the data are rarely modified.

[33] Biogenic Emission Inventory System (BEIS); www.epa.gov/air-emissions-modeling/biogenic-emission-inventory-system-beis

[34] Biogenic Emissions Landuse Database, Version 3 (BELD3); www.epa.gov/air-emissions-modeling/biogenic-emissions-landuse-database-version-3-beld3

Anthropogenic emissions

For the current analysis, the 2015 Air Pollutant Emissions Inventory (APEI) was used. This inventory was compiled by the Pollutant Inventories and Reporting Division of ECCC using both top-down and bottom-up approaches. Point source emissions were compiled using a bottom-up approach based on the facility level emissions reported to the National Pollutant Release Inventory (NPRI). Area and mobile source emissions were mainly compiled using a top-down approach based on source-specific activity data and emission factors. The inventory database was further processed by ECCC's Air Quality Policy-Issue Response Section using queries in ACCESS to generate inventory files in a format (FF10 format) compatible with the Sparse Matrix Operator Kernel Emissions modelling system (SMOKE v3.7). These files are identified as the 2015 Canadian reference case.

SMOKE is an emissions processing system that uses spatial surrogates and temporal allocation profiles to disaggregate emissions temporally and spatially. SMOKE creates gridded, chemically speciated, hourly emissions that can then be used as input for the GEM-MACH model.[35] SMOKE output data are grouped into area source (e.g., mobile and dust emissions) and point source (e.g., industrial stack emissions) emission files.[36] For example, point sources include facilities with a stack height greater than 15 metres for Canadian sources and greater than 30 metres for American and Mexican sources. Point source emissions are allocated on the basis of their geographic coordinates (latitude and longitude), and the system accounts for stack height and diameter, and ejection velocity and temperature. This information allows GEM-MACH to calculate the plume rise and extent. SMOKE also transforms model pollutant species, such as CO, NO_X, VOC, PM_{10} and SO_2, into chemical species used by the Acid Deposition and Oxidant Model (ADOM-II) chemical mechanism in GEM-MACH. The speciated pollutants include organics, PM species, and toxics. Temporal surrogates for Canada, the United States, and northern Mexico typically follow the default SMOKE profiles. Some adjustments were made to the Canadian temporal surrogates, such as for residential wood combustion and road dust.

The SMOKE data are then further processed by KornShell and Tool Command Language scripts to comply with the in-house binary format supported by GEM-MACH. To ensure reproducibility, traceability and documentation, all SMOKE runs are versioned in Git, a distributed version-control system, and GitLab, a web-based DevOps lifecycle tool (i.e., a set of practices combining software development and information-technology operations). Different quality assurance and control analyses are performed during modelling of anthropogenic emissions to verify that the GEM-MACH emissions reflect the values in the inventory and that the emissions are correctly distributed. These analyses include statistics and maps showing the spatial allocation of emissions.

A.2. GEM-MACH modelling

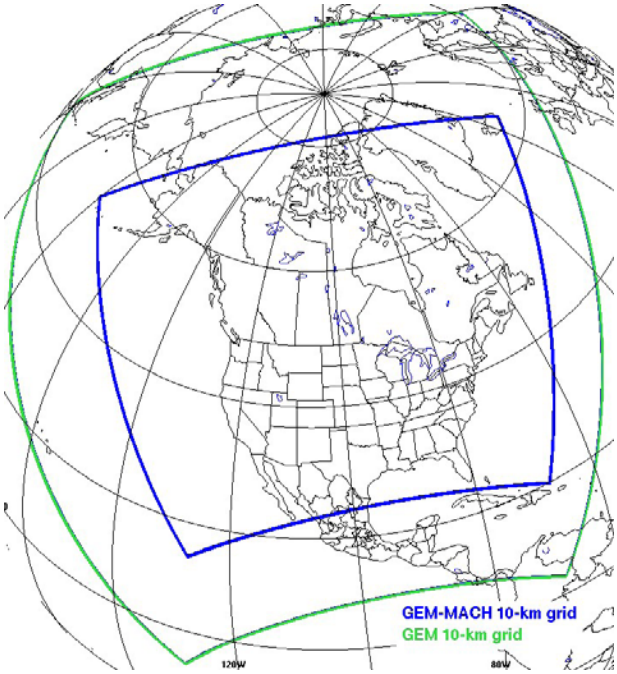
GEM-MACH is an on-line chemical transport model. It is embedded within ECCC's Global Environmental Multi-scale (GEM) weather forecast model. Figure A1, Appendix A, shows both the GEM domain (1108 × 1082 grid cells) and the GEM-MACH domain (768 × 638 grid cells). GEM-MACH is a one-way coupling model: meteorology influences chemistry but chemistry does not affect

[35] Community Modeling and Analysis system (CMAS), SMOKE; www.cmascenter.org/help/documentation.cfm

[36] Further processing of the SMOKE output was conducted to redistribute some point sources. For example, agriculture, airport landing and take-off, and industrial sources with small or no stack emissions, which are considered point source emissions in SMOKE, were defined as area sources in post-processing.

meteorology (Gong et al. 2015). GEM provides the initial atmospheric conditions needed by GEM-MACH including: temperature, precipitation, cloud cover, wind speed and direction, and humidity. Meteorological lateral boundary conditions come from the Regional Deterministic Prediction System at 10-km resolution.

FIGURE A1: GEM (in green) and GEM-MACH (in blue) 10-km grid domains used for air quality modelling



The following air quality processes are represented in the GEM-MACH model (Anselmo et al. 2010 and references therein):

- Gas-phase chemistry mechanism using ADOM-II
- Aqueous-phase chemistry mechanism using ADOM
- Heterogeneous chemistry mechanism using the heterogeneous partitioning code, HETV, based on the algorithms of ISORROPIA
- Aerosol dynamics through sedimentation, nucleation, condensation, coagulation, swelling, activation, sea-salt emissions, and inorganic gas-particle partitioning
- Secondary organic aerosol (SOA) formation using instantaneous SOA yield; with SOA comprising five groups of VOC species
- Dry deposition for gas species
- Dry deposition scheme dependent on particle size and density, land cover and relevant meteorological variables. It includes deposition via turbulent transfer, Brownian diffusion, impaction, interception, gravitational settling and particle rebound, as well as particle growth under humid conditions.
- Wet deposition via in-cloud and below-cloud scavenging of soluble gases and particles (size-dependent)

Aerosols (i.e., particulate matter) in GEM-MACH consist of the following chemical components: sulphate (SO_4), nitrate (NO_3), ammonium (NH_4), elemental carbon (EC), primary organic aerosol (POA), SOA, crustal material (CM), sea salt, and particle-bound water. EC, POA, CM and sea salt correspond to primary PM components (Gong et al. 2015). Aerosol particles are distributed by size in two bins: particles with an aerodynamic diameter smaller than 2.5 μm ($PM_{2.5}$) and particles with an aerodynamic diameter between 2.5 and 10 μm ($PM_{2.5–10}$). It is assumed that the particles are internally mixed; that is, the particle components are represented by a homogeneous material that reflects the chemical and physical average of all the contributing components (Lesins et al. 2002). Inter-bin condensational and evaporative transfers, as well as dry deposition velocities, are calculated by subdividing the two bins into sub-bins to better account for size dependencies (Anselmo et al. 2010).

GEM-MACH includes a physics and a chemistry processor. The physics processor is the meteorological component providing different parameters to the chemistry processor, such as cloud water mixing ratio (liquid and solid), precipitation production rate (auto-conversion and coalescence), precipitation evaporation rate, and precipitation fluxes (liquid and solid). The cloud microphysics resides in the physics processor. The chemistry processor includes nucleation scavenging (aerosol activation), aqueous-phase chemistry (mass transfer between gaseous and aqueous-phase, and aqueous-phase oxidation), and wet removal (including cloud-to-rain conversion and below-cloud scavenging) (Gong et al. 2015). The GEM version for the current simulations was x/4.8-u1.rc5., with version 4.8 for the dynamic components and version 5.8 for the physics components.

A.3. AQBAT–health endpoints and valuation

Health endpoints related to acute or chronic exposure, the associated concentration–response functions (CRFs), and the applicable population group(s) (e.g., age-specific groups) are predefined within the Air Quality Benefits Assessment Tool (AQBAT) and represent values drawn from the peer-reviewed health science literature and endorsed by Health Canada. The pollutants and their associated health effects considered in this analysis are listed in Table A1.

Each health endpoint was assigned a monetary value, expressed in Canadian dollars and temporally adjusted from the source years of the underlying studies based on the consumer price index. The current analysis used the currency year 2015 (CAD 2015). The endpoint values have inherent uncertainties, which are captured by a distribution of possible values with corresponding parameters (i.e., valuation estimates are entered as a distribution in AQBAT). Table 2 lists the valuation estimates used in the model and references to the studies from which they are derived.

TABLE A1: CRFs associated with NO_2, O_3 and $PM_{2.5}$ in the AQBAT–Averaging periods and associated health endpoints

Pollutant[a]	Averaging period	Health endpoint
NO_2	24 h	Acute exposure mortality[b,c]
O_3	1 h maximum	Acute exposure mortality[b]
O_3 summer (May–September)	1 h maximum	Acute respiratory symptom days
		Asthma symptom days
		Chronic exposure respiratory mortality
		Minor restricted activity days
		Respiratory emergency room visits
		Respiratory hospital admissions
$PM_{2.5}$	24 h	Acute respiratory symptom days
		Adult chronic bronchitis cases
		Asthma symptom days
		Cardiac emergency room visits
		Cardiac hospital admissions
		Child acute bronchitis episodes
		Chronic exposure mortality
		Respiratory emergency room visits
		Respiratory hospital admissions
		Restricted activity days

NO_2: nitrogen dioxide; O_3: ozone; $PM_{2.5}$: particulate matter with a diameter of 2.5 µm or less

[a] Unless otherwise specified, CRFs reflect an exposure to the pollutant at any time during the year;

[b] The CRF between acute exposure mortality and gaseous pollutants is from a model including CO, NO_2, O_3 and sulphur dioxide and may not precisely reflect the true attribution of risk to individual pollutants;

[c] It is recognized that the CRF for acute exposure mortality associated with NO_2 exposure may reflect a causal relationship with NO_2 or NO_2 may be acting as a surrogate for a specific component of the air pollution mixture, such as vehicle exhaust emissions.

TABLE A2: Economic value of health endpoints used in the AQBAT

Endpoint [reference]	Currency year	Source type	Form[a]	Parameter 1 (prob.)	Parameter 2 (prob.)	Parameter 3 (prob.)
Mortality [Chestnut and De Civita 2009]	2007	WTP/WR	Discrete	$3,500,000 (25%)	$6,500,000 (50%)	$9,500,000 (25%)
Acute respiratory symptom days [Stieb et al. 2002]	1997	WTP	Normal	$13	$7	–
Adult chronic bronchitis cases [Krupnick & Cropper 1992; Viscusi et al. 1991]	1996	WTP	Discrete	$175,000 (33%)	$266,000 (34%)	$465,000 (33%)
Asthma symptom days [Stieb et al. 2002]	1997	WTP	Triangular	$7	$28	$120
Cardiac emergency room visits[b] [Stieb et al. 2002]	1997	WTP	Normal	$4,400	$590	–
Child acute bronchitis episodes [Krupnick and Cropper 1989]	1996	WTP	Discrete	$150 (33%)	$310 (34%)	$460 (33%)
Elderly cardiac hospital admissions [Stieb et al. 2002]	1997	WTP	Normal	$5,200	$610	–
Minor restricted activity days [Stieb et al. 2002]	1997	WTP	Normal	$22	$9	–
Respiratory emergency room visits[b] [Stieb et al. 2002]	1997	WTP	Normal	$2,000	$210	–
Restricted activity days [Stieb et al. 2002]	1997	WTP	Normal	$48	$18	–

Adapted from Judek et al. (2019)

COPD: chronic obstructive pulmonary disease; prob.: probability of value being selected in analysis; WR: wage risk; WTP: willingness to pay

[a] For valuations represented by discrete values, parameters 1, 2 and 3 represent low, medium and high estimates, respectively. For valuations represented by normal distributions, parameters 1 and 2 represent the mean and standard error of the estimates, respectively. For valuations represented by triangular distributions, parameters 1, 2 and 3 represent minimum, most likely and maximum values, respectively;

[b] Respiratory and cardiac emergency room visits include the costs of subsequent hospital admissions based on the proportion of emergency room visits that result in admission to hospital. Hospital admissions are assigned a value of zero to avoid double counting of costs.

A.4. References

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APPENDIX B: AIR EMISSIONS–ADDITIONAL INFORMATION, TABLES AND FIGURES

Table B1 details particulate and VOC emissions from Canadian on-road vehicles. Particulate emissions are divided into three components: exhaust, brake and tire. Brake and tire emissions are also referred to as tire wear and brake lining (TWBL) emissions. Evaporative emissions reflect the VOCs in fuel that evaporate through the fuel and engine system during operation of the vehicle or when it is parked or stored. TWBL and evaporative emissions make up non-combustion or non-exhaust emissions.

The 2015 emissions inventory data indicate that particulates are released mainly via exhaust gases, followed by brake wear emissions. The data also suggest that brake wear and tire wear particles are generally larger in size than exhaust particles, as indicated by the difference between PM_{10} and $PM_{2.5}$ mass-based emission releases.[37] The exhaust values are comparable for $PM_{2.5}$ and PM_{10}, indicating that most particulate emissions are within the $PM_{2.5}$ size range ($PM_{2.5}$ emissions are included in the PM_{10} values). By contrast, brake and tire wear values differ considerably. It is also observed that the HDV8 sub-class (e.g., freight trucks) is the highest emitter of exhaust PM among on-road vehicle classes, despite the approximate ratio of 40 light-duty vehicles or trucks registered for every HDV8 vehicle in Canada.[38] HDV8 vehicles are much larger and heavier than passenger vehicles. Further, HDV8 vehicles are typically equipped with diesel engines that emit more particles than gasoline engines. Spark-ignition engines generally emit less particulate matter than compression ignition engines. Non-exhaust emissions from the smaller segments of the on-road fleet (i.e., LDT and LDV) are greater than HDV8 brake and tire emissions, owing to the total number of vehicles on the road. LDVs and LDTs also emit more non-exhaust PM emissions than those originating from exhaust processes.

[37] $PM_{2.5}$ emissions are included in PM_{10}. The difference between both values is equivalent to the $PM_{2.5\text{-}10}$ or coarse PM fraction.

[38] Statistics Canada, annual. Vehicle registrations, by type of vehicle. Table 23-10-0067-01. www150.statcan.gc.ca/t1/tbl1/en/tv.action?pid=2310006701

TABLE B1: Canadian on-road vehicle emissions in 2015, in tonnes–Detailed PM and VOC data

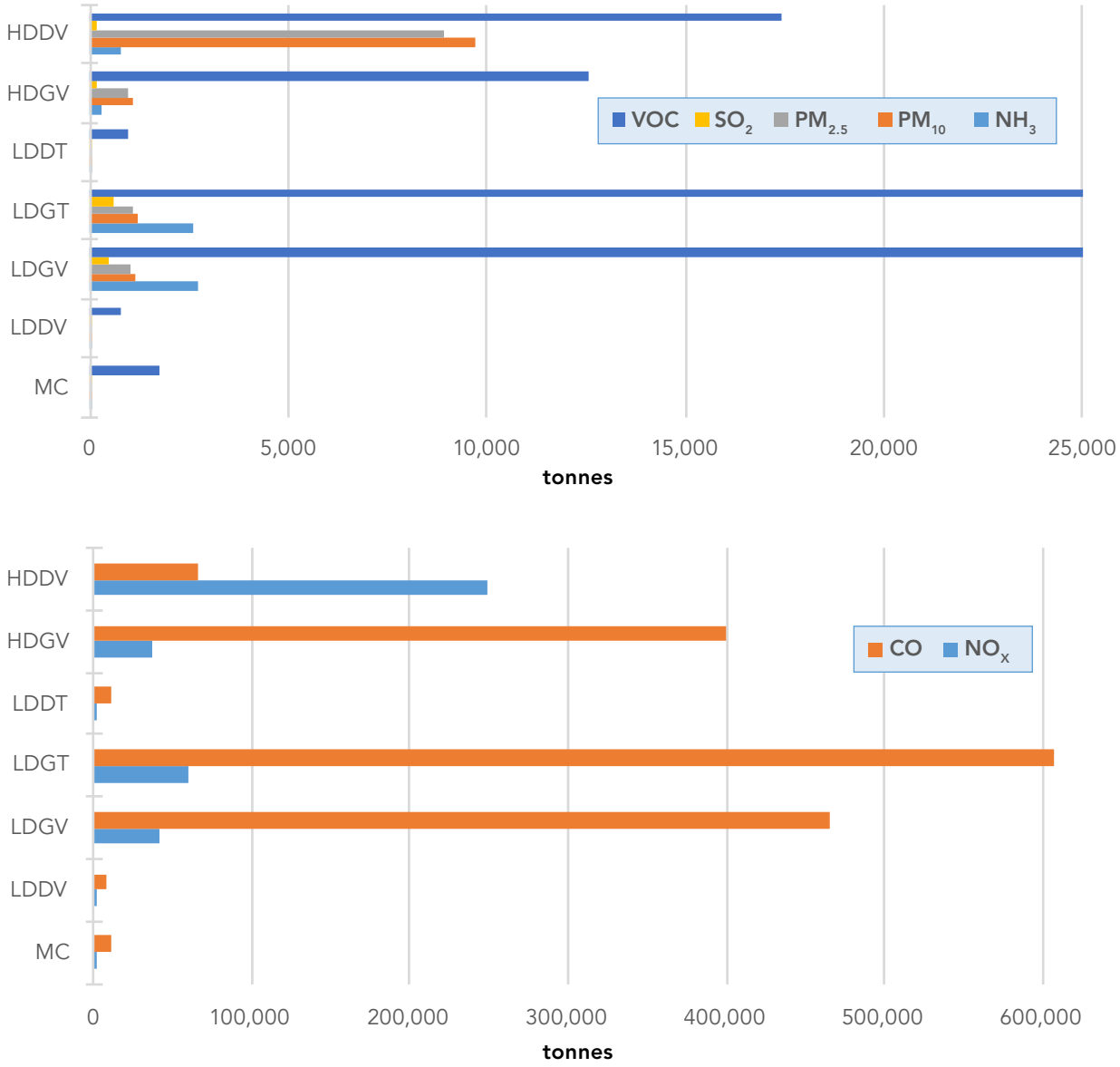
Pollutant	PM_{10}			$PM_{2.5}$			VOC	
Type	Exhaust	Brake	Tire	Exhaust	Brake	Tire	Exhaust	Evaporative
On-Road	**13,266**	**8,311**	**2,390**	**12,080**	**1,039**	**358**	**107,974**	**17,793**
HDV2B-3	3,544	418	155	3,228	52	23	14,666	780
HDV4-5	556	107	26	508	13	4	2,064	109
HDV6-7	1,127	315	57	1,033	39	9	3,506	355
HDV8	5,594	1,965	460	5,146	246	69	8,507	37
LDT	1,248	3,372	832	1,104	421	125	43,053	6,216
LDV	1,177	2,132	857	1,042	266	129	35,471	9,269
MC	21	1	3	18	0	0	707	1,028
Total	**23,966**			**13,477**			**125,767**	

HDV: Heavy-duty vehicle class; LDT: light-duty truck; LDV: light-duty vehicle; MC: motorcycle

Totals may not correspond due to rounding.

The values in Table B1 also show that VOC emissions in exhaust are approximately 6 times greater than evaporative emissions. The sub-class emissions reveal that the exhaust to evaporative ratio varies considerably among vehicle classes; they reflect the primary fuel type for each class. For example, the larger HDVs, especially class 8, are typically powered by compression engines and diesel fuel, which is less volatile than gasoline fuel. The exhaust VOC emissions for HDV class 8 are 230 times higher than evaporative emissions, whereas they are 10 times higher than evaporative emissions for HDV classes 6 and 7, which include more gasoline-fuelled vehicles. In further contrast, light-duty vehicles, which are mainly gasoline powered in Canada, emit only 5 times as much VOCs via exhaust than through evaporation. Motorcycles even emit more VOC through evaporative processes than via exhaust (approximately 50% more).

FIGURE B1: Canadian on-road vehicle emissions by vehicle sub-class in 2015, in tonnes
Top panel: NH_3, PM_{10}, $PM_{2.5}$, SO_2 and VOC. Bottom panel: CO and NO_X



HDDV: heavy-duty diesel vehicle; HDGV: heavy-duty gasoline vehicle; LDDT: light-duty diesel truck; LDDV: light-duty diesel vehicle; LDGT: light-duty gasoline truck; LDGV: light-duty gasoline vehicle; MC: motorcycle

TABLE B2: On-road vehicle emissions for the heavy-duty and light-duty fleets in 2015, in tonnes–Provincial, territorial and national values

Region and vehicle class[a,b]	CO	NH_3	NO_X	PM_{10}	$PM_{2.5}$	SO_2	VOC
Alberta	**292,000**	**1,067**	**90,034**	**4,963**	**3,045**	**186**	**23,238**
Heavy-duty	114,629	304	73,959	3,526	2,528	72	7,571
Light-duty	171,640	763	16,075	1,437	495	115	15,667
British Columbia	**253,743**	**816**	**59,643**	**3,058**	**1,878**	**152**	**21,574**
Heavy-duty	95,621	147	42,887	1,980	1,483	49	5,164
Light-duty	158,122	669	16,756	1,078	395	103	16,410
Manitoba	**78,088**	**284**	**16,706**	**1,059**	**620**	**45**	**6,948**
Heavy-duty	23,444	41	11,602	617	458	10	1,419
Light-duty	56,644	244	5,103	442	162	36	5,529
New Brunswick	**33,255**	**141**	**8,714**	**554**	**309**	**37**	**2,580**
Heavy-duty	7,390	23	6,325	348	247	8	592
Light-duty	25,864	118	2,389	206	62	29	1,988
Newfoundland and Labrador	**18,255**	**82**	**5,350**	**348**	**192**	**22**	**1,318**
Heavy-duty	4,240	16	4,197	220	156	5	382
Light-duty	14,016	66	1,153	128	36	18	936
Nova Scotia	**34,825**	**154**	**8,618**	**551**	**288**	**50**	**2,499**
Heavy-duty	7,548	24	6,232	317	223	11	574
Light-duty	27,276	129	2,386	233	65	39	1,925
Northwest Territories	**1,728**	**10**	**2,251**	**96**	**67**	**2**	**238**
Heavy-duty	1,056	8	2,195	91	64	1	171
Light-duty	672	2	57	5	2	0	66
Ontario	**435,918**	**2,198**	**95,316**	**6,766**	**3,297**	**575**	**34,773**
Heavy-duty	89,054	271	62,784	3,247	2,251	106	6,332
Light-duty	346,864	1,927	32,531	3,519	1,046	469	28,441
Prince Edward Island	**8,104**	**31**	**1,931**	**123**	**76**	**8**	**621**
Heavy-duty	1,717	3	1,339	76	60	1	125
Light-duty	6,387	27	591	47	15	7	495
Quebec	**272,365**	**1,307**	**68,458**	**4,632**	**2,531**	**301**	**20,598**
Heavy-duty	60,986	175	49,900	2,709	1,985	57	4,481
Light-duty	211,379	1,132	18,558	1,923	546	244	16,118
Saskatchewan	**141,543**	**389**	**31,375**	**1,797**	**1,162**	**80**	**11,312**
Heavy-duty	54,365	74	23,007	1,176	903	25	3,178
Light-duty	87,179	315	8,368	621	259	55	8,134
Yukon	**812**	**3**	**429**	**18**	**12**	**0**	**68**
Heavy-duty	404	1	397	15	11	0	34
Light-duty	408	2	32	3	1	0	34
Canada	**1,570,637**	**6,482**	**388,824**	**23,966**	**13,477**	**1,460**	**125,767**

Totals may not correspond due to rounding.

[a] Heavy-duty: heavy-duty vehicle classes 2–8; Light-duty: light-duty trucks, light-duty vehicles and motorcycles;

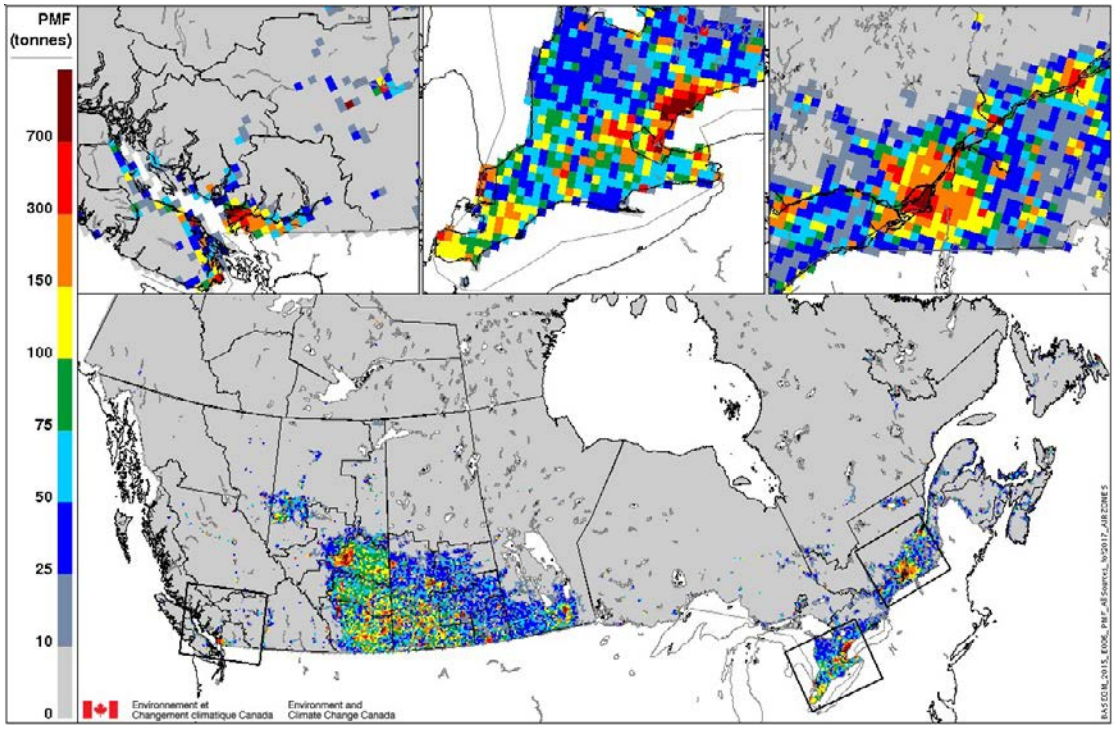
[b] No values available for Nunavut

Figures B2 and B3 show the distribution of $PM_{2.5}$ emissions for all sources and on-road vehicles, respectively. Emission levels generally coincide with the population distribution in Canada, with higher values in urban centres and surrounding areas. In addition, regions with extensive industrial activity are associated with higher $PM_{2.5}$ emissions, including the oil and gas production areas of northeastern British Columbia, Alberta and Saskatchewan, areas of mining activity in Saskatchewan, and in locations such as Timmins, Ontario and Val d'Or, Québec, and areas of metal smelting activity in the Saguenay–Lac-Saint-Jean region of Quebec. Agricultural activities are also associated with considerable $PM_{2.5}$ emissions in the Prairie provinces, Ontario, and Quebec. The on-road vehicle $PM_{2.5}$ emissions in Figure B3 indicate that emissions are mostly generated in urban centres and surrounding areas, where higher traffic activity is expected. Urban grid cells are associated with emission levels of 80 tonnes or more per year. Lower levels of emissions (less than 5 tonnes per year) are modelled across most grid cells.

The distribution of NO_X emissions (Figures B4 and B5) shares some similarities with that of $PM_{2.5}$. For all sources of NO_X emissions (Figure B4), higher values are projected in urban centres and in areas with considerable industrial activity, including regions associated with oil and gas production, mining and smelting. The emissions map for all sources also shows higher emissions along roadways and railway lines, notably in western Ontario and across the Prairies. Higher (500 tonnes or more per year) on-road vehicle NO_X emissions (Figure B5) are modelled in urban centres and surrounding regions. High-traffic areas, such as the Windsor–Québec corridor and the Edmonton–Calgary corridor, are also associated with higher NO_X emissions.

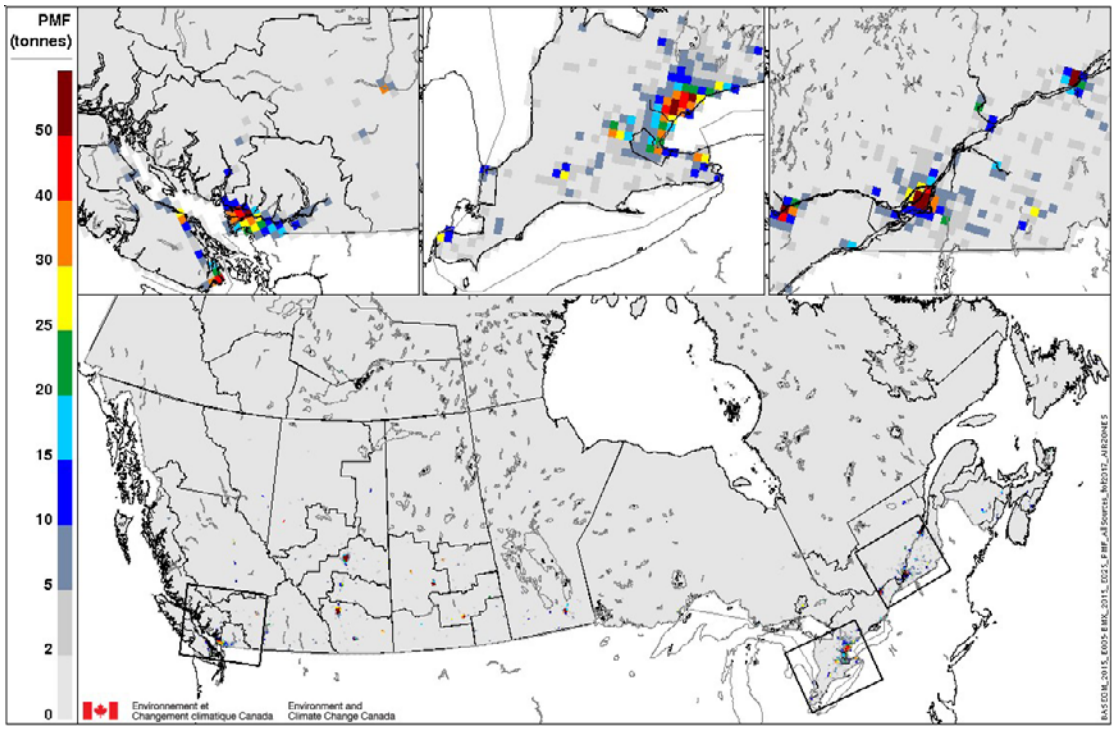
The maps for VOC emissions (Figure B6 and B7) show a comparable distribution and geographical extent to those of $PM_{2.5}$ and NO_X emissions, except in Saskatchewan. It appears that meaningful sources of VOC emissions are more limited (or unaccounted for) compared to other pollutants in that province. As noted earlier for $PM_{2.5}$ and NO_X, on-road vehicle VOC emission are higher in urban centres and surrounding grid cells, reaching 300 to 500 tonnes per year in urban grid cells.

FIGURE B2: $PM_{2.5}$ emissions (tonnes) in Canada in 2015–All sources



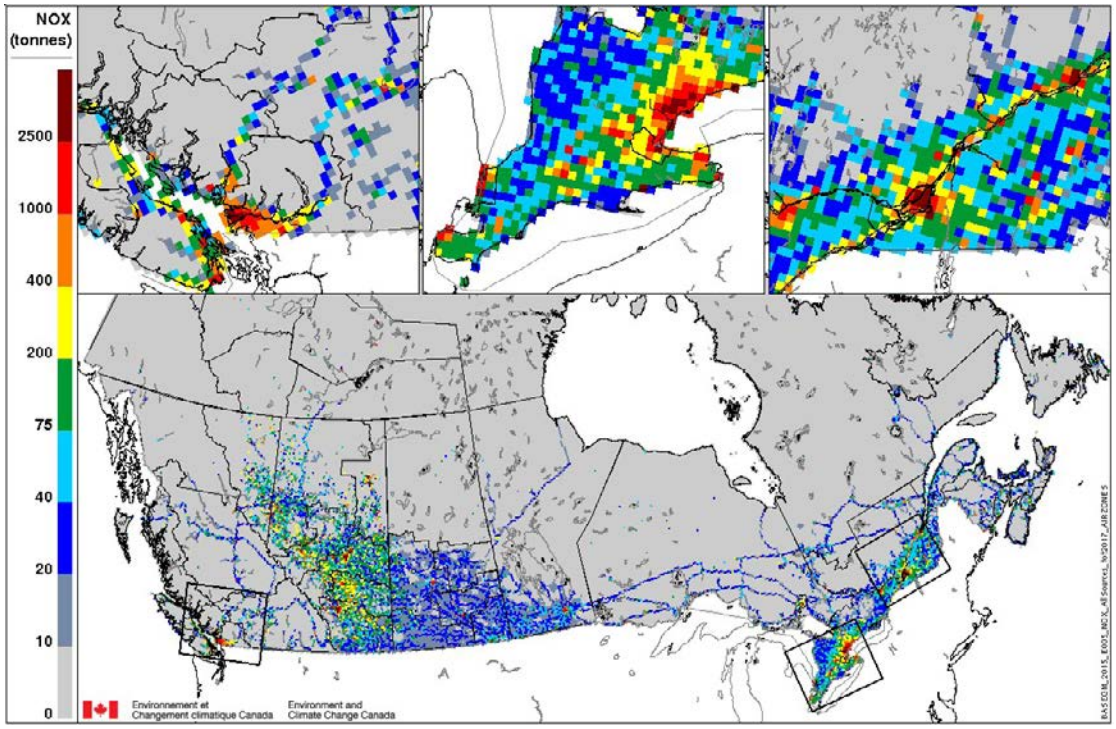
Notes: Insets for southern British Columbia, Ontario and Quebec

FIGURE B3: $PM_{2.5}$ emissions (tonnes) in Canada in 2015–On-road vehicles



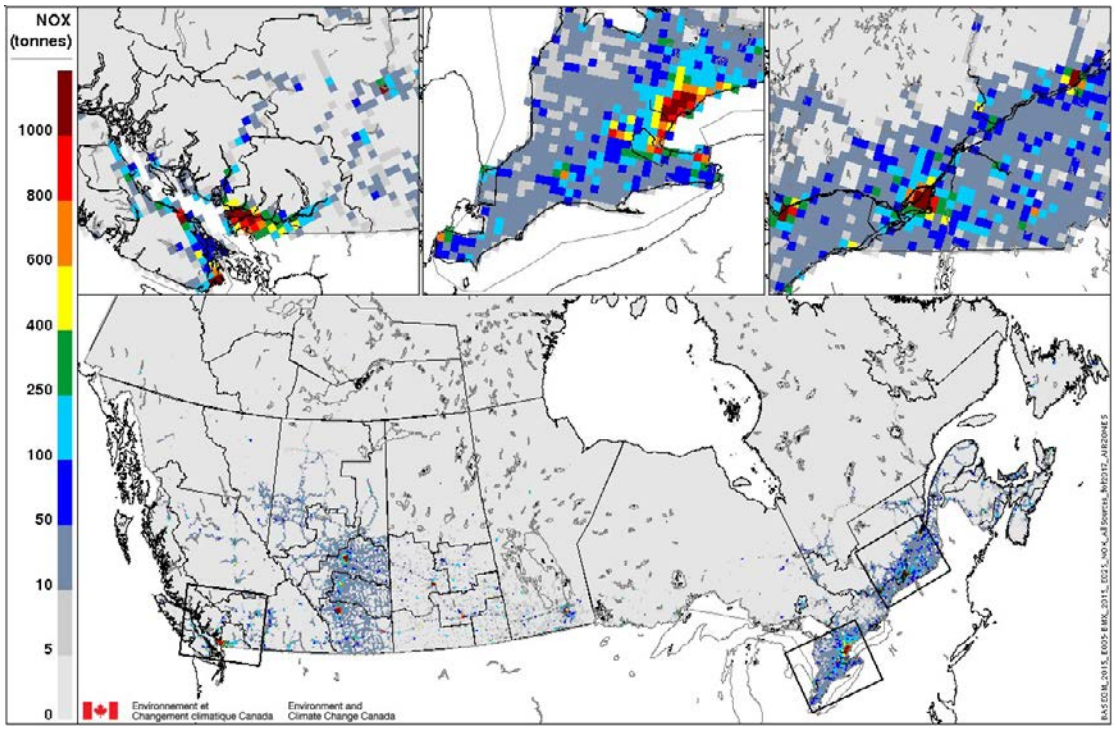
Notes: Insets for southern British Columbia, Ontario and Quebec

FIGURE B4: NO_x emissions (tonnes) in Canada in 2015–All sources



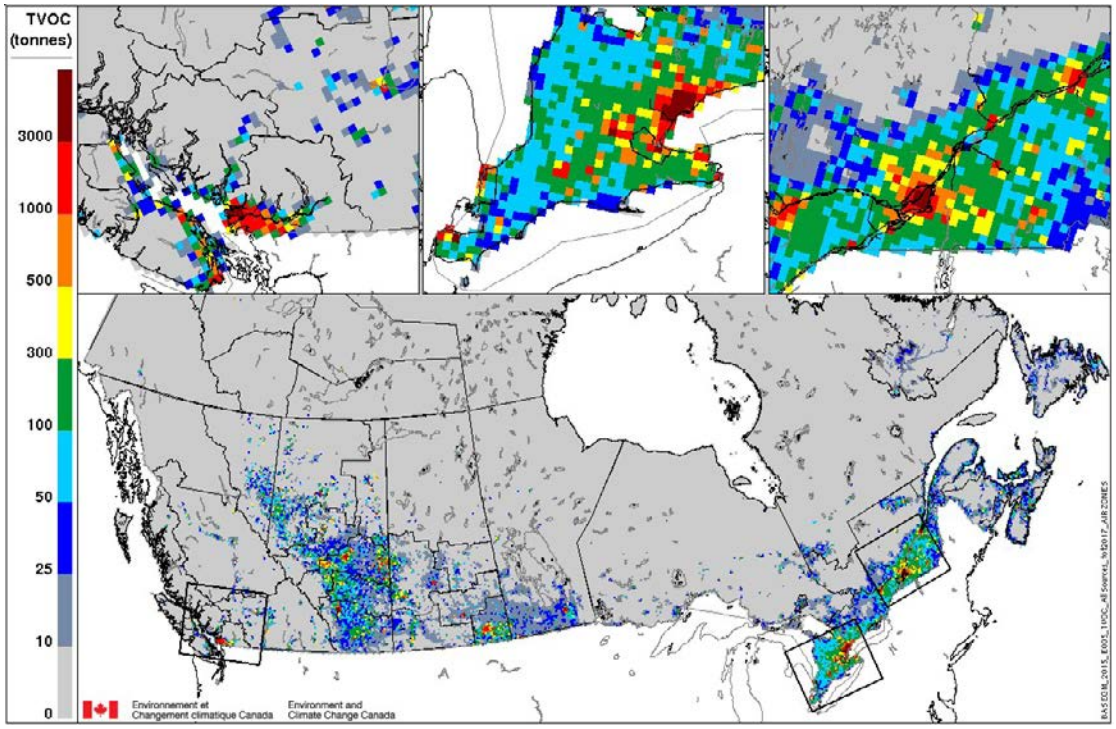
Notes: Insets for southern British Columbia, Ontario and Quebec

FIGURE B5: NO_x emissions (tonnes) in Canada in 2015–On-road vehicles



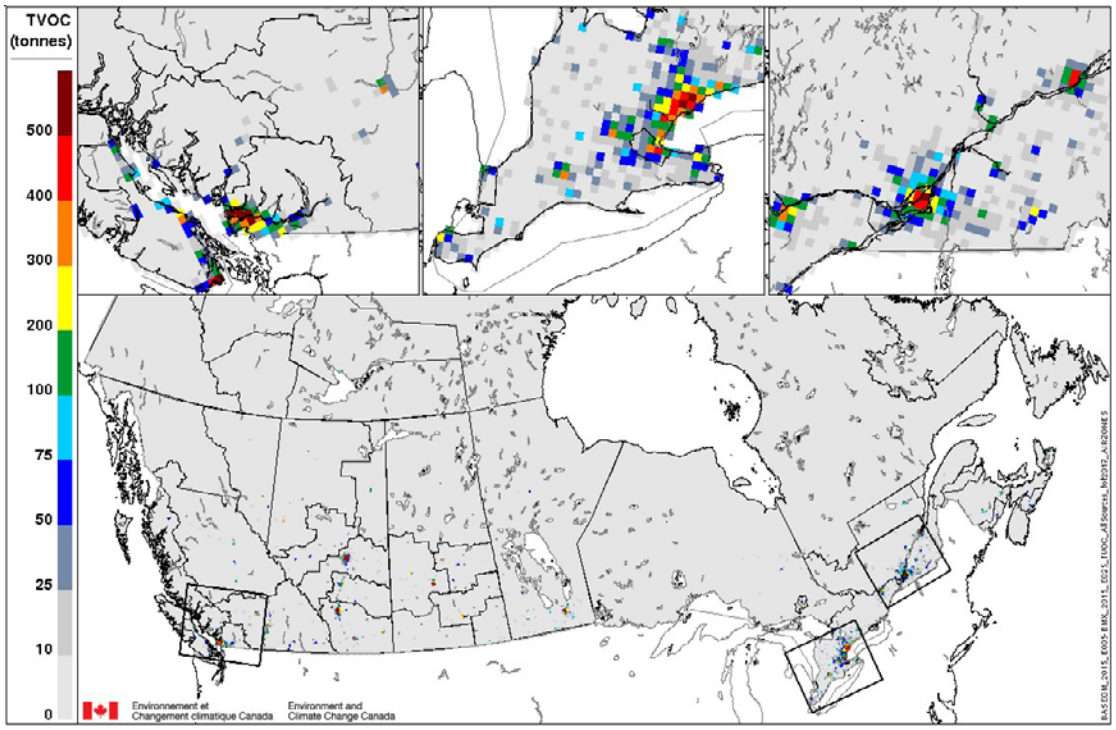
Notes: Insets for southern British Columbia, Ontario and Quebec

FIGURE B6: VOC emissions (tonnes) in Canada in 2015–All sources



Notes: Insets for southern British Columbia, Ontario and Quebec

FIGURE B7: VOC emissions (tonnes) in Canada in 2015–On-road vehicles



Notes: Insets for southern British Columbia, Ontario and Quebec

APPENDIX C: AMBIENT AIR CONCENTRATIONS–ADDITIONAL INFORMATION, TABLES AND FIGURES

This appendix includes air pollution maps showing the modelled ambient concentrations for the 2015 reference scenario. Maps indicating the relative contributions from Canadian on-road vehicle emissions (TRAP) are also included. The maps use the gridded results from the GEM-MACH model and have a resolution of 10 km. They do not include any population or area weighting. The insets are for southern British Columbia, Ontario and Quebec.

FIGURE C1: Annual average $PM_{2.5}$ concentrations (µg/m^3) in 2015–Reference scenario

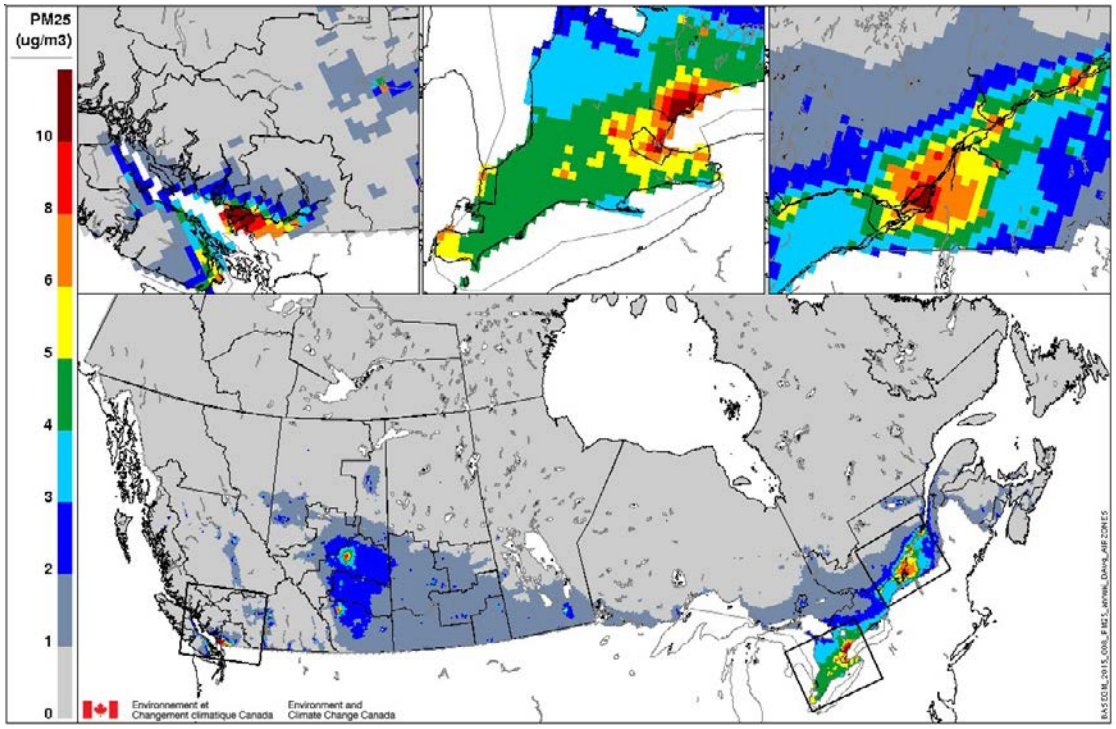


FIGURE C2: Relative contribution (%) to annual average $PM_{2.5}$ concentrations in 2015–TRAP

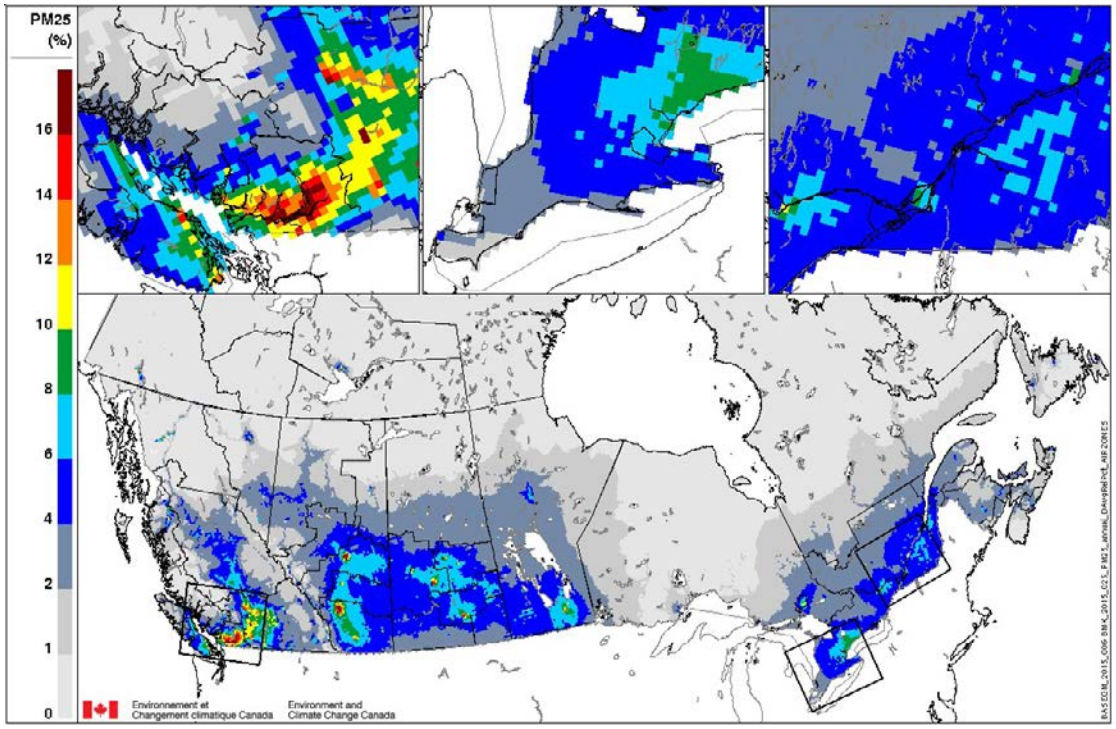


FIGURE C3: Annual average NO_2 concentrations (ppbv) in 2015–Reference scenario

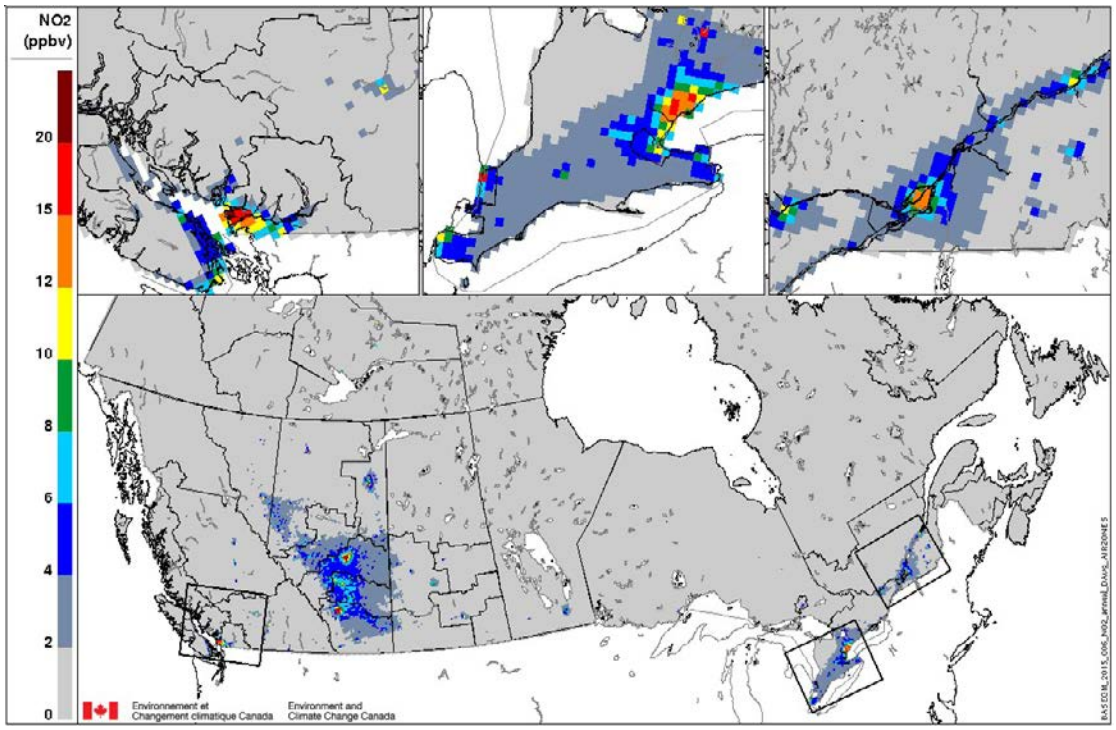


FIGURE C4: Relative contribution (%) to annual average NO_2 concentrations in 2015–TRAP

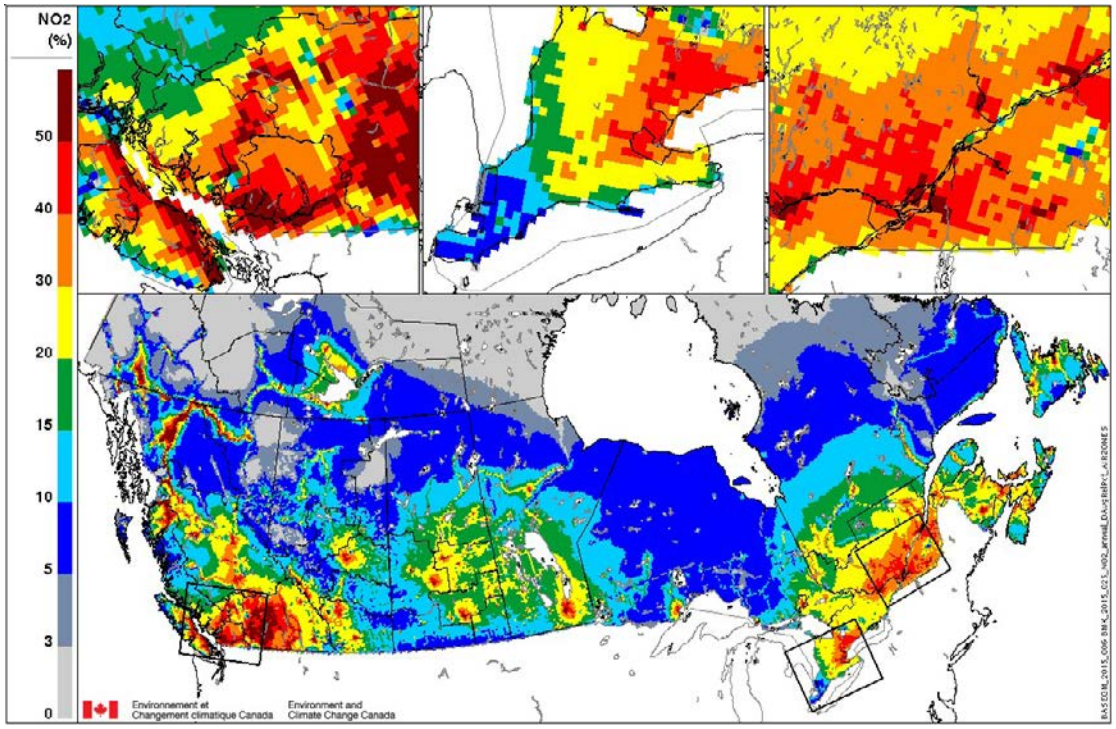


FIGURE C5: Summer average of daily 1-h maximum O_3 concentrations (ppbv) in 2015–Reference scenario

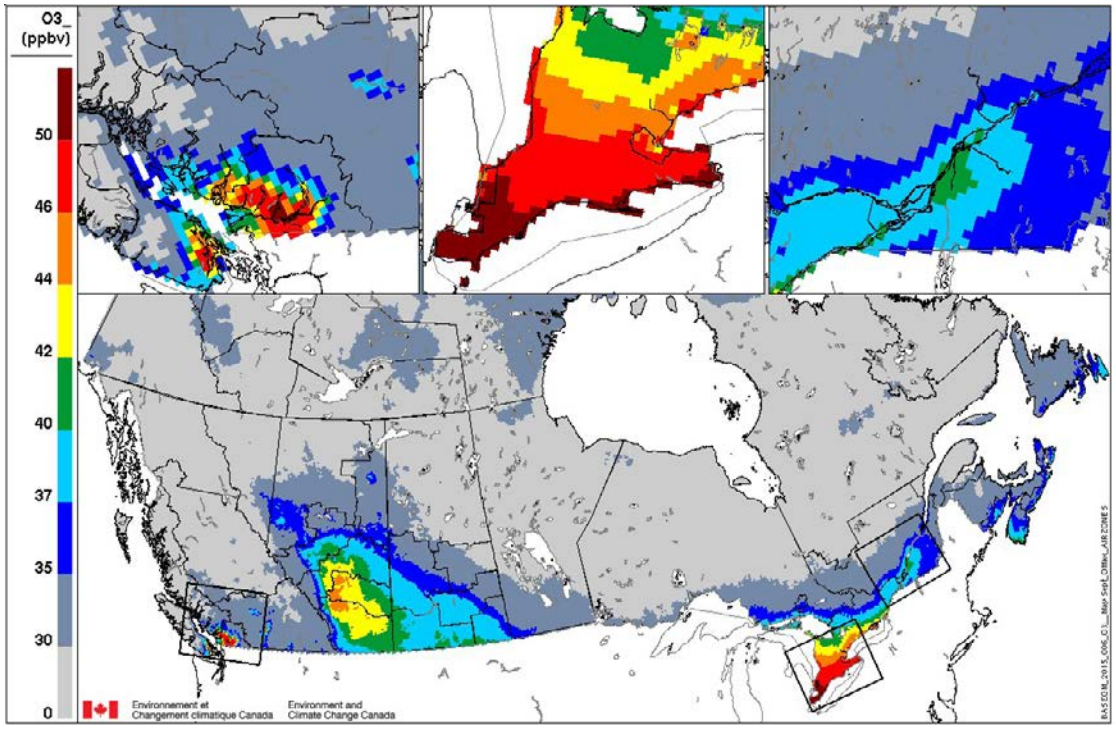


FIGURE C6: Relative contribution (%) to summer average daily 1-h maximum O_3 concentrations in 2015–TRAP

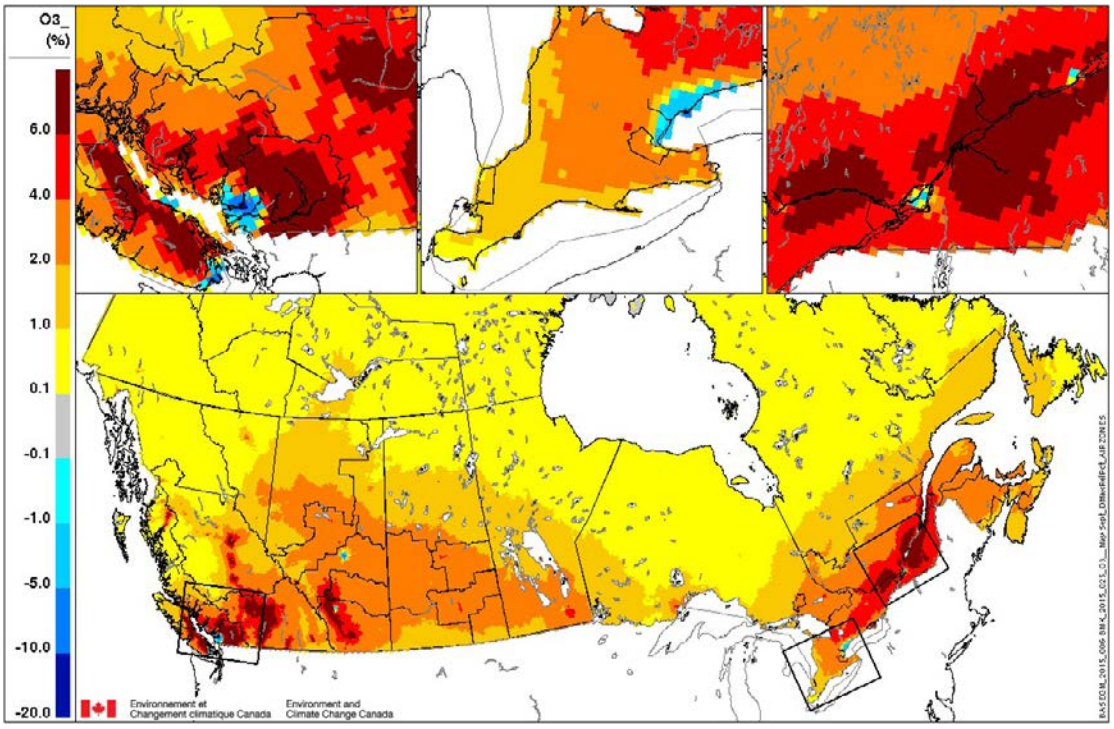


FIGURE C7: Annual average of daily 1-h maximum O_3 concentrations (ppbv) in 2015–Reference scenario

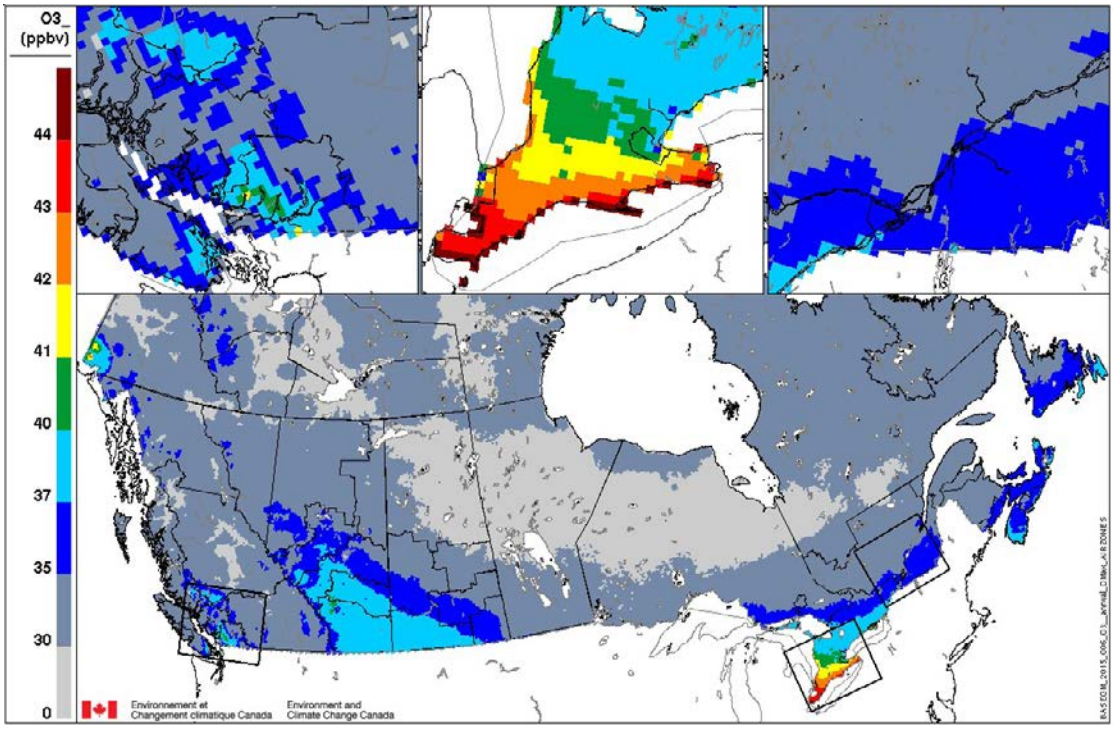
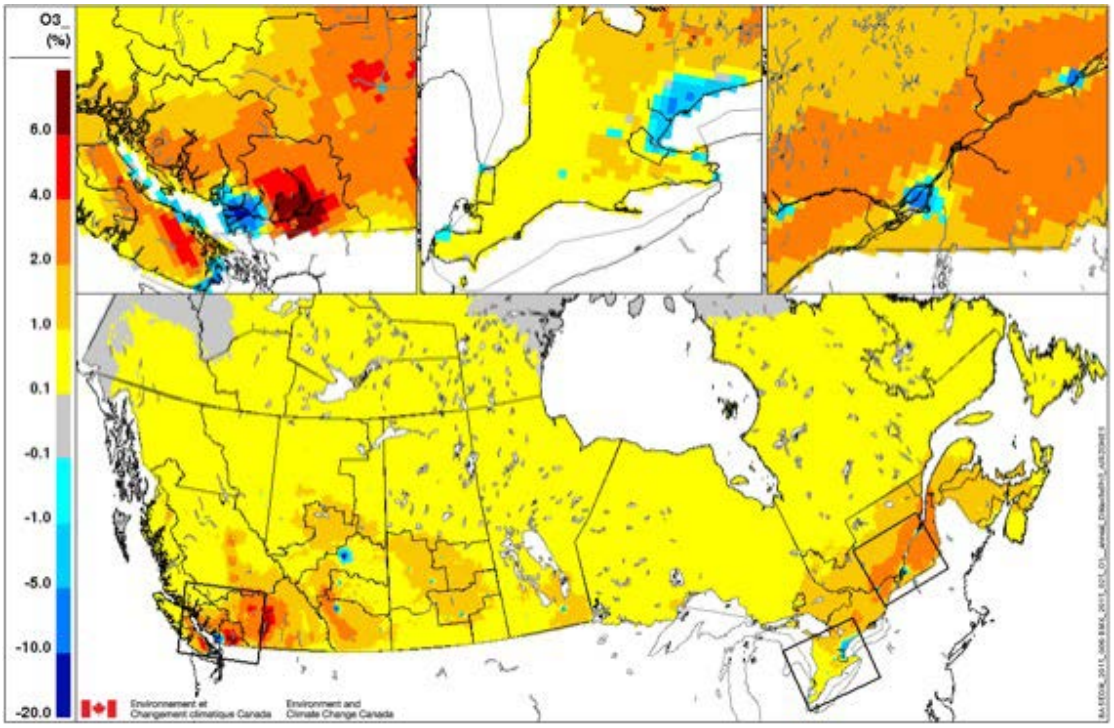


FIGURE C8: Relative contribution (%) to annual average daily 1-h maximum O_3 concentrations in 2015–TRAP



APPENDIX D: BASE CASE MODEL PERFORMANCE ANALYSIS

Comparisons between the GEM-MACH model results for the 2015 base case were compared to observations from the National Air Pollution Surveillance monitoring network for the year 2017. Monitoring data for the year 2017 were selected to correspond to the 2017 meteorology used for the simulations (see Section 2.2.1). The comparisons were performed with Environment and Climate Change Canada's Verification of Air Quality Models (VAQUM) tool. VAQUM can analyze several model results and perform statistical analyses. For the current analysis, mean bias error (MBE), Pearson correlation coefficient, and root mean square error (RMSE) were estimated for annual, summer, monthly, daily, and hourly comparisons for NO_2, O_3 and $PM_{2.5}$. The analyses were performed on three regions: Canada as a whole, eastern Canada, and western Canada.

In the current analysis, MBE refers to the difference between the mean of the modelled results and the observed concentrations. A positive value signifies an overestimation of the modelled results compared to the observations, while a negative bias indicates an underestimation. MBE values close to zero reflect a higher correlation between the modelled and observed concentrations.

The Pearson correlation coefficient measures the linear relation between pairs of variables. Correlation coefficients vary between -1 (perfect inverse relationship) and 1 (perfect positive relationship).

The RMSE compares the modelled and observed values by considering the square root of the average of the squared differences. The RMSE aggregates the magnitudes of the differences, also known as errors. The RMSE estimate is always positive. Lower RMSE values indicate a better fit between the model results and the observations.

Table 21 in the main report (section 4.3.2.5) shows annual statistics. The model overestimated annual NO_2 concentrations compared to the observed concentrations, whereas it underestimated annual O_3 and $PM_{2.5}$ concentrations. The lowest MBE value was estimated for $PM_{2.5}$ in eastern Canada. In terms of correlation coefficient, the results indicated a strong positive relationship of 0.7 for O_3 in eastern Canada and in Canada as a whole, as well as for NO_2 in eastern Canada. A moderate positive relationship of 0.6 was estimated for NO_2 concentrations in Canada as a whole and western Canada, and for O_3 concentrations in western Canada. A weak positive relationship was obtained for $PM_{2.5}$ concentrations in Canada as a whole (0.3) and in eastern Canada (0.4), and a very weak relationship in western Canada (0.2). The weak correlation for $PM_{2.5}$ was expected due to the absence of forest fire emissions in the 2015 base-case simulation.

Statistical analyses were performed for the summer months (May to September). They are presented in Table D1. The values in Table D1 are similar to the values found in Table 21. The MBE values for NO_2 are positive, indicating an overestimation of the modelled results compared to observations, whereas the model underestimated O_3 and $PM_{2.5}$ concentrations. Based on MBE results, NO_2 shows better agreement for all three regions studied. The best MBE was estimated for O_3 in eastern Canada (-1.2), while the worst value was reported or $PM_{2.5}$ in western Canada (-4.7). The latter is due to the omission of forest fire emissions in the 2015 base case.

TABLE D1: Summer performance evaluation statistics comparing the 2015 base case estimates in GEM-MACH and 2017 observations from NAPS

Region	MBE	Correlation[a]	RMSE
NO_2			
Canada	1.5	0.6	6.7
Eastern Canada	1.5	0.6	6.2
Western Canada	1.4	0.6	7.1
O_3			
Canada	-2.2	0.7	10.7
Eastern Canada	-1.2	0.7	10.1
Western Canada	-3.8	0.6	11.4
$PM_{2.5}$			
Canada	-3.1	0.2	11.2
Eastern Canada	-1.8	0.3	6.7
Western Canada	-4.7	0.2	14.9

MBE: Mean bias error; NAPS: National Air Pollution Surveillance program; RMSE: Root mean square error

[a] Pearson correlation coefficient

The Pearson correlation coefficient values in Table D1 were also similar to the values found in Table 21:

- O_3 values were comparable and suggested a strong positive correlation;
- NO_2 values for eastern Canada decreased by 0.1. and NO_2 values, overall, suggested a moderate positive relationship;
- $PM_{2.5}$ correlation values dropped for western Canada and Canada as a whole, in relation missing forest fire emissions.

The NO_2 RMSE values were lower in the summer compared to the annual estimates, suggesting less pronounced differences in summer period concentrations than across the entire year. The opposite was observed for O_3 and $PM_{2.5}$, with higher RMSE values during the summer.

Daily statistics comparing 2017 observations and the modelled 2015 base case concentrations for NO_2, O_3 and $PM_{2.5}$ were calculated only for NAPS stations located in metropolitan areas of Canada. Metropolitan areas considered herein corresponded to census metropolitan areas identified in the 2011 Census.[39] Figure D1 shows the relatively good agreement between observed and modelled daily average NO_2 concentrations in metropolitan regions across Canada. The model generally overestimated NO_2 concentrations, especially during the summer. Observed and modelled values were more similar during winter and fall.

Figure D2 shows the daily average O_3 concentrations for Canadian metropolitan regions. The model underestimated O_3 concentrations from January to July compared to observations. Better agreement was achieved from August to December.

Figure D3 shows the daily average $PM_{2.5}$ concentrations for stations in metropolitan areas. From January to April and from October to December, modelled values were greater than observations. An overestimation of residential wood combustion emissions was possibly responsible for the higher modelled ambient concentrations in the 2015 base case. Underestimation of the modelled concentrations during the summer months (May to September) was associated with the absence of forest fire emissions in the 2015 base case.

[39] Census Metropolitan Area and Census Agglomeration Cartographic Boundary Files–2011 Census; https://open.canada.ca/data/en/dataset/821ef476-d554-4bb4-bc32-bc916640fc9d

FIGURE D1: Modelled and observed daily average NO_2 concentrations for stations located in metropolitan areas across Canada

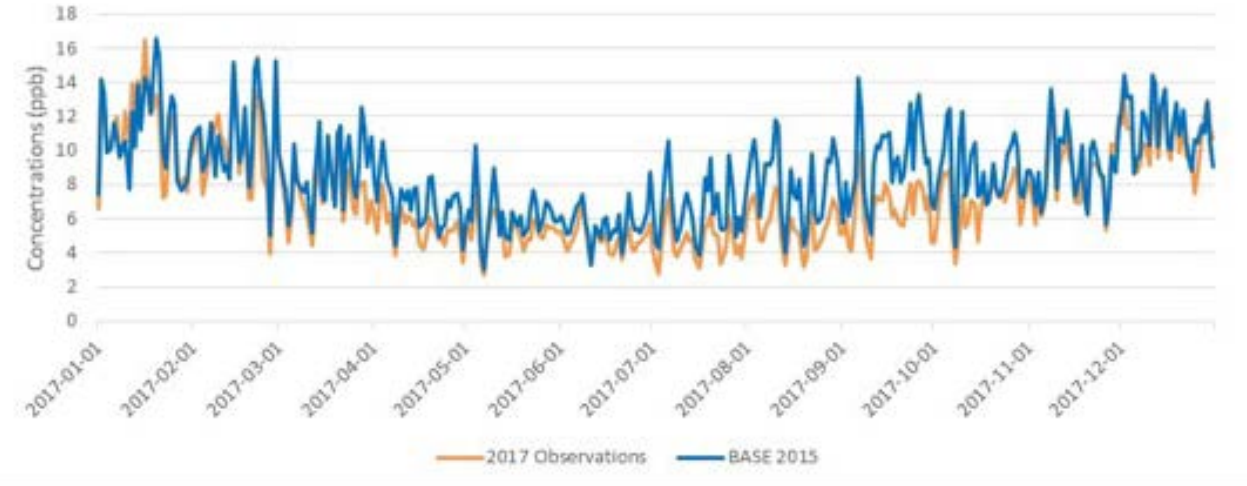


FIGURE D2: Modelled and observed average daily O_3 concentrations for stations located in metropolitan areas across Canada

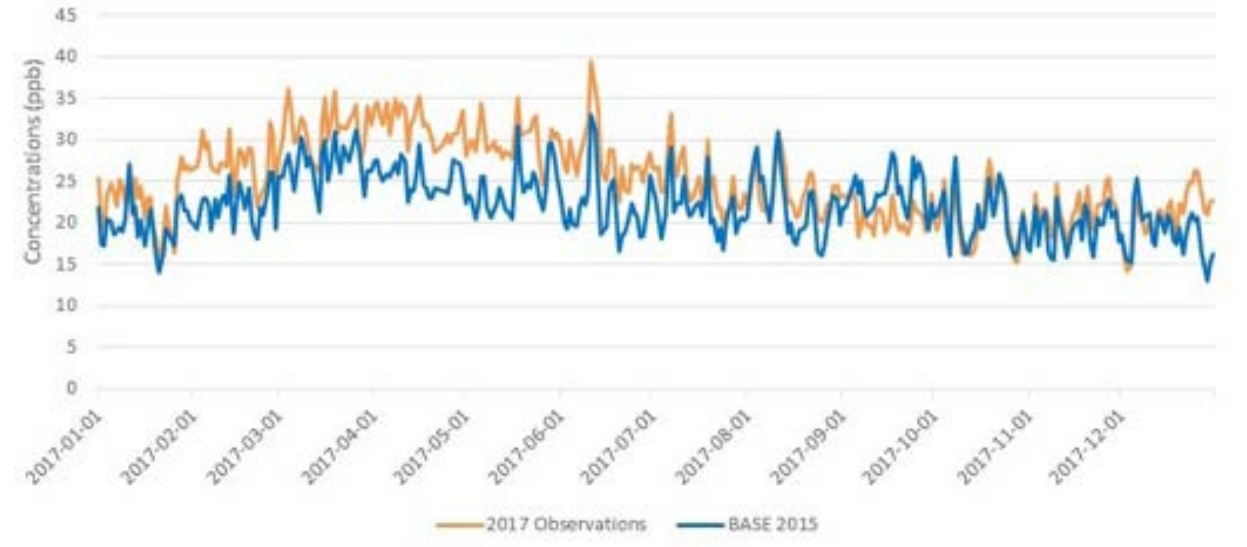


FIGURE D3: Modelled and observed average daily $PM_{2.5}$ concentrations for stations located in metropolitan areas across Canada