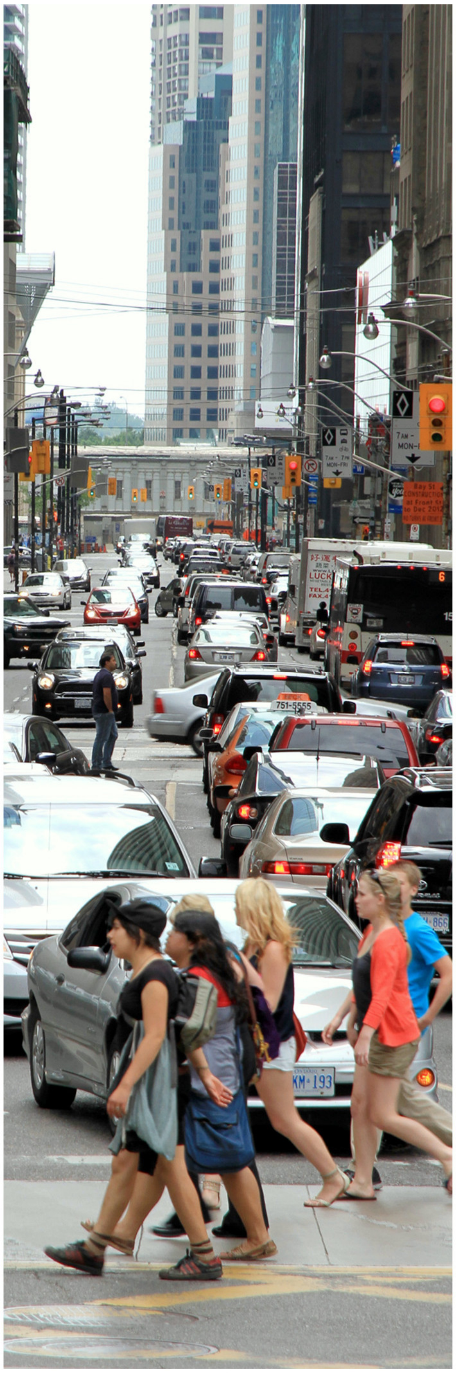

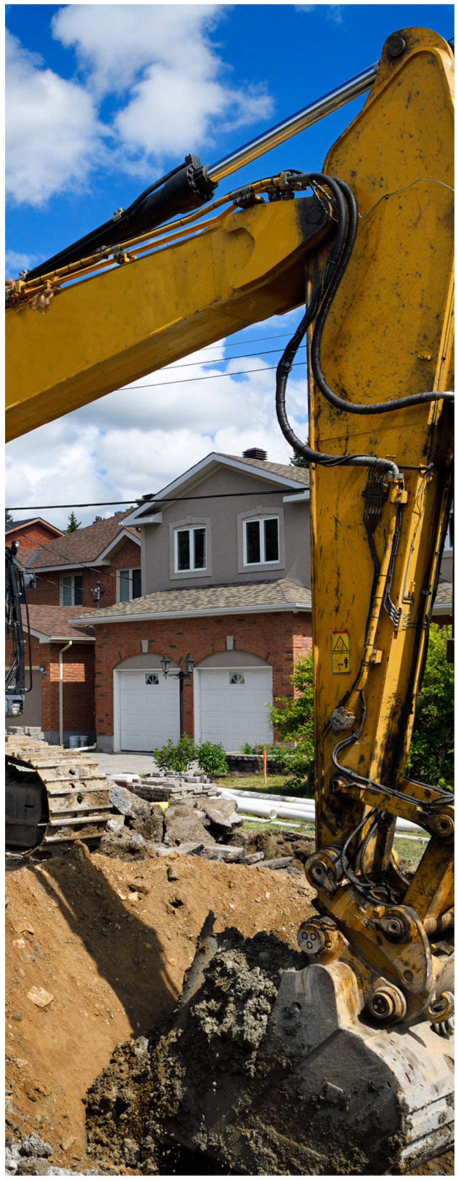

December 2022

HEALTH BENEFITS PER TONNE OF AIR POLLUTANT EMISSIONS REDUCTION

Region-, Sector-, and Pollutant-Specific Estimates for two Canadian Regions

Health Canada Santé Canada

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List of Acronyms

APEEP	Air Pollution Emission Experiments and Policy Analysis Model
AP2	Updated version of Air Pollution Emission Experiments and Policy Analysis Model
APEI	Air Pollutant Emissions Inventory
AQBAT	Air Quality Benefits Assessment Tool
AQMS	Air Quality Management System
BPT	Benefit Per Tonne
CAMx	Comprehensive Air Quality Model with Extensions
CA	Census Agglomeration
CAD	Canadian Dollar
CD	Census Division
CMA	Census Metropolitan Area
CMAQ	Community Multiscale Air Quality Model
CRF	Concentration-Response Function
EASIUR	Estimating Air Pollution Social Impact Using Regression Model
ECCC	Environment and Climate Change Canada
InMAP	Intervention Model for Air Pollution
GEM-MACH	Global Environmental Multi-scale – Modelling Air-quality and Chemistry
PM	Particulate Matter
SMOKE	Sparse Matrix Operator Kernel Emissions Model
SWBC	Southwestern British Columbia
TWBL	Tire Wear and Brake Lining
US	United States
USD	United States Dollar
US EPA	United States Environmental Protection Agency

VOC	Volatile Organic Compound
VSL	Value of One Statistical Life
WQCC	Windsor–Quebec City Corridor
WTP	Willingness To Pay

Executive Summary

A wide body of scientific literature provides evidence that exposure to ambient air pollution is associated with adverse health effects ranging from morbidity to mortality (Health Canada 2013, 2016; 2022; US EPA 2019, 2020). In Canada, it is estimated that 15,300 premature deaths annually are associated with the above-background component of ambient air pollution, with an estimated value of $120 billion per year (2016 CAD; Health Canada 2021).

The health benefits of air pollution mitigation strategies can comprise a significant portion of the overall benefits of initiatives affecting air pollution, including climate change initiatives. To aid in the assessment of health benefits from initiatives affecting air quality or climate, benefit-per-tonne (also referred to as BPT or marginal benefit) values are estimated in this report. BPTs are the monetized value of the health benefits of reducing air pollutant emissions by one tonne[1] ($ per tonne of emission reduction). This type of measure can be used to assess health impacts that broadly affect society.

Intended audience and use

To address the need for readily applicable estimates to support analyses of air pollution health impacts, Health Canada developed the set of BPT estimates presented in this report. These BPTs provide information for assessing the health impacts of air pollution mitigation strategies for regions and assessing the emissions changes of similar scale to those analyzed, without users having to undertake time-consuming, data-intensive, and costly air quality modelling. Health Canada's BPT estimates were developed using advanced air quality modelling that simulates complex atmospheric processes and interactions, and, importantly, the formation of secondary pollutants. Such modelling is necessary to understand regional air quality and pollutant transformation, and how air pollutants are transported, influencing air quality and health at large. While tools exist for assessing local air quality impacts that are less resource intensive (e.g., dispersion models), such tools do not account for pollutant transformation in the atmosphere and are not appropriate for assessing impacts far from the source.

This report is intended to support the estimation of health benefits of air pollution mitigation strategies by providing BPT estimates based on advanced regional air quality modelling that well represents the formation of secondary pollutants. It does not replace the need for advanced air quality modelling when the intervention in question differs substantially from those analyzed here, when the geographic region of the intervention differs from that assessed here, or when a regulation will result in large emissions changes or changes that affect Canada broadly. Due to assumptions made in this report and the broad categorization of emissions sectors analyzed, users may also wish to undertake their own advanced air quality modelling exercises to obtain higher-resolution data for their region.

[1] A tonne refers to a metric ton, i.e., 1,000 kg.

Scope

Health Canada has developed a set of region-specific, pollutant-specific, and sector-specific BPTs for two highly populated Canadian regions. The BPTs are intended to inform the evaluation of health impacts resulting from small to moderate emissions reductions in those regions, where full-scale modelling is not available or practical.

BPTs were estimated for two specific regions of Canada: southwestern British Columbia (SWBC), encompassing the Metro Vancouver region, Victoria, and the Fraser Valley Regional District; and the Windsor–Quebec City corridor (WQCC) extending from Windsor, Ontario, to Quebec City, Quebec. These two regions collectively represent over 23 million of the 35 million people in Canada based on the 2016 Census. BPTs were developed for a limited number of sectors, including on-road, off-road, an aggregate of manufacturing and ore and mineral industries, and agriculture, and for five emitted pollutants (primary $PM_{2.5}$, NO_x, SO_x, VOCs, and NH_3) in these regions. Health Canada has focused on estimating BPTs for on-road, off-road, manufacturing, and ore and mineral industry sectors as preliminary analyses identified these as important contributors to air pollution health impacts in Canada. The agricultural sector is the main source of NH_3 emissions in Canada and was added to assess the importance of NH_3 emissions reduction on ambient $PM_{2.5}$. The BPTs of oil and gas industries; commercial, residential, and institutional facilities; and the marine, rail, and aviation sectors were not derived due to resource limitations.

Methods

Health Canada identified concentration-response functions to estimate premature mortality and morbidity related to ambient $PM_{2.5}$, O_3, and NO_2 exposure for inclusion in this analysis, for which there was sufficient weight of evidence of causal effects. Exposure to these pollutants is associated with adverse health effects such as premature mortality, emergency room visits, hospitalizations, asthma exacerbation events, reduced activity days due to asthma, and other adverse effects.

As a result of complex chemical and physical atmospheric processes, emitted pollutants transform into secondary pollutants in the atmosphere, such as secondary $PM_{2.5}$ and O_3. The BPT estimates derived in this analysis rely on advanced air quality modelling conducted by Environment and Climate Change Canada (ECCC) to represent these complex processes, accounting for interactions between pollutants.

The modelling framework employed in this analysis entailed four steps:

1. Develop emissions reduction scenarios using 2015 emissions as the base case.
2. Model air quality changes anticipated from emissions reduction scenarios.
3. Estimate the health impacts in monetary terms resulting from the air quality changes.
4. Divide the total monetized health impacts by the total emissions change to yield BPT values.

This modelling is data-intensive, time-intensive, and resource-intensive, but it allows for the derivation of scientifically supported BPT estimates that can then be used broadly by jurisdictions to streamline the assessment of health impacts associated with changes in air pollutant emissions.

Results

BPT estimates reported in Table ES1 represent the overall monetary value of health impacts per tonne of emissions reduced, where those health impacts occur across the Canadian population (i.e., both within and outside of SWBC or the WQCC). Thus, BPTs in Table ES1 refer to the total health benefits from a one-tonne reduction of the pollutant in question in the specified region for the 2015 base case, taking into account its long-range transport and photochemical and physical transformations. BPTs are also reported by the region in which health impacts occur and by ambient pollutant associated with health impacts ($PM_{2.5}$, O_3, and NO_2) in the main report. The changes in emissions and resulting changes in health impacts used to derive these BPTs are presented in the main report.

Table ES1. Benefit-per-tonne (BPT) estimates by region, source sector, and emitted pollutant

Emitted pollutant	Source sector	BPT ($/tonne)[a,b,c,d]	
		SWBC	WQCC
Primary $PM_{2.5}$	On-road mobile	410,000	520,000
	Off-road mobile	470,000	480,000
	Manufacturing + Ore and mineral industries	340,000	380,000
NO_x	On-road mobile	-140[e]	15,000
	Off-road mobile	-2,700[e]	12,000
	Manufacturing + Ore and mineral industries	-3,900[e]	4,900
VOC	On-road mobile	13,000	3,900
	Off-road mobile	9,900	5,100
	Manufacturing + Ore and mineral industries	3,900	2,300
NH_3	On-road mobile	100,000	130,000
	Agriculture	46,000	26,000
SO_x	Manufacturing + Ore and mineral industries	–	10,000

[a] BPTs reflect the marginal change in societal economic welfare attainable from reducing emissions of a pollutant by one tonne, and include direct, indirect and intangible costs such as pain and suffering.

[b] BPTs are reported in 2015 CAD per tonne of reduction in emitted pollutant. Health benefits reflect the combined health impacts due to changes in ambient $PM_{2.5}$, O_3, and NO_2 concentrations.

[c] SWBC refers to the southwestern BC region; WQCC refers to the Windsor–Quebec City corridor.

[d] Estimates are rounded to two significant figures.

[e] Negative BPTs are due to NO_x titration and should be considered carefully. Users are referred to the "Limitations of Health Canada's BPT estimates" section in the main report for discussion.

Key highlights

- BPTs from primary $PM_{2.5}$ emissions reduction are the largest in magnitude of all emitted pollutants due to direct reductions in ambient $PM_{2.5}$ concentrations and associated risks, and range from $340,000 to $520,000 per tonne of primary $PM_{2.5}$ emissions reduction.
- BPTs for gas-phase pollutants (NO_x, VOCs, SO_x, NH_3) result to a large extent from their roles in the formation of secondary ambient pollutants and thus depend more strongly on weather conditions and the atmospheric mix of reactive pollutants.

- NH_3 has the largest BPTs of the gaseous precursor pollutants (NO_x, SO_x, VOCs, and NH_3) due to its role in the production of secondary $PM_{2.5}$ (ammonium sulphate and nitrate). NH_3 BPTs vary substantially across the sectors analyzed.
- BPTs for NO_x were positive for the WQCC, but negative for the SWBC sectors analyzed due to NO_x titration (i.e., a reduction in NO_x leads to an increase in O_3 and associated health impacts). Negative NO_x BPTs should be considered with caution and are expected to diminish with progressive and large-scale reductions in NO_x emissions (or NO_x and VOC emissions) that lead to a shift in the chemical state of the atmosphere. Further, negative NO_x BPTs can be offset by emissions reductions of other pollutants that lead to improvements in O_3, such as VOCs (e.g., off-road BPTs are \$9,900 per tonne of VOC vs -\$2,700 per tonne of NO_x in SWBC). Users are referred to the "Limitations of Health Canada's BPT estimates" section in the main report for a detailed discussion of negative BPTs.
- The vast majority of health impacts captured in BPTs are due to health impacts occurring within the same region as the emissions reduction.

The health benefits of emission mitigation options can be estimated by multiplying the emissions change resulting from the measure in question (i.e., the number of tonnes of emission of the pollutant) by the corresponding BPT. The health benefits of a given mitigation action thus depend not only upon the magnitude of BPT, but also the magnitude of the emissions change. For policies where emissions of multiple air pollutants are simultaneously reduced (e.g., $PM_{2.5}$, NO_x, VOCs, and NH_3 from adoption of zero-emission vehicles), the total health benefits of the mitigation measure are assumed to equal the sum of health impacts from reducing each emitted pollutant separately.

Limitations

The limitations of these BPT estimates require careful consideration to assess the suitability of applying BPTs to evaluate emission mitigation options. Users of these BPTs are referred to the main report for the full set of limitations. The key ones are summarized here:

- BPT estimates apply to the regions, sectors, and pollutants modelled and should be applied within similar emissions reduction contexts to those analyzed. BPTs represent the SWBC and WQCC regions broadly and may not specifically reflect impacts from a reduction in emissions from any particular location within the region. Intra-regional variability in BPTs may be significant and was not assessed. BPTs are most applicable to regional-scale analyses and may be less representative of smaller geographic areas that are distant from where the majority of the region's air pollutant emissions are. For users wishing to evaluate the health impacts of emissions reduction policies for smaller geographic areas, such as for an individual municipality within the region, these regional BPTs are considered to be more relevant than national BPTs available elsewhere, but with increased uncertainty. The level of uncertainty increases as the geographic area in question is located further away from the bulk of emissions in the region for the sector in question. BPTs are not transferrable to other regions, even those with similar population densities or similar geographic distributions of emissions relative to populations.

- BPTs were derived using emission reductions applied to one sector across the entire region; however, the health benefits estimated include benefits that occur within and outside of the study region.
- BPTs are appropriate for evaluating health benefits from small to moderate emissions changes. BPTs are recommended for emission changes of up to +/− 10% of the total pollutant emissions for the region of interest. Larger-scale changes in emissions are better assessed with full-scale air quality modelling due to nonlinear responses that are not represented by the scenario-based modelling approach used here. BPTs are therefore not recommended to be used for larger emission changes.
- BPTs reflect annual health benefits resulting from annual changes in emissions from the 2015 baseline levels, and do not represent time-varying estimates (e.g., seasonal or daily).
- Uncertainty in BPT estimates is reported in the main report and reflects uncertainties in concentration-response relationships and economic value assumptions. Additional uncertainties exist in modelling the macro economy, emissions, and complex atmospheric processes leading to estimates of ambient air pollutant concentrations, but these have not been quantified.

Introduction

A large body of scientific literature provides evidence that exposure to ambient air pollution is associated with a wide range of adverse human health effects ranging from respiratory symptoms to the development of disease and premature death. The collective scientific evidence on morbidity and mortality related to short- and long-term fine particulate matter ($PM_{2.5}$), ground-level ozone (O_3), and nitrogen dioxide (NO_2) exposures has been reviewed by Health Canada (Health Canada 2013, 2016, 2022) and internationally (e.g., US EPA 2019, 2020). The relationship between exposure and the risk of adverse health effects is well characterized for many endpoints, which allows for quantifying the population health impacts associated with a change in air pollution and valuation of those health impacts. This type of analysis is termed health impact assessment. This routinely draws on the epidemiological literature in combination with estimates of population counts, baseline incidence rates, and economic valuation to derive air pollution–attributable health outcomes and monetized values. In Canada, an estimated 15,300 premature deaths are associated with the anthropogenic component of $PM_{2.5}$, O_3, and NO_2 exposure annually, with a monetary value of $120 billion per year (2016 CAD; Health Canada 2021).

Health impact assessments provide key information to inform decision-making on initiatives affecting emissions of air pollutants and air quality, and environmental risk management more generally. Health impact assessments often rely on resource-intensive, regional air quality modelling to assess the formation of important secondary pollutants that affect health (e.g., secondary $PM_{2.5}$, O_3), and to assess relationships between emitted pollutants and the exposure of populations. There is a need for simple and robust tools to evaluate the health impacts of emissions sources directly, to avoid undertaking advanced air quality modelling for each emission mitigation measure in question. One such tool is the benefit-per-tonne (also known as BPT or marginal benefit), which refers to the monetary value of averted mortality and morbidity resulting from one tonne of air pollutant emissions reduction, expressed in units of dollars ($) per tonne of emitted pollutant. BPT is a tool that enables decision makers to quantify and compare the potential health benefits of multiple mitigation options based solely on a given emission decrease and without undertaking complex air quality modelling. BPT can also be applied to evaluate the health impacts of a small to moderate increase in emissions, rather than a reduction, in which case the sign of the impact is reversed (i.e., if an emission reduction incurs a benefit, an emission increase would incur a cost or disbenefit). The per-tonne standardized unit of BPT facilitates comparison across emission mitigation options, with the costs of emissions control in a benefit-cost analysis framework. BPT estimates can be informative for air pollution risk management where full-form air quality modelling is infeasible. Federal, provincial, and municipal governments can use BPTs to evaluate air pollution mitigation strategies, regulatory impact assessments, and enforcement cases. Other organizations may also want to use BPTs to evaluate mitigation options and project initiatives.

Health Canada led an analysis to estimate BPTs for select regions, sectors, and pollutants in Canada. Health Canada and Environment and Climate Change Canada (ECCC) conducted regional air quality modelling and health impacts modelling to produce the BPT estimates in this report. BPTs vary geospatially based on the proximity of emission sources to large populations, the height at which emissions are released, meteorological conditions affecting the transport pathway, and atmospheric conditions and mixtures affecting the transformation into secondary pollutants. BPTs estimated in this report are location-, sector-, and pollutant-specific, and are derived for two Canadian regions.

A number of modelling approaches exist for estimating BPTs that vary in complexity, resource requirements, and the ability to delineate source sectors, pollutants, locations, and emission time periods (e.g., seasons). Full-form air quality models (also referred to as chemical transport models) are the most resource-intensive method of generating BPTs. Chemical transport models estimate hourly, gridded ambient pollutant concentrations from emissions, meteorological, and geophysical input data. Examples of BPT estimation using such models are Fann et al. (2012), whose work employed source apportionment in the Comprehensive Air Quality Model with Extensions (CAMx) model for the United States (US), and Pappin and Hakami (2013) using the adjoint (a full-form sensitivity analysis tool) of the Community Multiscale Air Quality model (CMAQ). The chemical transport model used operationally by ECCC is the Global Environmental Multi-scale – Modelling Air-quality and Chemistry (GEM-MACH) model.

Of the available methods, chemical transport models incorporate the least simplifying assumptions and are better able to capture complex and often nonlinear pollutant transformations with the most accuracy. Compared with these models, a number of reduced-form or reduced-complexity models have been developed to evaluate the health benefits of emissions reductions and BPTs in the US (e.g., Fann et al. 2009; Gilmore et al. 2019; Goodkind et al. 2019; Heo et al. 2016; Lee et al. 2015; Muller and Mendelsohn 2012; Tessum et al. 2017). These tools vary in their resolution of locations/regions, sectors, and pollutants, and entail simplifying assumptions to enable a large number of scenarios to be efficiently run. Reduced-form or reduced-complexity models have not yet been developed or tailored for the Canadian context.

The goal of this report is to provide a set of **region-specific, pollutant-specific, and sector-specific BPTs** for two populous regions in Canada, developed using chemical transport modelling. The modelling approach used ECCC's operational chemical transport model, GEM-MACH, and Health Canada's Air Quality Benefits Assessment Tool (AQBAT). BPTs are estimated for the Windsor–Quebec City Corridor (WQCC) and southwestern British Columbia (SWBC) for four sectors (on-road, off-road/non-road, manufacturing and ore and mineral industries, and agriculture) and five emitted pollutants ($PM_{2.5}$, NO_x, SO_x, VOCs, and NH_3).

Methods

Prior to undertaking the air quality and health impacts modelling, two source regions were delineated. Emissions reduction scenarios were developed over these regions in order to estimate region-specific, sector-specific, and pollutant-specific BPTs. These regions were modelled in a larger domain covering Canada and the US (Figure 1) that accounted for emissions from both countries in order to estimate ambient pollutant concentrations. Further details on the modelling methods can be found in the methods section, "Step 2. Air quality/chemical transport modelling."

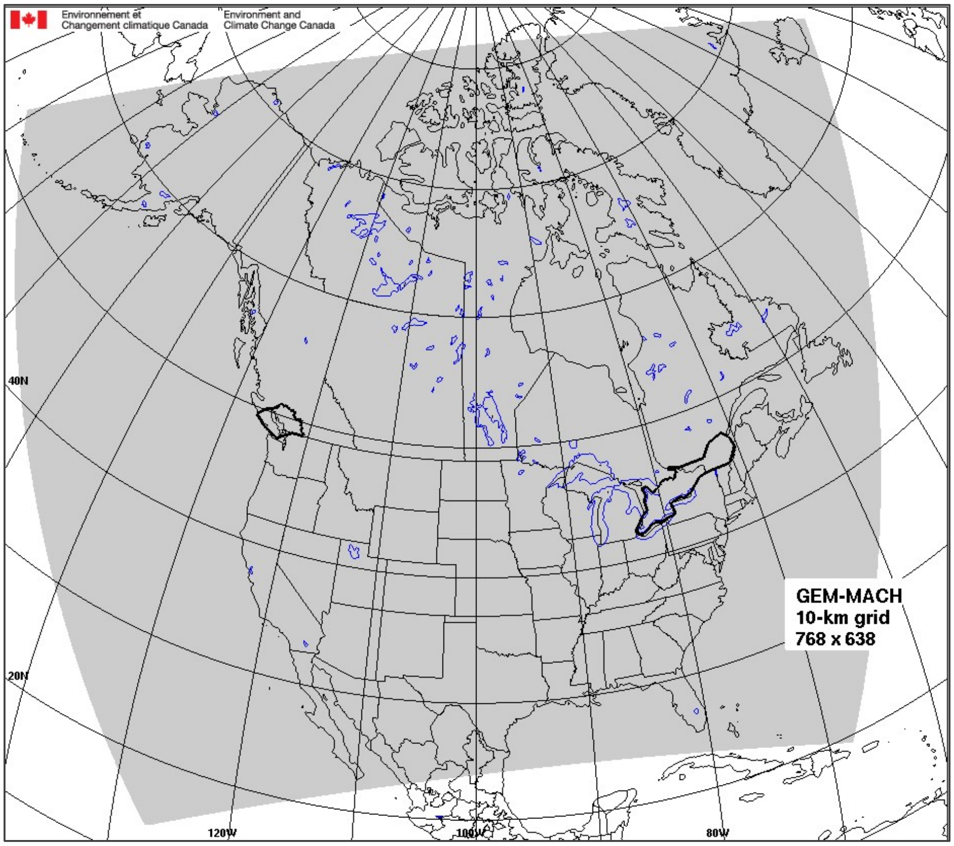

Figure 1. Map of GEM-MACH domain. Grey shaded area represents modelling domain with a 10-km grid cell resolution. Bold black polygons show boundaries of the southwestern BC (SWBC) and Windsor–Quebec City Corridor (WQCC) regions. Blue lines represent bodies of water.

Delineation of regions for BPT estimation

BPTs were estimated for the Southwestern British Columbia (SWBC) and the Windsor–Quebec City Corridor (WQCC) regions (Figure 2). SWBC and the WQCC together comprise 65% of Canada's population (Statistics Canada 2018) and possess characteristics of broad relevance to air quality management in Canada. These regions each contain multiple census metropolitan areas (CMAs), census agglomerations (CAs), and census divisions (CDs). BPTs developed for SWBC and the WQCC are intended for evaluating the health impacts of emissions reduction strategies that apply broadly to sources across the regions.

Region 1 – SWBC – contains metropolitan Vancouver and Victoria, and extends over all of the Lower Fraser Valley and Georgia Strait Air Zones, as well as the southern portion of the Coastal Air Zone (Figure 2; top panel). Air quality in SWBC is strongly influenced by local topography and stagnation events, which can lead to spikes in air pollution (FVRD et al. 2014). Key sources of emissions affecting ambient air pollution in SWBC include on-road transportation, off-road mobile sources, marine emissions, heating, industrial sources (FVRD et al. 2014) and wildfire smoke events. The SWBC region defined in this analysis has a population of 3.6 million people and contains 12 CMAs and CAs based on the 2016 census (Statistics Canada 2018). Figure 2 is mapped using boundaries from the 2011 Census for consistency with the health impacts modelling, which relied upon 2011 census geography.

Region 2 – the WQCC – was defined in this analysis as encompassing Windsor, Ontario, in the southwest through the Greater Toronto Area, the National Capital Region, Montreal, and Quebec City in the northeast (Figure 2; bottom panel). The WQCC contains 19.8 million people and 57 of the 156 CMAs in Canada based on the 2016 Census (Statistics Canada 2018). Key sources of air pollution for Ontario broadly include agriculture, transportation, electricity generation, industrial facilities, and residential wood combustion (analogous to home firewood burning[2]). Transboundary flow of air pollution across the Canada–US border contributes to background O_3 levels and plays an important role in local air quality due to the orientation and proximity of the region with respect to the US. The global background is also increasingly relevant as emissions have declined over time in North America (MOECC 2018).

The SWBC and WQCC regions were delineated using GIS tools to overlay boundaries of air zones, CDs, CMAs, and CAs. Air zones are management areas that have unique air quality characteristics, such as pollutant sources, topography, meteorological patterns, and population density (CCME 2019). ECCC geospatially mapped inventories of $PM_{2.5}$, NO_x, and VOC emissions over the modelling domain to ensure that key sources were captured within the regions. Where possible, BPT regions followed air zone boundaries in order to best support air quality management under the Canada-wide Air Quality Management System (AQMS).

[2] Home firewood burning is defined by ECCC as the "burning of wood, pellets and manufactured logs as fuel for space heating and hot water, [which includes] emissions from fireplaces, wood stoves and wood-fired boilers" (ECCC 2022).

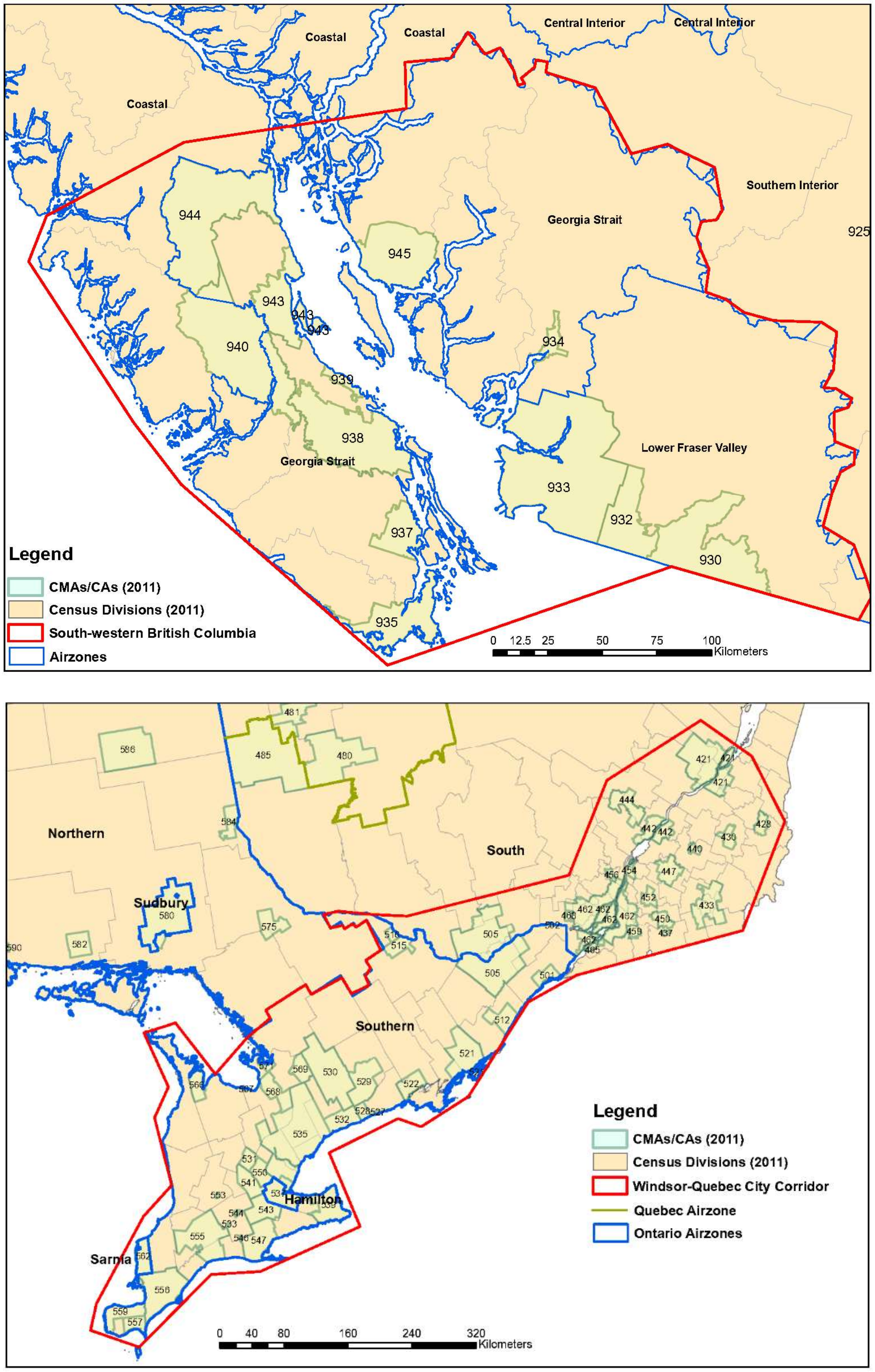

Figure 2. Delineation of the southwestern British Columbia (SWBC) region (top panel) and Windsor–Quebec City corridor (bottom panel). Census divisions (CDs), census metropolitan areas (CMAs), and census agglomerations (CAs) are based on 2011 Census geography as used in AQBAT. Blue and green lines depict boundaries of air zones.

Selection of emissions sectors

Previous studies have identified transportation, residential combustion, industrial sources, agriculture, and wildfires as key contributors to population-weighted ambient $PM_{2.5}$ concentrations in Canada (Meng et al. 2019). Preliminary analyses undertaken by Health Canada suggest that home firewood burning[3], transportation, and industrial sources sectors are important contributors to air pollution-related health impacts in Canada. Due to constraints in computational resources, BPTs were estimated for a limited number of emissions sectors including on-road transportation, off-road transportation, an aggregate of manufacturing and ore and mineral industries, and agriculture. Agriculture was included as it is an important source of NH_3 emissions in Canada, which impacts secondary $PM_{2.5}$ (through ammonium sulphate and ammonium nitrate), and studies on agricultural emissions are limited in Canada.

Definitions of the chosen sectors are provided in the section "BPT emissions reduction scenarios." BPTs were not estimated for other sectors (e.g., oil and gas industries; electricity generation; residential, commercial, and institutional emissions such as home firewood burning; and air, marine, and rail emissions); however, those sectors were included in the emissions inventory used for the base case scenario. Health Canada may consider additional sectors such as these in future iterations of this analysis.

Modelling framework

Four steps were undertaken in the modelling to estimate a set of BPTs for SWBC and the WQCC. First, it was necessary to estimate time-varying, gridded emissions of air pollutants across Canada and the US. Emissions were modelled for a series of emissions reduction scenarios, compared with 2015 base case emissions. Each emissions reduction scenario entailed a 10% reduction in emissions of one pollutant from one sector in one region, relative to the base case. Each of the scenarios was modelled using a chemical transport model (step 2) to estimate ambient pollutant concentrations across the country. In step 3, health impacts were estimated from the differences in pollutant concentrations between the base case and each emissions reduction scenario. The final step (step 4) was to calculate BPTs from the emissions and health impacts modelling. Details on each of these steps are provided in the sections that follow.

Modelling Framework:

1. Emissions modelling of base case and emissions reduction scenarios.
2. Chemical transport modelling.
3. Health impacts analysis and valuation.
4. Benefit-per-tonne estimation.

Using this methodology, BPTs were estimated for SWBC and the WQCC for four sectors (on-road, off-road/non-road, an aggregate of manufacturing and ore and mineral industries, and agriculture), and five emitted pollutants (primary $PM_{2.5}$, NO_x, SO_x, VOCs, and NH_3). The health benefits were assessed for

[3] Home firewood burning is defined by ECCC as the "burning of wood, pellets and manufactured logs as fuel for space heating and hot water, [which includes] emissions from fireplaces, wood stoves and wood-fired boilers (ECCC 2022).

exposure to three ambient air pollutants, including NO_2, $PM_{2.5}$ (including primary and secondary components as captured by the chemical transport model), and O_3.

Step 1. Emissions modelling

Emissions modelling was conducted for a base case and a number of emissions reduction scenarios for the year 2015. This section describes the emissions modelling that was undertaken (step 1 in the modelling framework) to develop these scenarios.

Base case scenario

The 2015 Air Pollutant Emissions Inventory (APEI) was used as the base case for this analysis and was developed by ECCC using methodology consistent with the 2017 APEI release year (Sassi et al. 2021). At the time of initiation of this work, 2015 was the most current inventory year available for the updated version of the chemical transport model used in step 2 (REQA GEM-MACH v3). The APEI is a comprehensive inventory of air pollutants at the national, provincial, and territorial levels. The APEI compiles emissions of 17 air pollutants contributing to smog, acid rain, and poor air quality since 1990. Wildfire emissions are not included in the emissions inventory. Emissions from the US are included in the modelling and are based on the 2011 NEI, projected to 2017 (US EPA 2016).

The 2015 APEI was mapped temporally, spatially, and by pollutant species using the Sparse Matrix Operator Kernel Emissions (SMOKE) model version 3.7 (UNC 2014) to develop gridded, temporally varying emissions for use in GEM-MACH (step 2). Wildfire emissions were not included in the GEM-MACH version used in this study, nor were aeolian dust emissions (distinct from anthropogenic fugitive dust emissions, including agricultural activities) nor lightning NO emissions.

The gridded 2015 base case emissions from SMOKE were summed over all grid cells within SWBC and the WQCC, and all hours within the year, to estimate the total emissions released from specific sector groupings in these regions. This calculation was performed separately for aggregate sectors, and for all emissions sources together (Table 1). Users are recommended to consult this table when evaluating the suitability of BPTs for emission mitigation applications (refer to the section "Limitations of Health Canada's BPT estimates" for further discussion).

Table 1. Emissions totals for the SWBC and WQCC regions based on allocation of 2015 Air Pollutant Emissions Inventory (APEI) using an emissions processor for a 10-km gridded domain.

Region	EMISSIONS (TONNES PER YEAR)[a]				
	Primary $PM_{2.5}$	NO_x	SO_2	VOCs	NH_3
Southwestern British Columbia (SWBC)					
Agriculture[b]	17	19	-	4,200	6,000
On-road	1,100	34,000	100	12,000	540
Off-road	910	8,500	10	6,300	13
Air, marine, and rail	350	18,000	500	1,700	21
Manufacturing, and ore and mineral industries	1,800	9,300	4,900	4,800	320
Other NPRI sources[c]	84	660	630	980	710
Upstream oil and gas	-	-	-	-	-
Other area sources[d]	8,700	4,500	240	32,000	160
Windsor–Quebec City Corridor (WQCC)					
Agriculture[b]	8,000	1,400	920	39,000	130,000
On-road	4,900	140,000	770	42,000	3,100
Off-road	6,400	63,000	72	46,000	94
Air, marine, and rail	730	33,000	970	5,000	26
Manufacturing, and ore and mineral industries	15,000	48,000	74,000	46,000	1,500
Other NPRI sources[c]	1,300	21,000	28,000	6,700	2,200
Upstream oil and gas	28	750	1,100	540	3
Other area sources[d]	90,000	34,000	3,500	220,000	1,300

[a] Estimates are rounded to two significant figures.

[b] Agriculture refers to all agricultural emissions, including dust and agricultural sources included in the "OTHER" category by SMOKE.

[c] Other NPRI sources refers to minor and major point sources in the National Pollutant Release Inventory (NPRI). Manufacturing and ore and mineral industries have been removed and are presented separately.

[d] Other area sources refers to all area sources not already isolated (including residential wood combustion, dust not related to agriculture, some upstream oil and gas, some residential/commercial/institutional sources, and some paint and solvents).

Adjustment factors were developed for specific emissions subsectors of the published 2015 APEI estimates (Sassi et al. 2021) to address errors identified in the original inventory. These factors were applied only to total annual emissions of $PM_{2.5}$ and PM_{10}, unless otherwise noted, prior to allocating them temporally, spatially, and chemically in SMOKE:

- Adjust non-residential (heavy) construction emissions by a factor of 0.05 (a decrease of 95%);
- Adjust paved road dust emissions by a factor of 0.15 (a decrease of 85%);
- Adjust unpaved road dust emissions by a factor of 1.11 (an increase of 11%);
- Remove wind erosion emissions from agriculture (set to zero); and

- Substitute 2010 APEI residential wood combustion emissions for all pollutants, representing decreases of 35% to 48%.

BPT emissions reduction scenarios

Each BPT estimate was derived from a scenario that entailed a 10% reduction in emissions of a relevant pollutant, emitted from one sector, in one region, relative to the 2015 base case. Emission reductions were not applied nationally nor were they applied to multiple sectors or multiple pollutants simultaneously. Emissions reduction scenarios were developed for four broad emissions sectors (on-road mobile, off-road mobile, an aggregate of industrial sources, and agriculture), and five emitted pollutants ($PM_{2.5}$, NO_x, SO_x, VOCs, and NH_3). These sectors are defined below and are based broadly on APEI categorizations.

On-road mobile sector

- Includes exhaust emissions, evaporative emissions, and tire wear and brake lining (TWBL) emissions from on-road light-duty and heavy-duty vehicles of all fuel types, as well as motorcycles. Evaporative emissions from refuelling stations or fuel storage facilities were not captured within this sector in this analysis.
- Modelled using the Motor Vehicle Emission Simulator (MOVES) v2014 (US EPA 2014).

Off-road mobile sector

- Includes exhaust emissions from equipment used off-road of all fuel types, including construction equipment, lawn equipment, agricultural equipment, go-karts, snowmobiles, etc. Air, marine, and rail emissions were excluded from off-road emissions as a BPT sector due to their dissimilar spatial distribution compared with other off-road sources, but were included in the base case.
- Modelled using the NONROAD model v2012C (US EPA 2005).

Manufacturing and ore and mineral industries

- Manufacturing includes all sub-sectors as defined in the APEI: abrasives manufacturing, bakeries, biofuel production, chemicals industry, electronics, food preparation, glass manufacturing, grain industry, metal fabrication, plastics manufacturing, pulp and paper industry, textiles, vehicle manufacturing, wood products, and “other” manufacturing.
- Ore and mineral industries include all sub-sectors as defined in the APEI: aluminium industry, asphalt paving industry, cement and concrete industry, foundries, iron and steel industry, iron ore industry, mineral products industry, mining and rock quarrying, and non-ferrous refining and smelting industry.
- While APEI manufacturing and ore and mineral industries contain dissimilar sources, these sectors were modelled together in order to represent stationary sources from these sectors in aggregate. Sources from the petroleum industry were not included in this BPT sector, but were included in the base case.

- Industrial point source emissions were allocated geographically in SMOKE using the latitude, longitude, and facility's stack information (e.g., stack height). Industrial area source emissions in SMOKE do not have stack information and were allocated using spatial surrogates.

Agriculture

- Includes emissions from animal production (NH_3 volatilization from manure, particulate matter (PM) from feeding and housing, and non-methane VOCs from livestock feeding, housing, and manure management); crop production (harvesting, inorganic fertilizer application, sewage sludge application, and tillage practices); and fuel use distinct from off-road agricultural equipment (captured in off-road sector). Wind erosion emissions were set to zero as they are not an anthropogenic source. Emissions from agricultural equipment were included in the off-road mobile sector.

Step 2. Air quality / chemical transport modelling

In step 2 of the modelling, ECCC's GEM-MACH model was used to assess the effect of changes in emissions (between the 2015 base case and emissions reduction scenarios) on ambient pollutant concentrations. GEM-MACH is a multi-phase, multi-pollutant chemical transport model that considers the interactions of gas-, aqueous-, and particle-phase chemical components. The version of GEM-MACH used in this analysis was REQA GEM-MACH v3 (hereafter referred to as GEM-MACH). GEM-MACH uses the ADOM-2 gas-phase chemical mechanism to model gas-phase chemistry with 42 species and 114 reactions (Stockwell and Lurmann 1989; Stroud et al. 2008; Venkatram et al. 1992). Gas-phase chemistry in GEM-MACH does not include halogens (halogen effects on O_3 are not captured). Aqueous-phase chemistry in GEM-MACH uses an updated version of the ADOM mechanism with 13 species and 25 reactions (Fung et al. 1991; Gong et al. 2006). Aerosol dynamics include parameterizations of nucleation, condensation, coagulation, dry deposition, aerosol-cloud interactions, and cloud scavenging (Gong et al. 2003); inorganic aerosol thermodynamics (Makar et al. 2003); secondary organic aerosol chemistry using the modified Odum yield approach (Stroud et al. 2011); and cloud processing (Gong et al. 2015; Makar et al. 2003, Mashayekhi et al. 2021; Stroud et al. 2011).

GEM-MACH represents ambient PM by eight chemical components, including elemental carbon, primary organic matter, secondary organic matter, sulphate, nitrate, ammonium, crustal material, and sea salt. GEM-MACH uses a simplified, two-bin, sectional PM size distribution with Stokes diameter size bins of 0–2.5 and 2.5–10 µm to represent fine PM ($PM_{2.5}$) and the coarse fraction of PM_{10}, respectively, to reduce computational expense (Mashayekhi et al. 2021). The two-bin distribution is temporarily redistributed to a finer size resolution for aerosol microphysics processes which are heavily size dependent, such as coagulation. The $PM_{2.5}$ diagnostic is dry $PM_{2.5}$ at ambient temperature and pressure.

BPTs were derived for the following emitted pollutants in GEM-MACH:

- Primary $PM_{2.5}$ (includes directly emitted primary organic carbon, elemental carbon, crustal material, sulphate, nitrate, and ammonium)
- NO_x ($NO + NO_2$)
- SO_x ($SO + SO_2$)
- NH_3

- VOCs (includes alkenes, alkanes, acetaldehyde, aromatics, propane, O-creosol, ethene, formaldehyde, isoprene, methyl ethyl ketone, and toluene, with differing chemical reactivity)

Each emissions reduction scenario entailed a 10% reduction in emissions of one of these pollutants from one sector in SWBC or the WQCC, relative to the 2015 base case. It is noted that CO emissions contribute to the formation of O_3, but CO BPTs were not modelled in the analysis because of resource limitations, and as their magnitudes are expected to be small compared with other precursors in urban areas.

GEM-MACH was run over a gridded, continental domain with 10-km by 10-km grid cells (Figure 1) and 80 hybrid layers extending from the surface (1.5 m) to ~60 km elevation (0.1 hPa). Ambient pollutant concentrations were computed within the model at 5-minute intervals and output on an hourly basis for the full calendar year of 2019. The GEM-MACH simulations used meteorological data (e.g., sunlight, wind speed and direction, relative humidity, temperature) from 2019. At the time of initiation of the modelling, 2019 was the only meteorological year available for running REQA GEM-MACH v3. As the GEM-MACH domain extends over Canada and the US, emissions from both countries are modelled (Figure 1).

The performance of various GEM-MACH versions has been evaluated in several ways against surface measurements and peer models (Mashayekhi at al. 2021). Studies suggest good overall performance compared to observations and the model is considered to be among one of the best systems when compared to its peers (Manseau et al. 2021). The performance of the current GEM-MACH model used by REQA (REQA's GEM-MACH v3) was evaluated using the 2015 emissions and 2019 meteorology compared to the 2019 air quality observations for $PM_{2.5}$, O_3, and NO_2. Performance evaluation statistics compare modelled and observed hourly concentrations across Canada over a one-year period. For NO_2, a correlation (R) of 0.65 and a mean bias of 0.40 ppbv were estimated, with a higher overestimation of NO_2 in eastern Canada. For O_3, a correlation of 0.71 and mean bias of -3.29 ppbv, representing an overall underestimation, were found. For $PM_{2.5}$, a correlation of 0.33 and a mean bias of -0.99 µg/m^3 were estimated over Canada. A stronger correlation and smaller underestimation were found for $PM_{2.5}$ over eastern Canada compared to western Canada. This is most likely due to the lack of forest fire emissions included in the 2015 modelling, whose large influence can be seen in the observations (Popadic et al. 2021). Gridded, hourly GEM-MACH concentration estimates were spatially and temporally processed for use in AQBAT, whose geographic unit of analysis is based on census geography rather than GEM-MACH grid cells. More on this process can be found in "Step 3. Health impacts assessment."

Step 3. Health impacts analysis

Health impacts or benefits were estimated using Health Canada's Air Quality Benefits Assessment Tool (AQBAT) version 3.0 in step 3 of the modelling. Health impacts were derived for changes in ambient $PM_{2.5}$, O_3, and NO_2 exposure for each emissions reduction scenario compared with the base case. Estimating these health impacts relies on the following parameters:

1. Ambient pollutant concentrations for base case and emissions reduction scenarios
2. Population
3. Baseline mortality or morbidity rate (baseline health endpoint rate)
4. Concentration-response function (risk/effect estimate, shape of curve)
5. Monetization

The geographic unit of analysis in AQBAT is the CD. There are 293 CDs across Canada in AQBAT based on 2011 Census geography (Statistics Canada 2012). Area-weighted average concentrations of $PM_{2.5}$, O_3, and NO_2 were calculated for each CD from GEM-MACH 10-km, gridded concentrations. Population and baseline health endpoint rates are assigned to each CD in AQBAT. The concentration-response function, or CRF, describes the relationship between exposure and risk of the health endpoint in question, and is applied to estimate counts of morbidity or mortality, which are also monetized into dollar values. Details on these five parameters are provided below. Users looking for additional information are referred to the AQBAT 3.0 User Guide (Judek et al. 2019).

Concentration change

Air pollutants emitted from a source become mixed in ambient air and may transform into other pollutants through chemical and physical processes. For example, emitted $PM_{2.5}$ contributes directly to primary $PM_{2.5}$ concentrations in ambient air, while emissions of NO_x, SO_x, VOCs, and NH_3 contribute to other $PM_{2.5}$ components – referred to as secondary $PM_{2.5}$ – such as secondary organic aerosol (SOA), sulphate, nitrate, and ammonium. NO_x and VOC emissions also influence the photochemical production of O_3 (a secondary pollutant), and thus affect ambient O_3 concentrations. This distinction between **pollutant emissions** and **ambient pollutant concentrations** is important. BPTs represent the health impacts of reducing an **emitted pollutant** (primary $PM_{2.5}$, NO_x, SO_x, NH_3, or VOCs) on **ambient concentrations of $PM_{2.5}$, O_3, and NO_2.** Those impacts are represented as health impacts due to changes in ambient $PM_{2.5}$, O_3, and NO_2 concentrations.

AQBAT was used to estimate health impacts resulting from changes in ambient concentrations of $PM_{2.5}$, O_3, and NO_2 (µg/m^3 or ppb; Equation 1) between the base case and each emissions reduction scenario. A positive change in concentration (Equation 1) indicates an improvement relative to the base case, or a reduction in exposure, and thus a positive health impact or benefit:

$$\Delta\bar{C} = \bar{C}_{base\ case} - \bar{C}_{scenario} \qquad \text{Equation 1}$$

where:

$\Delta\bar{C}$ is the change in ambient pollutant concentration ($PM_{2.5}$, O_3, or NO_2) between the base case and the emissions reduction scenario in a census division (µg/m^3 for $PM_{2.5}$ and ppb for O_3 and NO_2);

$\bar{C}_{base\ case}$ is the ambient concentration of a specific pollutant ($PM_{2.5}$, O_3, or NO_2) in a census division in the base case scenario (µg/m^3 or ppb); and

$\bar{C}_{scenario}$ is the ambient concentration of a specific pollutant ($PM_{2.5}$, O_3, or NO_2) in a census division in a scenario where emissions from one pollutant-sector-region are reduced by 10% from the base case (µg/m^3 or ppb).

Gridded, hourly GEM-MACH concentration outputs from step 2 were post-processed by ECCC to estimate $\bar{C}_{base\ case}$ and $\bar{C}_{scenario}$ (defined in Equations 2 and 3). GEM-MACH concentrations were first **temporally averaged** for use in AQBAT, for consistency with averaging periods used in epidemiological studies. Averaging periods and the units of the resulting ambient concentration estimates are listed below:

- For $PM_{2.5}$ (µg/m^3) and NO_2 (ppb):
    - Annual average of all hourly concentrations (equivalent to annual average of 24-hr averages)
- For O_3 (ppb):
    - Annual average of all daily 1-hr maximum concentrations (from January to December, inclusive of summer months)
    - Summer average of daily 1-hr maximum concentrations from May to September

For further discussion on the use of averaging periods to estimate health impacts, refer to the "Health impacts estimation" section below.

Temporally averaged, gridded GEM-MACH concentrations were then **spatially processed** into CD-average estimates for use in AQBAT. These CD-average concentration estimates represent area-weighted averages of the temporally averaged, gridded GEM-MACH concentrations, and were interpolated using a normalized conservative interpolation to estimate average concentration values over the area occupied by the CD, according to Equations 2 and 3:

$$\bar{C}_{base\ case} = \frac{\sum(C_{base\ case} \times w)}{\sum w} \qquad Equation\ 2$$

$$\bar{C}_{scenario} = \frac{\sum(C_{scenario} \times w)}{\sum w} \qquad Equation\ 3$$

where:

C is the ambient pollutant concentration ($PM_{2.5}$, NO_2, or O_3) in a given GEM-MACH grid cell, under the base case ($C_{base\ case}$) or the pollutant-sector-region emissions reduction scenario ($C_{scenario}$) (µg/m^3 or ppb);
w is the surface area of the grid cell overlying the census division (unitless); and
$\bar{C}$ is the area-weighted average ambient pollutant concentration in the census division, under the base case ($\bar{C}_{base\ case}$) or the pollutant-sector-region emissions reduction scenario ($\bar{C}_{scenario}$) (µg/m^3 or ppb).

The resulting CD-level concentrations, $\bar{C}_{base\ case}$ and $\bar{C}_{scenario}$, are spatially averaged estimates and thus do not reflect local effects nor microenvironments. Note that CD-level concentration changes ($\Delta\bar{C}$ in Equation 1) that were < 0.005 ug/m^3 for $PM_{2.5}$ or < 0.005 ppb for O_3 and NO_2 were set to zero by ECCC, as these changes were not considered to be significant and are subject to uncertainty. As a result, health impacts used to derive BPTs do not include impacts resulting from air quality changes smaller than this. Overall, this results in a likely underestimation of BPTs.

Population

Age-specific, CD-level population estimates were obtained from Statistics Canada and are included in AQBAT 3.0. Populations were projected from the 2011 Census of Population (Statistics Canada 2012) to the year 2015 using a medium growth projection.

Concentration-response functions

Concentration-response functions or CRFs mathematically describe the relationship between exposure to an air pollutant and the risk of an adverse health endpoint. The health endpoints included in this analysis, and the CRFs describing the exposure-response curves for each pollutant, are listed in Table 2. These CRFs have log-linear or purely linear forms and apply to the entire population or a specified segment of the population (e.g., asthmatics vs. non-asthmatics or adults vs. children). Health impacts for each endpoint were estimated for the segment of population that is considered at risk, which may represent a proportion of the general population, as listed in Table 2. Further details on the studies used to derive these CRFs, including confounding by co-pollutants, can be found in Appendix A. Note that AQBAT accounts for overlapping and related health endpoints to avoid double-counting.

Table 2. AQBAT 3.0 concentration-response functions (CRFs) used to estimate health impacts included in BPTs.

Pollutant	Health endpoint[a]	Averaging period[b]	Population[c]	CRF form[d,e]	Mean β (per ppb or per ug/m³)[f]	β standard error (per ppb or per ug/m³)[f]	Source
Mortality							
$PM_{2.5}$	Chronic exposure mortality	24 hr	All members ≥ 25 years	Log-linear	0.00953	0.00232	Crouse et al. 2012
NO_2	Acute exposure mortality	24 hr	All members of all ages	Log-linear	0.000748	0.000249	Burnett et al. 2004
O_3	Acute exposure mortality	1 hr	All members of all ages	Log-linear	0.000839	0.000136	Burnett et al. 2004
O_3	Chronic exposure respiratory mortality	1 hr (May–Sept)	All members ≥ 25 years	Log-linear	0.00392	0.00132	Jerrett et al. 2009
Morbidity							
$PM_{2.5}$	Respiratory emergency room visits	24 hr	All members of all ages	Linear	0.000754	0.000132	Burnett et al. 1995; Stieb et al. 2000
$PM_{2.5}$	Respiratory hospital admissions	24 hr	All members of all ages	Linear	0.000754	0.000132	Burnett et al. 1995
$PM_{2.5}$	Restricted activity days	24 hr	Adults and non-asthmatic children aged 5–19 years	Log-linear	0.00481	0.00101	Ostro 1987; Ostro and Rothschild 1989; Chestnut et al. 1999
$PM_{2.5}$	Child acute bronchitis episodes	24 hr	All children aged 5–19	Log-linear	0.00893	0.00575	Hoek et al. 2012; Dockery et al. 1996
$PM_{2.5}$	Cardiac emergency room visits	24 hr	All members of all ages	Linear	0.000711	0.000170	Burnett et al. 1995; Stieb et al. 2000

$PM_{2.5}$	Cardiac hospital admissions	24 hr	All members of all ages	Linear	0.000711	0.000170	Burnett et al. 1995
$PM_{2.5}$	Asthma symptom days	24 hr	Asthmatic children aged 5–19	Log-linear	0.00655	0.00265	Mortimer et al. 2002; Weinmayr et al. 2010; Ward and Ayres 2004; PHAC 2018
$PM_{2.5}$	Adult chronic bronchitis cases	24 hr	All members ≥ 25 years	Log-linear	0.0132	0.00680	Abbey et al. 1995
$PM_{2.5}$	Acute respiratory symptom days	24 hr	Adults and non-asthmatic children aged 5–19 years	Linear	0.00266	0.00139	Krupnick et al. 1990
O_3	Respiratory emergency room visits	1 hr	All members of all ages	Log-linear	0.000791	0.000355	Burnett et al. 1997; Stieb et al. 2000
O_3	Respiratory hospital admissions	1 hr	All members of all ages	Log-linear	0.000791	0.000355	Burnett et al. 1997
O_3	Acute respiratory symptom days	1 hr	Adults and non-asthmatic children aged 5–19 years	Linear	0.000786	0.000386	Krupnick et al. 1990
O_3	Asthma symptom days	1 hr	Asthmatic children aged 5–19	Log-linear	0.00238	0.000219	Mortimer et al. 2002; Schildcrout et al. 2006
O_3	Minor restricted activity days	1 hr	Adults and non-asthmatic children aged 5–19 years	Log-linear	0.000530	0.00291	Ostro and Rothschild 1989

[a] Chronic exposure mortality refers to chronic exposure non-accidental mortality. AQBAT 3.0 includes the option of modelling chronic exposure mortality from all internal causes or specific causes (chronic obstructive pulmonary disease, ischemic heart disease, lung cancer, stroke) associated with $PM_{2.5}$ exposure (the former approach was taken here). The log-linear CRF from Crouse et al. (2012) was used in this analysis as it was selected for AQBAT by expert elicitation.
[b] 1-hr averaging period for O_3 refers to the daily maximum 1-hr average concentration.

[c] Adults are defined in AQBAT as persons > 19 years of age.

[d] The form of CRF refers to the statistical regression type in the epidemiological study, and mathematically describes the relationship between exposure and risk. Log-linear refers to a CRF of the form LogRR = βΔC or LogHR = βΔC or LogOR = βΔC, whereas a linear form of CRF refers to the form RR = βΔC, where RR refers to relative risk, HR refers to the hazard ratio, OR refers to the odds ratio, and C refers to the concentration of ambient pollutant ($PM_{2.5}$ [μg/m^3], O_3, or NO_2 [ppb]). β is defined as below, in footnote f. RRs are typically derived from Poisson or similar log-linear regression models employed in time-series studies of day-to-day variability in air pollution exposure and health outcomes. HRs are derived from Cox proportional hazard or similar models applied to cohort studies of long-term exposure. ORs may be derived from logistic regression models employed in cross-sectional (exposure and outcome at single point in time), case-control (contrasting exposure in those with vs. without disease) or case-crossover (contrasting exposure in same individual on event vs. non-event days) studies.

[e] Log transformations use base "e."

[f] β represents the risk per unit increase in exposure (i.e., per μg/m^3 for $PM_{2.5}$ and per ppb for NO_2 and O_3).

Baseline health endpoint rates

Baseline health endpoint rates refer to reference, morbidity, and mortality incidence rates in the population. They are expressed as the number of health endpoint incidences per year, per 1,000,000 specified population. Baseline health endpoint rates for 2015 were assigned in AQBAT to CDs across Canada to calculate health impacts. For some health endpoints (acute respiratory symptom days, asthma symptom days, child acute bronchitis episodes, adult chronic bronchitis cases, restricted activity days, and minor restricted activity days), a single baseline rate is applied across the country in the absence of geographically resolved data. A summary table of baseline health endpoint rates used in this analysis is provided in Appendix B. Further discussion of baseline health endpoint rates can be found in previous Health Canada reports (Health Canada 2021; Judek et al. 2019). Details regarding data sources (e.g., death certificates, hospital admission records provided by Statistics Canada or epidemiological studies) and algorithms used within AQBAT to estimate annual baseline rates across the population can also be found in the AQBAT 3.0 User Guide (Judek et al. 2019).

Economic valuation

Health endpoints are routinely monetized to convert morbidity and premature mortality outcomes into economic welfare values. This step allows for comparisons with the costs of air quality improvements. The majority of monetized benefits in these BPT estimates are associated with mortality risk reductions. Health Canada uses a mortality risk valuation of $65 (2007$) for each 1 in 100,000 reduction in risk (Chestnut and De Civita 2009) and can be interpreted as society's willingness to pay (WTP) to avoid the risk of premature death. Multiplying the $65 by a population of 100,000 gives the value of one statistical life (VSL) equal to $6.5 million. Inflated into 2015 dollars, the VSL used in this analysis is $7.4 million. Valuation estimates for non-fatal health endpoints take into account treatment costs borne by the individual and the health care system, lost productivity, and pain and suffering. BPT estimates derived in this analysis should not be interpreted solely as costs to the health care system, but rather as society's WTP to avoid adverse health effects associated with a one tonne reduction in air pollutant emissions.

A summary of the economic valuations applied to morbidity and mortality endpoints in AQBAT 3.0 is presented in Table 3.

Table 3. Representative estimates of monetized valuations for mortality and morbidity in AQBAT 3.0. Estimates are provided in CAD and for the currency year 2015 (2015 CAD).

Health Endpoint	Valuation, $ per case (2015 CAD)[a,b,c]	Source [year of original estimate]
Premature mortality[d]	7,400,000	Chestnut 2009 [2007]
Acute respiratory symptom days	18	Stieb et al. 2002 [1997]
Adult chronic bronchitis cases	430,000	Krupnick and Cropper 1992; Viscusi et al. 1991 [1996]
Asthma symptom days	72	Stieb et al. 2002 [1997]
Cardiac emergency room visits	6,200	Stieb et al. 2002 [1997]
Cardiac hospital admissions	–[e]	Stieb et al. 2002 [1997]
Child acute bronchitis episodes	440	Krupnick and Cropper 1989 [1996]
Minor restricted activity days	31	Stieb et al. 2002 [1997]
Respiratory emergency room visits	2,800	Stieb et al. 2002 [1997]
Respiratory hospital admissions	–[e]	Stieb et al. 2002 [1997]
Restricted activity days	67	Stieb et al. 2002 [1997]

[a] These values are economic welfare values. AQBAT provides economic valuation estimates that consider the potential welfare consequences, including treatment costs, lost productivity, pain and suffering, and the impacts of increased mortality risk. For a detailed explanation of these values see the AQBAT 3.0 User Guide (Judek et al. 2019).

[b] Valuations applied within AQBAT follow normal or discrete distributions rather than single estimates. For simplicity, valuations reported here represent 1) the mean for normal distributions, or 2) probability-weighted estimates for discrete distributions. Parameters describing the full distributions can be found in the AQBAT 3.0 User Guide (Judek et al. 2019).

[c] Estimates are adjusted for Inflation using the Consumer Price Index (CPI; Statistics Canada 2022), which is applied to the source year of the original estimate (1996, 1997, or 2007; third column). CPI conversions used were 1.42 for 1996 to 2015; 1.40 for 1997 to 2015; and 1.14 for 2007 to 2015. No adjustment was made for income growth.

[d] Premature mortality includes chronic exposure non-accidental mortality, acute exposure mortality, and chronic exposure respiratory mortality.

[e] Hospital admission valuation is included for the proportion of ED visits resulting in admission to hospital.

Health impacts estimation

Health impacts are estimated in AQBAT via an impact function, which relies upon economic valuation, baseline health endpoint rates, population, the CRF, and the change in ambient concentration (estimated via Equation 1). This function is shown in Equation 4 and quantifies health impacts due to a **single health endpoint and ambient pollutant pair** (e.g., chronic exposure non-accidental mortality associated with $PM_{2.5}$ exposure). A positive value of health impacts, ΔHI, in Equation 4 indicates a health benefit (or improvement):

$$\Delta HI = EV \times M_0 \times P\left(e^{\beta \Delta \bar{C}} - 1\right) \quad \text{Equation 4}$$

where:

ΔHI is the change in health impacts between the base case and the pollutant-sector-region emissions reduction scenario, for a health endpoint and ambient pollutant pair, expressed as an economic value ($ per year);

EV is the economic valuation for the health endpoint ($);

M_0 is the baseline health endpoint rate in a census division (events per million population, per year);

P is the specified population for the health endpoint and pollutant pair in a census division (e.g., all ages or a specific age group; millions);

β is the risk estimate for the health endpoint and pollutant pair from the epidemiological study (per µg/m^3 for $PM_{2.5}$ and per ppb for O_3 and NO_2); and

$\Delta\bar{C}$ is the change in ambient pollutant concentration of $PM_{2.5}$, O_3, or NO_2 between the base case and the pollutant-sector-region emissions reduction scenario in a census division (µg/m^3 for $PM_{2.5}$ and ppb for O_3 and NO_2; Equation 1).

A simplifying assumption made in AQBAT is the use of long-term average concentrations to estimate acute health impacts. For $PM_{2.5}$ and NO_2, changes in the annual average concentration are assumed to equal the average change in 24-hr average concentrations over the year. Similarly, changes in annual (or seasonal) average daily 1-hr maximum O_3 are assumed to equal the average change in daily 1-hr maximum O_3 over the year (or season). This approximation is accurate provided that the CRF is linear or near-linear (i.e., risk changes linearly or almost linearly with concentration), and that the CRF follows a non-threshold response. Baseline health endpoint rates for acute health endpoints are thus expressed on an annual basis.

For each pollutant-sector-region emissions reduction scenario, the total health impacts resulting from that scenario are estimated by summing Equation 4 across all health endpoint and ambient pollutant pairs, and across all CDs in Canada, to estimate the totality of health impacts in Canada. Equation 4 can also be summed over a smaller number of CDs that, for example, are within SWBC or the WQCC only, to provide an estimate of regional health impacts occurring closer to the emissions reduction. For further discussion on this, refer to the section “Location of emissions reduction vs. location of health impacts” in step 4.

Equation 4 applies to pairs of pollutants and health endpoints in AQBAT that follow a log-linear form of concentration-response, as listed in Table 2 (such as mortality). Alternative formulations of Equation 4 that apply to health endpoints with other forms of CRFs (e.g., linear) can be found in the AQBAT 3.0 User Guide (Judek et al. 2019). Note that when the concentration change is negative, the formulation of Equation 4 is mathematically equivalent to the attributable fraction expressed as a percent, and the concentration change is entered into AQBAT as a positive number (Judek et al. 2019). It is noted that AQBAT accounts for the distribution of parameter estimates in Equation 4 in a Monte Carlo framework, and accounts for overlapping and related health endpoints to avoid double-counting.

Step 4. BPT formulation

Region-specific, sector-specific, and pollutant-specific BPTs were estimated in step 4 of the modelling, once the emissions modelling, GEM-MACH modelling, and AQBAT modelling were completed. BPTs were estimated via Equation 5, which divides the change in health impacts by the change in emissions, with that change referring to the difference between the base case and the emissions reduction scenario. Equation 5 is calculated for each scenario to estimate the pollutant-, sector-, and region-specific BPT:

$$BPT = \frac{\Delta HI}{\Delta E} \qquad \text{Equation 5}$$

where

BPT is the benefit per tonne of emissions reduction of a single pollutant, from a single sector, in a single region, with a positive value defined as an improvement in health ($ per tonne);

ΔHI is the change in total health impacts between the base case and the pollutant-sector-region emissions reduction scenario, summed over all health endpoint and ambient pollutant ($PM_{2.5}$, O_3, and NO_2) pairs, summed over all CDs in Canada or over CDs closer to the emission reduction region ($ per year); and

ΔE is the change in emissions of a single emitted pollutant (primary $PM_{2.5}$, NO_x, SO_x, VOC, or NH_3) between the base case and the pollutant-sector-region emissions reduction scenario (tonne per year).

As described for Equation 4, the change in total health impacts in Equation 5, ΔHI, is estimated for all health endpoints related to $PM_{2.5}$, O_3, and NO_2 exposure listed in Table 2. Changes in health impacts are summed over these three ambient pollutants in order to provide a measure of the totality of impact on population health in Canada. Health impacts are also reported separately as BPTs for each ambient pollutant pathway (i.e., BPTs resulting from changes in ambient $PM_{2.5}$ concentrations vs. ambient O_3 concentrations vs. ambient NO_2 concentrations), in Figures 3–7 in the "Results" section. The change in emissions in Equation 5, ΔE, is determined by the emissions modelling in step 1 and represents the difference between the base case and the pollutant-, sector-, and region-specific emissions reduction scenario. For the emissions reduction scenarios, a perturbation size of −10% was used, as smaller perturbations may yield numerical errors (particularly for sectors with smaller emissions quantities), and perturbations larger in magnitude are more likely to encounter nonlinearity in atmospheric responses of ambient pollutant concentrations to emissions. The choice of a −10% emissions change is discussed further in the "Limitations of Health Canada's BPT estimates" section.

It is noted that the magnitude of the emissions change is important, in addition to the magnitude of the BPT estimate in Equation 5, for evaluating the health impacts of emission mitigation options.

Emitted pollutants vs. ambient pollutant concentrations

For a discussion of the difference between emitted pollutants and ambient pollutant concentrations, users are referred to the section "Concentration change" in step 3.

BPT estimates are provided in this report as 1) total health impacts due to exposure to all three ambient pollutants ($PM_{2.5}$, O_3, and NO_2), and 2) separately, by ambient pollutant ($PM_{2.5}$, O_3 [annual, May–September], and NO_2). For example, NO_x BPTs are reported as a total impact (total $ per tonne), as well as for $PM_{2.5}$, O_3 (annual and May–September) and NO_2 health impacts separately (i.e., $ per tonne from changes in ambient $PM_{2.5}$ vs. O_3 vs. NO_2 concentrations).

Location of emissions reduction vs. location of health impacts

Air pollutants emitted from a source are transported such that a source may affect ambient concentrations nearby and/or further downwind. The range of influence depends on the lifetime of the pollutant, meteorology, topography, and other factors. Health impacts depend largely on the geographic distribution of populations relative to the source.

The distinction between the **location of emissions reduction** and the **location of health impacts** is important. BPTs in this analysis represent the impacts of a **regional** emissions reduction (in SWBC or the WQCC) on health both within and outside of the region (for all CDs in Canada). For users seeking more resolved information, BPTs encompassing health impacts within the region only (i.e., SWBC or the WQCC) are reported and are referred to as “within-region BPTs”. BPTs encompassing health impacts within the province only (i.e., British Columbia, Ontario, or Quebec) are also reported and are referred to as “within-province BPTs”. These BPTs represent regionally and provincially aggregated health impacts and were extracted from the full set of Canada-wide impacts.

Results

BPT estimates – total health impacts

Table 4 lists the pollutant-, sector-, and region-specific BPTs for SWBC and the WQCC by source sector and emitted pollutant. BPTs represent the monetized health benefit per one tonne of emissions reduction of a specific pollutant ($PM_{2.5}$, NO_x, VOC, NH_3, or SO_x) from a specific source sector (on-road, off-road, manufacturing and ore and mineral industries, or agriculture) in either SWBC or the WQCC region. BPTs are reported as economic values in Table 4 rather than as the number of avoided health endpoints, as is standard practice for reporting health impacts on a per-tonne of emissions basis.

Table 4. BPT estimates for southwestern British Columbia (SWBC) and the Windsor–Quebec City corridor (WQCC), by source sector and emitted pollutant.

Emitted pollutant[a,b]	Source sector	BPT ($/tonne)[c,d]	
		SWBC	WQCC
Primary $PM_{2.5}$	On-road mobile	410,000	520,000
	Off-road mobile	470,000	480,000
	Manufacturing + Ore and mineral industries	340,000	380,000
NO_x	On-road mobile	-140[e]	15,000
	Off-road mobile	-2,700[e]	12,000
	Manufacturing + Ore and mineral industries	-3,900[e]	4,900
VOC	On-road mobile	13,000	3,900
	Off-road mobile	9,900	5,100
	Manufacturing + Ore and mineral industries	3,900	2,300
NH_3	On-road mobile	100,000	130,000
	Agriculture[f]	46,000	26,000
SO_x	Manufacturing + Ore and mineral industries	–[g]	10,000

[a] Primary $PM_{2.5}$ emissions are the summation of directly emitted crustal material, primary organic carbon, elemental carbon, sulphate, nitrate, and ammonium.

[b] Emitted VOCs consist of alkenes, alkanes, acetaldehyde, aromatics, propane, O-creosol, ethene, formaldehyde, isoprene, methyl ethyl ketone, and toluene.

[c] BPTs reflect the marginal change in societal economic welfare attainable from reducing emissions of a pollutant by one tonne, and include direct, indirect and intangible costs such as pain and suffering. BPTs are reported in 2015 CAD per tonne of emission reduction.

[d] BPTs represent the summation of health impacts due to changes in ambient $PM_{2.5}$, O_3, and NO_2 concentrations. BPTs for each ambient pollutant are reported separately in Figures 3–7.

[e] Negative BPTs are due to NO_x titration and should be considered carefully. Users are referred to the "Limitations of Health Canada's BPT estimates" section for discussion.

[f] Emissions from agricultural equipment were included in the off-road mobile sector.

[g] BPT estimate could not be derived, as concentration differences resulting from SO_x emissions reduction scenario in SWBC resulted in ambient pollutant concentration differences < 0.005 (µg/m^3 for $PM_{2.5}$ or ppb for O_3/NO_2) at the census division (CD) level. ECCC set any concentration differences < 0.005 to zero prior to delivering the data to Health Canada, as these differences were not considered to be significant and are subject to uncertainty.

It is important to consider that while the magnitude of BPTs varies in Table 4 across emitted pollutants (i.e., primary $PM_{2.5}$, NO_x, SO_x, VOC, and NH_3), the magnitude of emissions of these pollutants also varies. Tables 5 and 6 list BPT estimates, along with the emissions changes and health impacts used to derive those BPTs. Emissions changes listed in Tables 5 and 6 represent 10% of the total emissions for the pollutant, sector, and region in question, equivalent to the emissions change between the base case and the pollutant-, sector-, and region-specific emissions reduction scenario, or ΔE. Health impacts (ΔHI) refer to the resulting change in health impacts between the base case and the pollutant-, sector-, and region-specific emissions reduction scenario. Health impacts are summed across all CDs in Canada and across all three ambient pollutants ($PM_{2.5}$, O_3, and NO_2) and health endpoints as described for Equation 4. Details on the derivation of these values can be found in Equations 1–5. As the goal of this analysis is to derive monetized health impacts on a unit emissions basis, changes in mortality and morbidity counts are not reported, and in many cases amount to less than one case. Overall, the monetary value of mortality constitutes 92% to 99% of these BPTs.

BPT estimates are reported as central values in Tables 5 and 6, along with 95% confidence intervals. These 2.5th and 97.5th percentiles are derived within AQBAT using Monte Carlo simulations of parameter distribution estimates for CRFs and the economic valuation. These uncertainty estimates do not account for uncertainty in health endpoint rates nor population estimates. Further, uncertainty in ambient concentration estimates from macro-economic, emissions, meteorological, and chemical transport modelling is not included in these confidence intervals. Uncertainty estimates are not routinely produced for GEM-MACH model outputs and were not produced for the scenarios modelled in this analysis.

Table 5. Emission changes, total health impacts, and BPTs with 95% confidence intervals for southwestern British Columbia (SWBC), by source sector and emitted pollutant.

Emitted pollutant[a, b]	Source sector	10% Emissions change, ΔE (tonne per yr)[c]	Total health impacts, ΔHI ($ millions per yr) [95% confidence interval][d,e]; 2015 CAD	BPT ($ per tonne) [95% confidence interval][e,f]; 2015 CAD
Primary $PM_{2.5}$	On-road mobile	110	44 [18–81]	410,000 [160,000–750,000]
	Off-road mobile	91	43 [18–79]	470,000 [200,000–870,000]
	Manufacturing + Ore and mineral industries	180	59 [23–110]	340,000 [130,000–620,000]
NO_x	On-road mobile	3,400	(-0.49) [(-18)–20][g]	(-140) [(-5,400)–5,900][g]
	Off-road mobile	850	(-2.3) [(-8.6)–4.4][g]	(-2,700) [(-10,000)–5,100][g]
	Manufacturing + Ore and mineral industries	930	(-3.6) [(-7.8)–(-0.097)][g]	(-3,900) [(-8,500)–(-100)][g]
VOC	On-road mobile	1,200	16 [9.3–24]	13,000 [7,600–20,000]
	Off-road mobile	630	6.2 [3.0–10]	9,900 [4,800–16,000]
	Manufacturing + Ore and mineral industries	480	1.9 [0.90–3.1]	3,900 [1,900–6,500]
NH_3	On-road mobile	54	5.6 [2.3–10]	100,000 [43,000–190,000]
	Agriculture	600	28 [11–52]	46,000 [19,000–87,000]

[a] Primary $PM_{2.5}$ emissions are the summation of directly emitted crustal material, primary organic carbon, elemental carbon, sulphate, nitrate, and ammonium.

[b] Emitted VOCs consist of alkenes, alkanes, acetaldehyde, aromatics, propane, O-creosol, ethene, formaldehyde, isoprene, methyl ethyl ketone, and toluene.

[c] Emissions change represents the difference between the base case and the 10% emissions reduction scenario, where one emitted pollutant is reduced from one sector in SWBC. No emissions changes were imposed outside of SWBC.

[d] Total health impacts are due to changes in ambient $PM_{2.5}$, O_3, and NO_2 concentrations and are summed across 293 CDs in Canada to represent the totality of impacts on health in Canada.

[e] 95% confidence intervals reflect the 2.5th and 97.5th percentiles in health impact estimates. Uncertainty resulting from the emissions and air quality modelling are not reported.

[f] BPTs reflect the marginal change in societal economic welfare attainable from reducing emissions of a pollutant by one tonne, and include direct, indirect and intangible costs such as pain and suffering. BPTs are reported in 2015 CAD per tonne.

[g] Negative health impacts and BPTs are due to NO_x titration and should be considered carefully. Users are referred to the "Limitations of Health Canada's BPT estimates" section for discussion.

Table 6. Emission changes, total health impacts, and BPTs with 95% confidence intervals for the Windsor–Quebec City Corridor (WQCC), by source sector and emitted pollutant.

Emitted pollutant[a,b]	Source sector	10% Emissions change, ΔE (tonne per yr)[c]	Total health impacts, ΔHI ($ millions per yr) [95% confidence interval][d,e]; 2015 CAD	BPT ($ per tonne) [95% confidence interval][e,f]; 2015 CAD
Primary $PM_{2.5}$	On-road mobile	490	260 [100–470]	520,000 [210,000–960,000]
	Off-road mobile	640	310 [120–570]	480,000 [190,000–900,000]
	Manufacturing + Ore and mineral industries	1,500	590 [240–1,100]	380,000 [160,000–720,000]
NO_x	On-road mobile	14,000	200 [86–350]	15,000 [6,300–26,000]
	Off-road mobile	6,300	73 [29–140]	12,000 [4,700–22,000]
	Manufacturing + Ore and mineral industries	4,800	24 [8.3–45]	4,900 [1,700–9,300]
VOC	On-road mobile	4,200	16 [8.0–26]	3,900 [1,900–6,300]
	Off-road mobile	4,600	23 [11–38]	5,100 [2,400–8,200]
	Manufacturing + Ore and mineral industries	4,600	11 [5.2–17]	2,300 [1,100–3,700]
NH_3	On-road mobile	310	39 [16–73]	130,000 [52,000–240,000]
	Agriculture	13,000	330 [130–610]	26,000 [10,000–48,000]
SO_x	Manufacturing + Ore and mineral industries	7,400	74 [29–140]	10,000 [4,000–19,000]

[a] Primary $PM_{2.5}$ emissions are the summation of directly emitted crustal material, primary organic carbon, elemental carbon, sulphate, nitrate, and ammonium.

[b] Emitted VOCs consist of alkenes, alkanes, acetaldehyde, aromatics, propane, O-creosol, ethene, formaldehyde, isoprene, methyl ethyl ketone, and toluene.

[c] Emissions change represents the difference between the base case and the 10% emissions reduction scenario, where one emitted pollutant is reduced from one sector in the WQCC. No emissions changes were imposed outside of the WQCC.

[d] Total health impacts are due to changes in ambient $PM_{2.5}$, O_3, and NO_2 concentrations and are summed across 293 CDs in Canada to represent the totality of impacts on health in Canada.

[e] 95% confidence intervals reflect the 2.5th and 97.5th percentiles in health impact estimates. Uncertainty resulting from the emissions and air quality modelling are not reported.

[f] BPTs reflect the marginal change in societal economic welfare attainable from reducing emissions of a pollutant by one tonne, and include direct, indirect and intangible costs such as pain and suffering. BPTs are reported in 2015 CAD per tonne.

Tables 4–6 reveal that BPTs are highly variable across emitted pollutants, and variable across regions, ranging from -\$3,900 per tonne of NO_x to \$470,000 per tonne of $PM_{2.5}$ emissions reduction in SWBC. In the WQCC, BPTs range from \$2,300 per tonne of VOCs to \$520,000 per tonne of $PM_{2.5}$ emissions reduction. BPTs are generally within the same order of magnitude between SWBC and the WQCC for primary emitted $PM_{2.5}$. BPTs for gas-phase precursors NO_x, SO_x, NH_3, and VOCs differ more widely between the regions. This is due to the dependency of secondary pollutant formation on the atmospheric mix of pollutants, which varies between SWBC and the WQCC and affects how a one tonne reduction in precursor emissions translates into changes in ambient concentrations of $PM_{2.5}$, O_3, and/or NO_2.

Overall negative BPTs exist for the sectors analyzed in SWBC due to NO_x titration, where a reduction in NO_x emissions leads to an increase in O_3, most likely to occur in dense urban areas. Ambient O_3 is chemically produced and its concentration depends largely on the ratio of NO_x to VOCs in the atmospheric mix. In dense urban areas, NO_x titration may occur when this ratio is high, a condition referred to as a NO_x-inhibited or VOC-limited condition. While localized increases in urban O_3 may happen, urban plumes react chemically as they move downwind of the urban core, resulting in a decreasing ratio of NO_x:VOCs and an eventual reduction in O_3 downwind (mainly in suburban and rural areas). As the WQCC has many population centres downwind of urban cores, these eventual reductions in O_3 offset the negative urban core impacts. In addition, the chemical mixture in SWBC differs from that of the WQCC, in part due to the role of natural VOC emissions, leading to different BPT estimates. VOC BPTs are consistently larger for SWBC and likely reflect VOC-limited chemistry in the region. Overall, BPTs are likely to be underestimated (or more negative for NO_x), as concentration differences ($\Delta\bar{C}$ in Equation 1) less than 0.005 (µg/m^3 for $PM_{2.5}$ or ppb for O_3/NO_2) were set to zero by ECCC as these changes were not considered to be significant and are subject to uncertainty. Health impacts associated with concentration changes smaller than this were zero as a result.

Health impacts (ΔHI), or the product of the BPT and change in emissions (ΔE) in Tables 5 and 6, vary by sector and region. For example, for the on-road sector in the WQCC, health impacts are similar in magnitude for 10% reductions in primary $PM_{2.5}$ and NO_x emissions (\$260 million per year for primary $PM_{2.5}$ emissions reductions and \$200 million per year for NO_x), despite vast differences in their BPTs. The variability in magnitudes of BPTs, emissions quantities, and health impacts in Tables 5 and 6 suggests that the health impacts of emission mitigation options can be highly specific to the pollutants, sectors, and regions in question.

BPT estimates – health impacts by region and ambient pollutant

The scenario-based approach used in the GEM-MACH and AQBAT modelling provides geographically resolved health impacts for all CDs across Canada. To capture the totality of health impacts, BPTs are reported in Tables 4–6 as the **total** health benefits, summed across all CDs in the country, per tonne of **regional** emissions reduction in SWBC or the WQCC. Jurisdictions may, however, be interested in health benefits that occur within SWBC or the WQCC only, or at the provincial level. Figures 3–7 present BPTs that are due to health benefits occurring 1) within the same region as the emissions reduction only (i.e., within SWBC or the WQCC); 2) within the provinces of British Columbia, Ontario, or Quebec only; and

3) nationally (as reported in Tables 4–6). Definitions for these BPTs are provided below. BPTs are also reported for each ambient pollutant pathway (i.e., showing the influence of a one tonne emissions reduction on health impacts due to ambient $PM_{2.5}$, O_3, or NO_2, separately).

Within-region BPTs for a specific ambient pollutant ($PM_{2.5}$, annual O_3, May–September O_3, or NO_2) are equal to the health impacts due to that ambient pollutant ($PM_{2.5}$, annual O_3, May–September O_3, or NO_2), summed across all CDs within the BPT region (SWBC or the WQCC), divided by the emissions change for the scenario reported in Tables 5 and 6. In cases where a CD only partially overlaps the region, that CD's entire health impacts are included in the within-region BPT calculation. Health impacts occurring outside of the region's CDs are excluded.

Within-province BPTs are equal to the health impacts due to that ambient pollutant ($PM_{2.5}$, annual O_3, May–September O_3, or NO_2), summed across all CDs within the specified province (British Columbia for SWBC scenarios and Ontario/Quebec for WQCC scenarios), divided by the emissions change for the scenario.

Within-Canada BPTs are equal to the total health impacts for that ambient pollutant ($PM_{2.5}$, annual O_3, May–September O_3, or NO_2), summed across all CDs in Canada, divided by the emissions change for the scenario. These BPTs are equivalent to those reported in Tables 4–6.

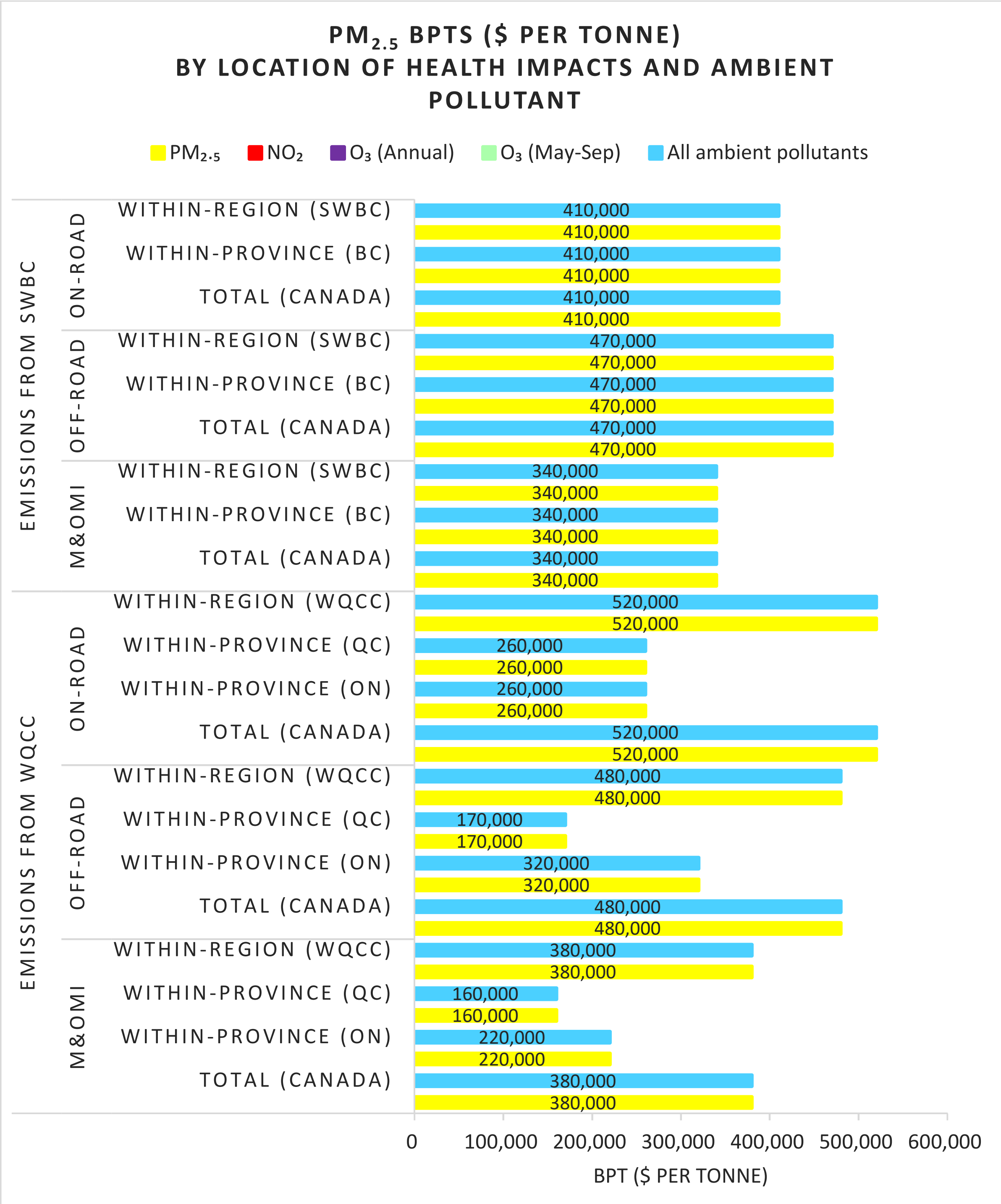

Figure 3. Within-region, within-province, and total BPTs for $PM_{2.5}$ emissions reduction, by ambient pollutant ($PM_{2.5}$, O_3, and NO_2) pathway.

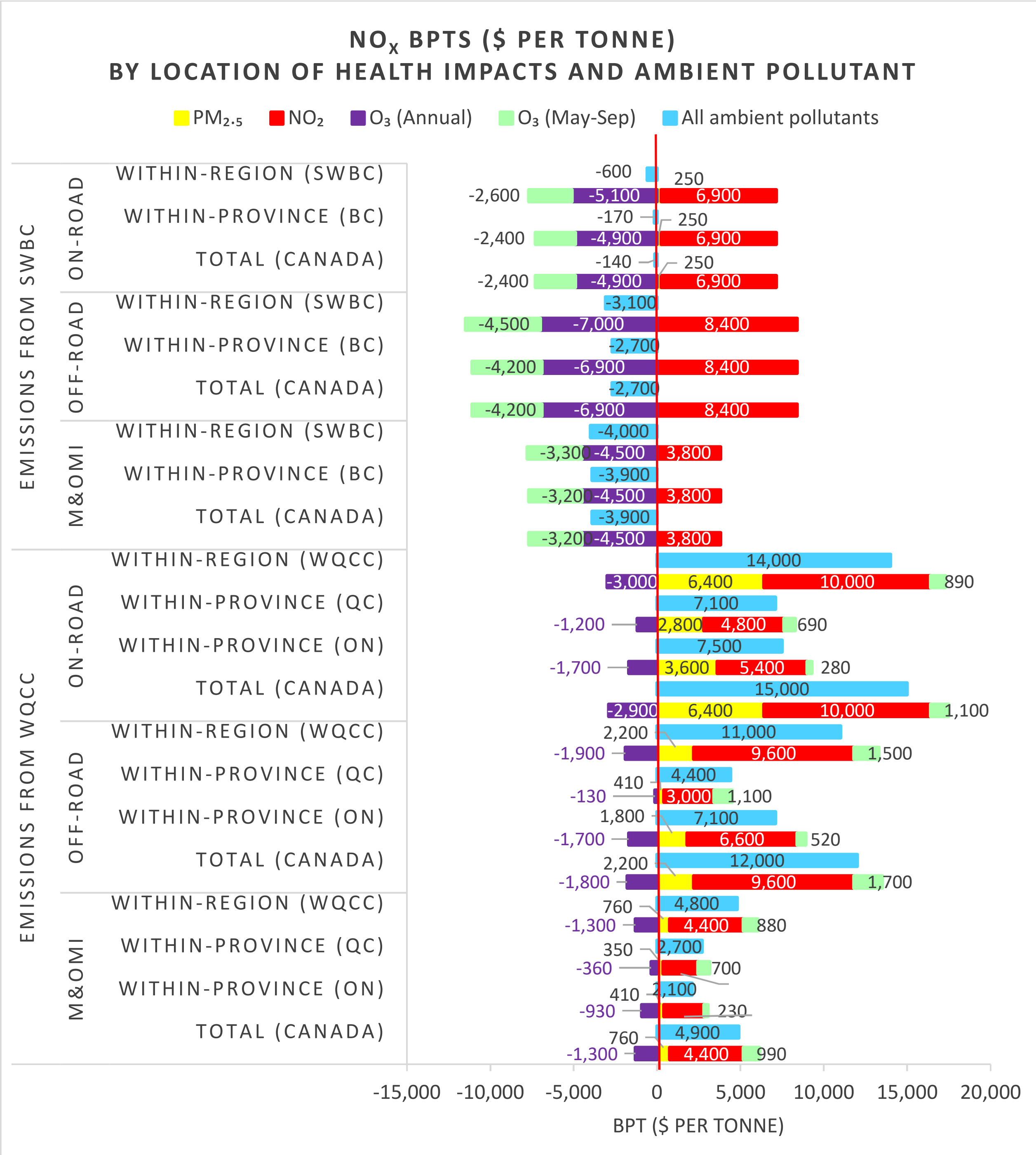

Figure 4. Within-region, within-province, and total BPTs for NO_x emissions reduction, by ambient pollutant ($PM_{2.5}$, O_3, and NO_2) pathway.

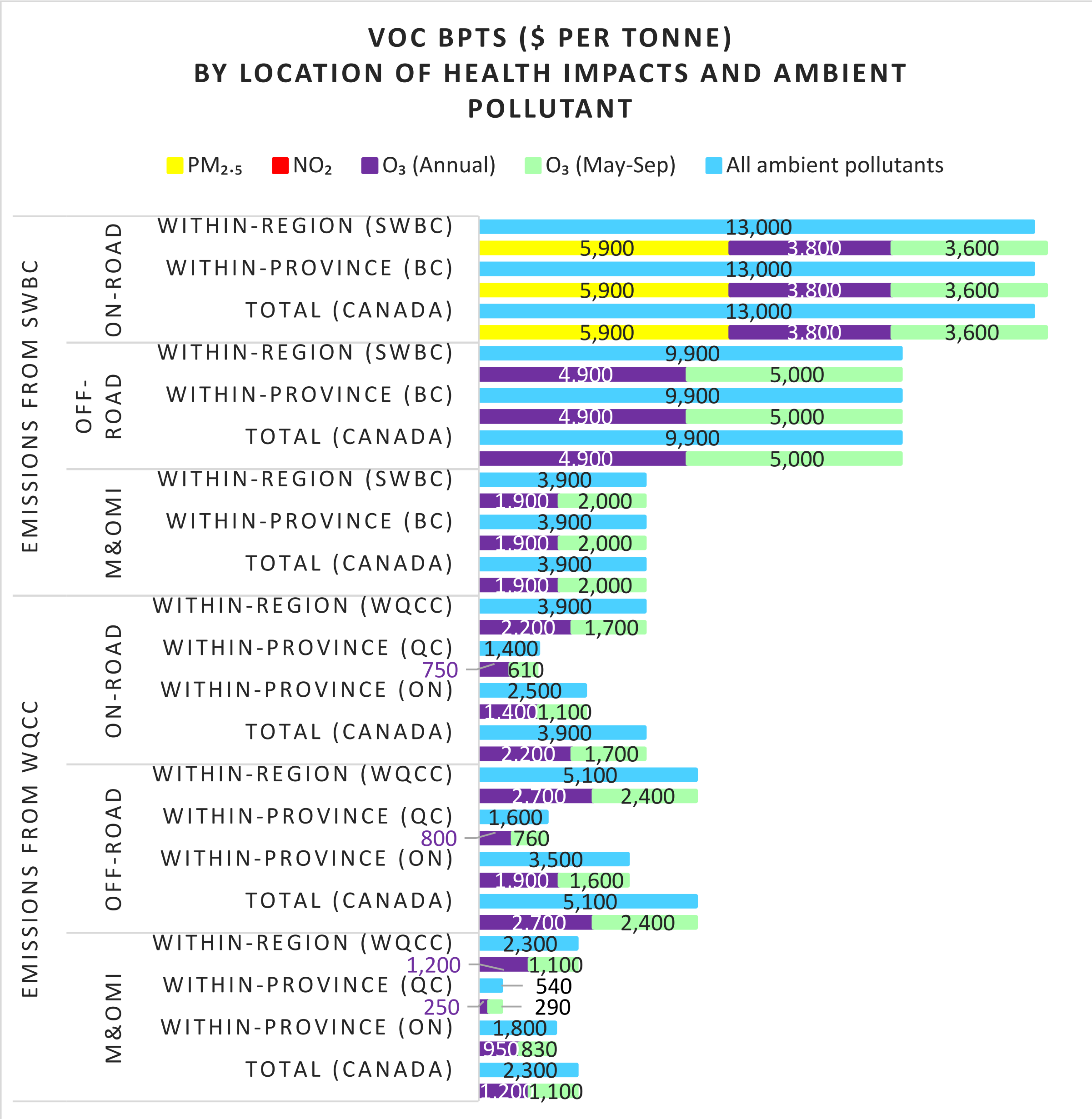

Figure 5. Within-region, within-province, and total BPTs for VOC emissions reduction, by ambient pollutant ($PM_{2.5}$, O_3, and NO_2) pathway.

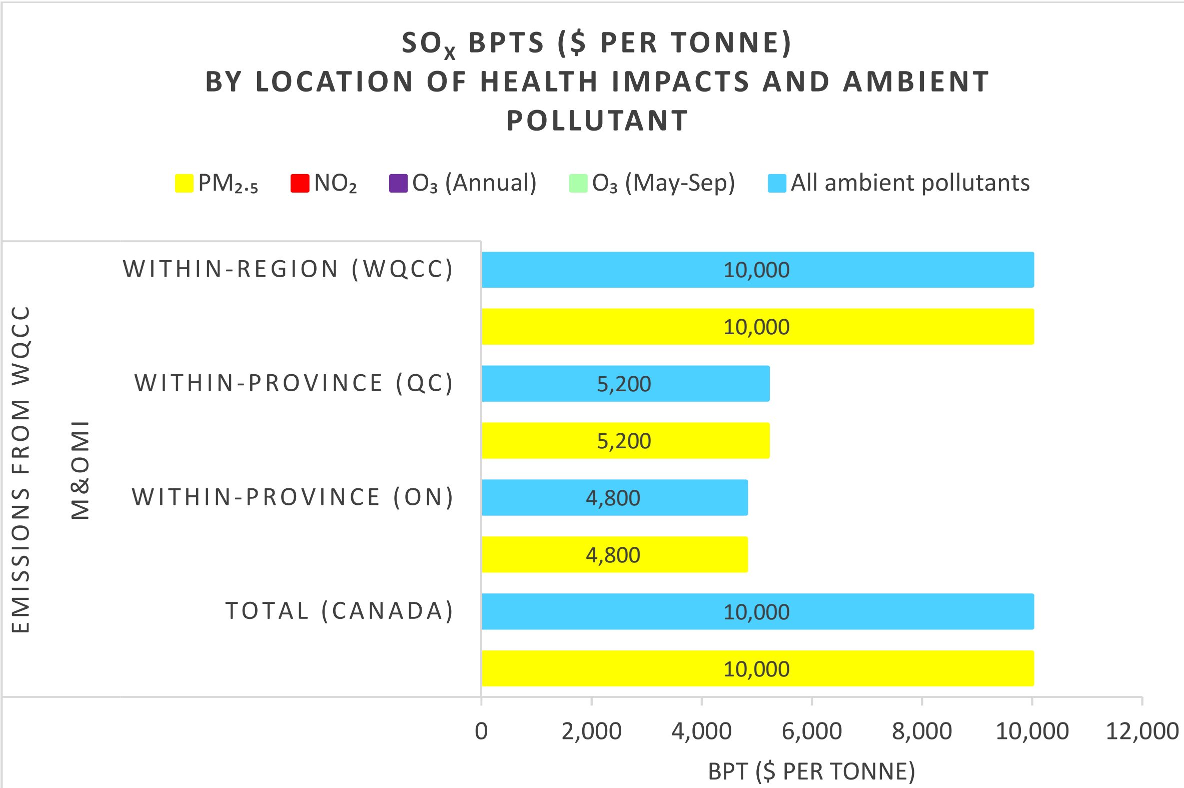

Figure 6. Within-region, within-province, and total BPTs for SO_x emissions reduction, by ambient pollutant ($PM_{2.5}$, O_3, and NO_2) pathway. SO_x BPTs could not be derived for manufacturing and ore and mineral industries in SWBC as the concentration differences ($\Delta\bar{C}$ in Equation 1) for all CDs were < 0.005 ($\mu g/m^3$ for $PM_{2.5}$ or ppb for O_3/NO_2) for the emissions reduction scenario.

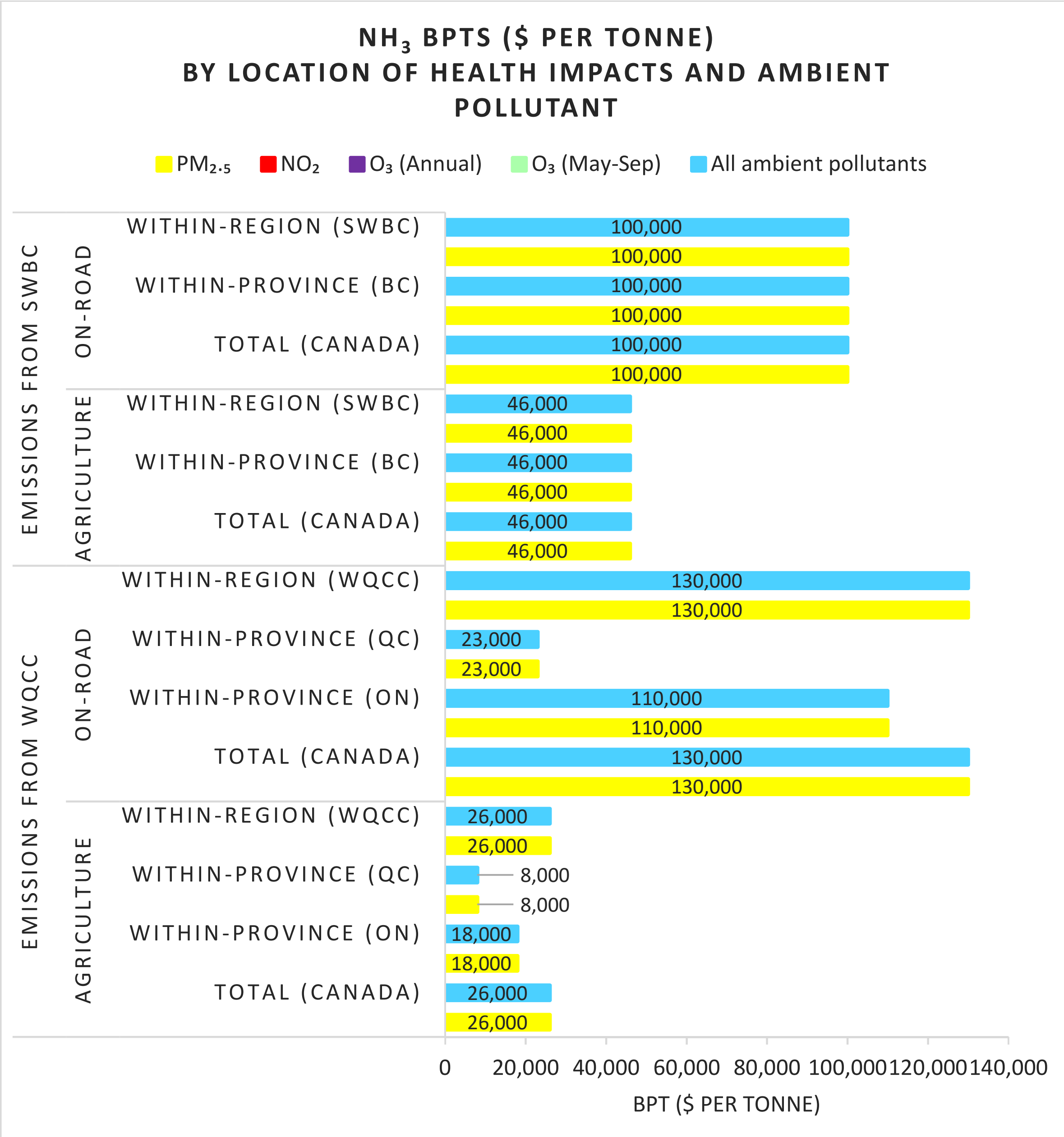

Figure 7. Within-region, within-province, and total BPTs for NH_3 emissions reduction, by ambient pollutant ($PM_{2.5}$, O_3, and NO_2) pathway.

Analysis of Figures 3-7 indicates that $PM_{2.5}$, SO_x, and NH_3 BPTs are due to reductions in ambient concentrations of $PM_{2.5}$ only and associated health impacts. In contrast, NO_x BPTs result from changes in ambient NO_2 and O_3 concentrations (and $PM_{2.5}$ for some sectors), and VOC BPTs result from reductions in ambient O_3 concentrations (and $PM_{2.5}$ for some sectors). For most NO_x and VOC BPTs, health impacts from reductions in ambient $PM_{2.5}$ were estimated to be zero, as the $PM_{2.5}$ concentration changes between the base case and the scenario were less than 0.005 μg/m^3 and were not considered to be significant by ECCC.

Further analysis of Figures 3–7 indicates that the vast majority of health benefits captured in the BPT estimates are due to health impacts occurring in the same region as the emission reduction (i.e., health impacts in SWBC or the WQCC; captured in the "within-region" BPTs). Out-of-region health impacts are zero or near-zero for $PM_{2.5}$, VOCs, SO_x, and NH_3 BPTs, indicating a strong localized impact of those emissions reductions on ambient $PM_{2.5}$ (and O_3 in the case of VOC BPTs). BPTs for NO_x are largely driven by within-region health impacts, though some O_3-related health impacts occur outside of SWBC and the WQCC, as O_3 is a more regional pollutant compared with NO_2 and $PM_{2.5}$. Within-region NO_x BPTs are generally smaller (or more negative for SWBC) than total BPTs due to downwind reductions in O_3 that occur outside of the region as a result of the NO_x emissions reduction, resulting in downwind health benefits. Within-Canada BPTs are often larger than within-region BPTs for NO_x, as within-region BPTs capture only near-range impacts, while within-Canada BPTs also capture impacts further from the source. The geographic distribution of populations in Canada, as well as the exclusion of concentration differences < 0.005 μg/m^3 ($PM_{2.5}$) or < 0.005 ppb (O_3/NO_2), likely contribute to the relatively small health impacts occurring outside of the regions studied.

Discussion

Key findings

A number of key findings are summarized below for the BPTs derived in this analysis. A discussion of limitations and uncertainties follows.

- **Primary $PM_{2.5}$** BPTs are the largest among the emitted pollutants modelled, ranging from \$340,000 to \$520,000 per tonne of primary $PM_{2.5}$ emissions reduction. This is due to the fact that $PM_{2.5}$ emission reductions directly reduce ambient $PM_{2.5}$ concentrations without having to undergo chemical and physical transformations in the atmosphere. Further, $PM_{2.5}$ is associated with the largest risks of mortality per unit of exposure among the three ambient pollutants ($PM_{2.5}$, O_3, and NO_2).
- BPTs for **gas-phase precursor pollutants** (NO_x, VOC, SO_x, NH_3) result to a large extent from their roles in the formation of secondary ambient pollutants such as O_3 or $PM_{2.5}$, and thus depend more strongly on weather conditions and the atmospheric mix of reactive pollutants.
- **NO_x** BPTs are generally the smallest in magnitude and are positive for sectors in the WQCC. NO_x BPTs for SWBC are overall negative due to increased ambient O_3 concentrations resulting from NO_x emissions reductions. Ambient NO_2/$PM_{2.5}$ health impacts from the same NO_x emissions reductions attenuate this negative impact from O_3. Negative BPTs for NO_x require careful consideration and are discussed further in points 3) and 5) of the "Limitations of Health

Canada's BPT estimates" section. Marginal NO_x emission reductions in the WQCC overall result in health benefits as NO_x titration in dense urban cores is balanced by positive health impacts occurring downwind.

- **NH_3** BPTs are positive and generally larger than those of NO_x, SO_x, and VOCs for the sectors and regions modelled. Health impacts of NH_3 emissions reductions are due to the role of NH_3 in the production of $PM_{2.5}$ ammonium nitrate and ammonium sulphate. NH_3 BPTs for agriculture (a distributed area source) are markedly lower than NH_3 from on-road vehicles, the latter of which is located in closer proximity to population centres and in proximity to vehicle NO_x emissions where NH_3 is more likely to be a limiting factor in secondary $PM_{2.5}$ formation.
- **SO_x** emission reductions result in health benefits due to reductions in ambient $PM_{2.5}$ concentrations and tend to be smaller in magnitude than NH_3 BPTs, but larger than NO_x. As SO_x emissions are primarily due to industrial emissions, SO_x BPTs are highly dependent upon the location of large sources in the regions.
- **VOC** BPTs largely reflect O_3 health impacts, and are smaller in magnitude than SO_x BPTs.

Limitations of Health Canada's BPT estimates

A number of limitations and uncertainties associated with the estimation of BPTs in this report are discussed below. Careful consideration of these limitations is required to assess the suitability of BPTs for estimating health impacts and to appropriately interpret estimates derived from these BPTs. For a discussion of situations where BPTs can be used, users are referred to the section "Applying BPTs for evaluating emission mitigation measures."

1. **BPT estimates apply to the regions, sectors, and pollutants modelled.** Each BPT estimate was derived from a scenario where emissions of a single pollutant from a single sector in a single region were reduced by 10% from the 2015 base case. As a result, BPTs are representative of the pollutant, sector, and region modelled. Emissions reductions were not applied nationally in the modelling, so BPTs cannot be applied to estimate health impacts of policies applied nationally. Sound judgement should be exercised when applying BPT estimates to sectors or subsectors that differ from those modelled, with consideration of differences in underlying emissions profiles and the geographic and temporal distributions of emissions. Refer to the Methods section for a listing of the subsectors considered within the broader "on-road," "off-road," "manufacturing and ore and mineral industries," and "agriculture" sectors. BPTs are location-specific and depend on factors such as atmospheric chemistry, the transport pathway, and proximity of populations to emissions sources. These BPTs should therefore not be applied to other regions of Canada, even those having similar population densities.
2. **BPTs are regional estimates.** BPTs in this analysis represent health impacts resulting from reductions in a hypothetical, regionally representative source in SWBC or the WQCC. BPTs represent these regions broadly and may not specifically reflect impacts from a reduction in emissions from any single source or from any particular location within the region. Intra-regional variability in BPTs may be significant (e.g., Pappin and Hakami 2013) and was not assessed. BPTs were derived using a 10% reduction in emissions applied to all sources within the specified region that emit the pollutant in question and belong to the source sector in question. While

this includes both larger and smaller sources distributed throughout the region, larger sources contribute a greater share of the overall health impacts used to estimate BPTs. BPTs are most applicable to regional-scale analyses and may be less representative of smaller geographic areas that are distant from where the majority of the region's air pollutant emissions are. For the on-road and off-road sectors, high population density areas are a major source of emissions, and as such, BPTs are influenced heavily by emissions in large urban population centres within the region. For the agricultural sector, BPTs reflect more distributed sources in rural areas of the region. For users wishing to evaluate the health impacts of emissions reduction policies for smaller geographic areas, such as for an individual municipality within the region, these regional BPTs are considered to be more relevant than national BPTs available elsewhere, but will have increased uncertainty. The level of uncertainty increases as the geographic area in question is located further away from the bulk of emissions in the region for the sector in question.

3. **BPTs are appropriate for relatively small emissions changes.** BPTs represent how ambient pollutant concentrations, and health impacts, respond to an emissions reduction of -10% from 2015 levels. Over larger changes in emissions, BPTs are expected to change with the level of emissions reduction implemented. The rate-of-change depends on the pollutant in question, the ambient mixture, weather and climate conditions, the relative importance of emissions sectors locally, the baseline level of emissions, and the exposed population. For example, the response of ambient O_3 concentrations to NO_x emissions is highly nonlinear across larger perturbations in emissions (e.g., 50%), particularly for urban areas undergoing shifts in atmospheric chemical composition (e.g., Hakami et al. 2003), implying a change in BPTs with emissions.

   An analysis of the rate of change of NO_x BPTs resulting from widespread, large NO_x emissions reductions across the US was conducted by Pappin et al. (2015). The authors ran the chemical transport model, CMAQ, and its full-form, adjoint or backwards sensitivity analysis tool for 2007 emissions levels. All pollutant emissions (including NO_x and VOCs) from mobile sources (including both on-road and off-road) vs. all point sources (industrial and electricity generating units, etc.) were reduced nationally in increments of 20%. NO_x BPTs generally increased slowly for the first 20%–40% reductions in emissions, with BPTs increasing more rapidly as emissions declined to zero; the rate of change in BPTs with emissions levels varied by city. Pappin et al. (2016), in a sensitivity analysis following Pappin et al. (2015), estimated the rate of change in NO_x BPTs when NO_x emissions alone were reduced in increments of 20% nationally. NO_x BPTs were found to rise more rapidly when only NO_x emissions were reduced compared to when all pollutants (including NO_x and VOCs) were reduced simultaneously. There is no universal level of NO_x emissions reduction that shifts NO_x BPTs from negative to positive (i.e., the transition between NO_x-inhibited/VOC-limited and NO_x-limited chemical regimes). The rate-of-change in BPTs with emissions levels varies by location (e.g., Pappin et al. 2015) and depends on the atmospheric mix of pollutants, topography, urban density, averaging period, etc. For other ambient pollutants, such as secondary $PM_{2.5}$, studies indicate nonlinearities in the response to changes in precursor pollutants (Koo et al. 2007; West et al. 1999; Vayenas et al. 2005). Nonlinearity in the CRF between $PM_{2.5}$ and mortality (e.g., Pappin et al. 2019; Burnett et al. 2018), though not included in the AQBAT modelling here, may add another layer of complexity.

There is no fixed rule as to the level of emissions reduction to which BPT estimates are applicable. Nonlinearity in the response of ambient pollutant concentrations (and health impacts) to emissions reductions becomes substantial when the state of the atmosphere and its composition changes. A shift in atmospheric composition is most likely to occur when a broad range of emissions reductions are applied to multiple sectors (as in the 2015 study by Pappin et al.). Changes to the atmospheric composition can also happen at the local scale when emissions from a single large source change significantly. Users of BPTs are advised to evaluate the size of the source in question for which health impacts are sought, in terms of its magnitude and spatial distribution, when evaluating whether BPTs are applicable. As a general rule, if the source in question constitutes less than 10% of the region's overall emissions and is ubiquitously distributed, then even a complete removal of that source's emissions would be unlikely to constitute a major change in the state of the atmosphere. In that case, the health impacts of an emission mitigation measure to reduce that source by any proportion – including complete removal of that source – would be reasonably quantified using Health Canada's BPT estimates. For the case of spatially heterogeneous sources whose emissions may have significant influences on the local atmospheric mixture (e.g., large industries), users are advised to apply BPTs for emissions reductions up to the quantities listed for that sector in Tables 5 and 6 (i.e., representing a 10% change of that source, rather than the regional emissions total). Regional emissions quantities for the SWBC and WQCC regions are listed in Table 1 for reference. For specific sub-sectors, such as ore and mineral industry sub-sectors that emit SO_x, recent emissions trends (2015–current) must be taken into account when assessing the 10% criteria if the analysis year is more recent than 2015.

4. **BPTs reflect annual health benefits resulting from annual changes in emissions.** BPTs were derived from a full-year chemical transport model simulation spanning all four seasons using 2019 meteorology. BPTs should be interpreted as the health benefits associated with reducing an emitted pollutant by one tonne over the year and not over shorter time frames, such as a single season or day. In reality, BPTs vary temporally with changes in photochemistry (e.g., the production of ambient O_3 is highest during the May–September/October period), thermodynamics, mixing layer height, etc. An analysis by ECCC revealed that 2019 average annual temperatures were one degree warmer on average nationally than the 1961–1990 climatological average. Annual average temperature anomalies were -0.5 to 1.5 °C in SWBC and -1.0 to 0.5 °C in the WQCC compared with the 1961–1990 reference period. This should be considered when conducting health impact analyses using these BPTs for years other than 2019, where meteorology may differ.
5. **NO_x titration / negative BPTs should be considered carefully.** NO_x BPTs for the WQCC are, overall, positive when health impacts due to $PM_{2.5}$, O_3, and NO_2 are added together. For SWBC, the sectors analyzed have overall negative BPTs due to NO_x titration. Negative BPTs indicate that an incremental decrease in NO_x emissions leads to local increases in ambient O_3 concentrations (Figure 4) that are not outweighed by reductions in ambient NO_2 and $PM_{2.5}$. Negative BPTs for O_3 are enhanced for health impacts based on annually averaged O_3 compared with those based on summer-season O_3 (Figure 4). In general, NO_x titration is more likely to exist in periods of low sunlight and low O_3 production (e.g., at night). Such conditions are more frequent during colder

months, leading to a more negative impact. O_3 disbenefits resulting from a reduction in NO_x emissions can be offset by emissions reductions of other pollutants that lead to improvements in O_3, such as VOCs. For example, a one tonne reduction in VOC emissions from the on-road sector in SWBC is estimated to have a health benefit of $9,900, while a one tonne reduction in NO_x emissions is estimated to have a disbenefit (or cost) of -$2,700. Co-reductions in both NO_x and VOC emissions for this sector may yield overall health benefits depending upon the relative amounts of NO_x to VOC emissions reductions. In this analysis, NO_x BPTs are likely underestimated (i.e., more negative) as any concentration changes between the base case and pollutant-sector-region scenario that were smaller than 0.005 µg/m^3 ($PM_{2.5}$) or 0.005 ppb (O_3/NO_2) were set to zero as ECCC did not consider these changes to be significant. As the impact of NO_x emissions reductions on ambient $PM_{2.5}$ concentrations is smaller than this in many pollutant-sector-region scenarios, NO_x BPTs do not always include positive health impacts on $PM_{2.5}$, which act to offset negative impacts from O_3 (Figure 4).

Negative BPTs due to NO_x titration are less likely to persist, and become gradually less negative, with large-scale reductions in NO_x emissions [or both NO_x and VOC emissions together; refer to discussion point 3)], such as those achieved through motor vehicle emissions standards or industrial emissions controls rolled out nationally (Hakami et al. 2003; Pappin et al. 2015). For example, the widespread emission reductions of NO_x, more so than VOCs, in the US from 1998 to 2013 have led to reductions in mean and peak summertime O_3 concentrations in the vast majority of urban areas of the US, though 5th percentile O_3 and wintertime means have increased in most urban areas (Simon et al. 2015). A recent analysis of satellite-based indicators of O_3 production chemistry across the US from 1996 to 2016 revealed that for the 1996–2000 period, seven major urban areas studied were characterized by a NO_x-inhibited chemical regime for mean summer O_3 (i.e., NO_x titration). By 2013–2016, the centre of only three urban areas (New York City, Los Angeles, and Chicago) remained NO_x-inhibited (Jin et al. 2020). Studies on the impacts of lockdowns during early stages of the COVID-19 pandemic, where NO_x emissions from sectors such as transportation declined dramatically, have revealed mixed effects on O_3 concentrations, which further illustrates the complexity of O_3 chemistry.

6. **Uncertainty in these BPT estimates exists and is not fully quantified.** BPT estimates presented in Tables 5 and 6 include 95% confidence intervals that represent uncertainty in health risks (i.e., CRFs) and monetization in AQBAT, but do not capture uncertainties related to other aspects of the health impacts modelling, such as population projections, baseline health endpoint rates, or the geographic resolution of health impacts (i.e., CDs). Importantly, confidence intervals in Tables 5 and 6 do not include uncertainties in ambient pollutant concentrations estimated from GEM-MACH (refer to section "Step 2. Air quality / chemical transport modelling" for a comparison of GEM-MACH concentrations to observations). These include inherent uncertainties in the methods used to derive emissions quantities, including macro-economic modelling, as well as in the estimation of weather/meteorological variables, and in the representation of complex atmospheric processes in chemical transport modelling. The scenario-based modelling approach employed, also referred to as the brute force method, can result in small, erroneous concentration differences due to the propagation of numerical errors

in the model. Concentration differences between the base case and pollutant-sector-region scenario ($\Delta\bar{C}$ in Equation 1) that were smaller than 0.005 µg/m^3 ($PM_{2.5}$) or 0.005 ppb (O_3/NO_2) at the census division level were set to zero by ECCC as these changes were not considered to be significant and are subject to uncertainty. As a result, health impacts (ΔHI) used to derive BPTs do not include impacts resulting from air quality changes smaller than this.

7. **BPTs do not quantify the impact of transboundary flow of air pollution across the Canada–US border.** BPTs reflect health impacts in Canada due to reductions in Canadian emissions. BPTs were not estimated for US emissions sources, nor do these BPTs account for health impacts in the US population resulting from emissions reductions in Canada. The chemical transport modelling in this analysis did, however, account for both Canadian and US emissions to estimate ambient air pollutant concentrations in the base case and emissions reduction scenarios. In many populous regions of Canada, the transport of air pollution across national borders is a substantial contributor to local air pollution (Environment Canada 2012). Emissions reductions from sources located in the US may result in sizeable health benefits for the Canadian population, or could shift the chemical conditions in Canada so as to modify these BPTs. These transboundary benefits are particularly relevant for emissions sources in the northeastern US that affect air quality in the WQCC and maritime provinces.

Comparison with other Canadian BPT estimates

To date, Canadian BPTs are available in a few published studies. Some of these studies use simplified methods to derive national-average or regional-average BPTs for a set of emitted pollutants. Others use chemical transport modelling to estimate location-specific BPTs across Canada. A full set of location-, sector-, and pollutant-specific estimates against which to compare these BPTs is not currently available in the literature for Canada. A comparison with sector- and pollutant-specific BPTs estimated by the US EPA therefore follows.

Pappin and Hakami (2013) conducted a study to attribute health impacts to location-specific emissions sources across Canada and the US, and estimated BPTs. The authors used an advanced sensitivity analysis tool in the chemical transport model, CMAQ, run for the summer of 2007. BPTs accounted for acute exposure mortality due to O_3 and NO_2 in Canada based on AQBAT's CRFs and an economic valuation of $5.7 million per statistical life (2011 CAD). As the sensitivity analysis tool was not yet developed for $PM_{2.5}$, health impacts of $PM_{2.5}$ were excluded. Health impacts for the US included acute exposure mortality for O_3 (Bell et al. 2004) and were combined with Canadian health impacts into a single BPT metric. BPTs for NO_x and VOC emissions were estimated for every 36 by 36 km model grid cell (Figures 4A and 4B in Pappin and Hakami 2013). The largest NO_x BPTs in Canada were found for grid cells within SWBC and the WQCC, ranging from roughly $10,000 to upwards of $75,000 per tonne (2011 CAD). It is noted that the modelling study spanned the summer season only and did not include O_3-related chronic exposure mortality, which may result in more positive NO_x BPTs. VOC BPTs were consistently lower, and ranged from less than $500 to $8,000 per tonne for grid cells within SWBC and the WQCC. Regional averages of Pappin and Hakami's BPTs for SWBC and the WQCC, though not reported, would lie within these ranges.

In a recent report, Ramboll estimated BPTs for the Metro Vancouver region and the Fraser Valley Regional District, overlapping with the SWBC region modelled in this analysis (Ramboll 2019). To estimate BPTs, Ramboll conducted a historical regression of the Metro Vancouver region's ambient pollutant concentration measurements against their emissions inventory. CRFs and economic valuations were applied from AQBAT 3.0 for 13 endpoints, including a valuation for mortality of $7.6 million (2017 CAD). BPT estimates were reported by Ramboll to be $357,000 per tonne of $PM_{2.5}$, $30,300 per tonne of NO_x, $52,000 per tonne of VOC, and $19,800 per tonne of SO_x. BPTs were not sector-specific and NH_3 BPTs were not derived. In the Regional Ground-Level Ozone Strategy developed for the Lower Fraser Valley (FVRD et al. 2014), regional chemical transport modelling and analysis of observational data revealed that the western portion of the Lower Fraser Valley (LFV) was always VOC-limited (akin to a negative NO_x BPT), and the eastern portion was VOC-limited on most days, except during the hottest summer days when VOC emissions increased, resulting in a NO_x-limited condition (a positive NO_x BPT). The report concluded that in the western LFV, VOC emissions reductions should be prioritized and any NO_x emissions reductions must be accompanied by an equal or greater VOC reduction. This intra-regional and temporal variability in O_3 chemistry is not captured in Health Canada's BPT estimates and should be kept in mind as a limitation when applying and interpreting these BPTs.

Earlier estimates of BPTs for Canada were reported in an analysis of the costs of transportation conducted by Transport Canada (2008). As part of the analysis, RWDI Consultants developed a reduced form or reduced-complexity tool – the Reduced Form Source Receptor Tool (ReFSoRT) – for estimating the air quality impacts of transportation emissions for the year 2000 at the CD level (Transport Canada 2008). CRFs were applied to estimate health impacts for 10 health endpoints included in an earlier version of AQBAT. National-average BPTs for the transportation sector were estimated to be $12,600 per tonne for $PM_{2.5}$ excluding paved road dust, $13,900 per tonne for $PM_{2.5}$ including paved road dust, $3,960 per tonne of SO_2, $3,580 per tonne of NO_x, and $436 per tonne of VOC. BPTs for NH_3 were not derived.

Due to the reduced-complexity modelling approaches employed by Ramboll (2019) and RWDI (Transport Canada 2008), their BPT estimates may not accurately represent nonlinear or complex relationships (such as the relationship between precursor emissions and health impacts due to secondary $PM_{2.5}$ and O_3), and should not be interpreted as the benchmark against which to evaluate BPTs in this report. They are included here for the completeness of the existing BPT literature for Canada.

Comparison with US estimates

Comparison with US EPA BPTs

The US EPA has generated BPT estimates using various modelling approaches (Fann et al. 2009, 2012; Wolfe et al. 2019), with earlier work focusing on BPTs for urban regions, and later work developing sector and subsector-specific BPTs for broader geographic regions or nationally. The US EPA's BPTs differ in a number of ways from this analysis. The US EPA's BPTs:

- are for a different population with different characteristics and density, and baseline rates of mortality and morbidity may differ;
- represent $PM_{2.5}$ health impacts rather than $PM_{2.5}$, O_3, and NO_2 health impacts;

- use different CRFs and economic valuations;
- are reported in USD and are derived for different years;
- are estimated using different modelling approaches to that used in this analysis; and
- are reported on a ton[4] vs. (metric) tonne of emissions.

Despite these differences, the US EPA's BPTs remain relevant for an order-of-magnitude comparison across sectors and pollutants in the absence of a benchmark for Canadian-specific estimates. The BPT estimates derived by Health Canada, Fann et al. (2009; 2012) and Wolfe et al. (2019) are summarized in Table 7, with a description of the US EPA's studies following. Note that the US EPA's BPTs represent benefits per ton (not tonne) of emissions.

[4] A ton refers to a short ton, i.e., 2,000 lb.

Table 7. Summary of Health Canada and US EPA BPT estimates, by source sector and emitted pollutant.

Emitted pollutant[a]	Source sector	BPT			
		Health Canada estimates (2015 CAD per tonne)[b,c,d,h,i]	Fann et al. 2009 (2006 USD per ton)[b,c,d,e]	Fann et al. 2012 (2010 USD per ton)[b,c,d,f]	Wolfe et al. 2019 (2015 USD per ton)[b,c,d,g]
$PM_{2.5}$	On-road mobile	410,000–520,000	550,000 (150,000–1,700,000)	370,000	410,000–700,000
	Off-road mobile	470,000–480,000		310,000	110,000–630,000
	Manufacturing + Ore and mineral industries	340,000–380,000	460,000 (65,000–1,100,000)	260,000	--
NO_x	On-road mobile	(-140)–15,000	10,000 [(-8,700)–43,000]	7,400	5,700–7,100
	Off-road mobile	(-2,700)–12,000		6,700	3,100–7,500
	Manufacturing + Ore and mineral industries	(-3,900)–4,900	9,700 [(-4,500)–28,000]	6,200	--
VOC	On-road mobile	3,900–13,000	2,400 (560–5,700)	--	---
	Off-road mobile	5,100–9,900		--	---
	Manufacturing + Ore and mineral industries	2,300–3,900		--	--
NH_3	On-road mobile	100,000–130,000	95,000 (36,000–140,000)	--	--
	Agriculture	26,000–46,000	38,000 [(-4,100)–53,000]	--	--
SO_x	Manufacturing + Ore and mineral industries	10,000	59,000 (9,100–550,000)	39,000	--

[a] Health Canada's $PM_{2.5}$ BPTs are derived for emitted $PM_{2.5}$ that includes crustal material, primary organic carbon, elemental carbon, sulphate, nitrate, and ammonium. Fann et al. (2009)'s BPTs are reported for primary carbonaceous particles and are listed under $PM_{2.5}$ in this table. Wolfe et al. (2019)'s BPTs are reported for emissions of elemental and organic carbon in the $PM_{2.5}$ size fraction.

[b] Health Canada's BPTs represent health impacts due to changes in ambient $PM_{2.5}$, O_3, and NO_2. The US EPA's BPTs represent health impacts due to changes in ambient $PM_{2.5}$ (not O_3 or NO_2) using various CRFs, as described following Table 7.

[c] Health Canada's BPTs are derived from 2015 modelling; Fann et al.'s (2009) BPTs are for 2015 projected modelling; Fann et al.'s (2012) BPTs are for 2016 projected modelling; and Wolfe et al.'s (2019) BPTs are for 2025 projected modelling. BPTs were not converted to a common currency year due to different methodologies used to adjust for inflation and income growth.

[d] Health Canada's BPTs listed represent the range over the two regions modelled. For Fann et al. (2009), national-average BPTs are listed, followed by the range of estimates across the 9 urban areas studied in parentheses. For Fann et al. (2012), national average BPTs are listed. For Wolfe et al. (2019), national average BPTs are listed as ranges across sub-sectors, as aggregate source categories were not reported.

[e] Fann et al.'s (2009) primary carbonaceous particle ($PM_{2.5}$) BPTs for manufacturing and ore and mineral industries are from the closest match: "EGU & Non-EGU carbon" sector. Fann et al.'s (2009) NO_x and SO_x BPTs for manufacturing and ore and mineral industries are from the closest match: "Non-EGU" sector. EGU refers to electricity generating units.
[f] Fann et al.'s (2012) BPTs for manufacturing and ore and mineral industries are taken from the closest match: "industrial point sources" (includes cement kilns, pulp and paper facilities, refineries, iron and steel facilities, coke ovens, integrated iron and steel facilities, electric arc furnaces, ferroalloy and all other point source emissions). BPTs are reported for SO_2, not SO_x.
[g] BPTs listed for Wolfe et al. (2019) show a range of national average BPTs across on-road sub-sectors (heavy-duty diesel, heavy-duty gas & CNG, light-duty diesel, light-duty gas cars and motorcycles, and light-duty gas trucks) and off-road sub-sectors (agriculture, commercial, construction, lawn & garden commercial, lawn & garden residential, recreational, and other). BPTs listed for Wolfe et al. are based on Krewski et al. (2009) CRF.
[h] Health Canada BPT estimates refer to this analysis.
[i] BPTs reflect the marginal change in societal economic welfare attainable from reducing emissions of a pollutant by one tonne, and include direct, indirect and intangible costs such as pain and suffering.

In their 2009 paper, Fann et al. estimated BPTs for nine urban areas of the US, and nationally, for four broad sectors (mobile sources, area sources, and combinations of electricity generating units (EGUs) and non-EGU point sources). BPTs were estimated using the US EPA's response surface model (RSM), derived from CMAQ simulations at a 36-km spatial resolution for 2015 projected emissions levels. Health impacts were modelled using population and baseline mortality/morbidity rate projections for 2015. CRFs for $PM_{2.5}$-related mortality were from the reanalysis of the American Cancer Society (ACS) cohort (Pope et al. 2002) and the Harvard Six Cities study (Laden et al. 2006). Morbidity endpoints were also included. An economic valuation of $6.2 million (2006 USD) was applied to premature mortality, and a 3% discount rate was used to account for the timing of health impacts. BPTs were derived for primary carbonaceous particles, NO_x, SO_x, NH_3, and VOCs and represent health impacts due to changes in $PM_{2.5}$ exposure only (O_3 and NO_2 were not included).

Fann et al. (2009) consistently found primary carbonaceous particles to have the highest BPTs among the emitted pollutants, ranging from $460,000 to $720,000 per ton across sectors nationally. SO_x BPTs ranged from $40,000 to $82,000 per ton nationally. NH_3 BPTs were larger for mobile sources ($95,000 per ton) compared with area sources ($38,000 per ton). NO_x BPTs were generally the lowest and ranged from $9,700 to $15,000 per ton nationally. VOC BPTs were an estimated $2,400 per ton nationally for all sectors combined. A large degree of variability was evident in BPT estimates across the nine urban areas in the study. The authors reported negative NO_x BPTs in five urban regions for some source sectors due to negative secondary $PM_{2.5}$ impacts (rather than negative O_3 impacts).

In a later study, Fann et al. (2012) estimated national-average BPTs for $PM_{2.5}$, NO_x, and SO_2, emissions in the US for 17 source sectors. BPTs were estimated for 2005 and 2016 based on health impacts related to $PM_{2.5}$ exposure – the summary below includes only the 2016 estimates. Fann et al. (2012) applied a source apportionment tool within the Comprehensive Air Quality model with Extensions (CAMx); a regional chemical transport model. BPTs were modelled at a 36-km spatial resolution using 2005 emissions projected to 2016, along with 2016 population, baseline mortality rate, and income growth projections. CRFs from the ACS Cancer Prevention Study (CPS)-II study (Krewski et al. 2009) and the Harvard Six Cities study (Laden et al. 2006) were applied for premature mortality due to $PM_{2.5}$ exposure. An economic valuation of $8.9 million (2007 USD, using 2016 income growth projections) was applied to premature mortality, along with a cessation lag to account for time of death and a 3% discount rate to account for time preferences. National-average BPTs reported by Fann et al. (2012) varied across sectors, ranging from \$46,000 to \$510,000 per ton of primary $PM_{2.5}$. SO_2 BPTs generally ranged between \$12,000 and \$98,000 per ton, with one estimate being substantially higher (\$410,000 per ton for iron and steel industries). NO_x BPTs ranged from \$1,900 to \$16,000 per ton. BPT estimates for sectors similar to those analyzed here are listed in Table 7.

More recently, Wolfe et al. (2019) derived BPTs for the US for 17 transportation sub-sectors. The authors used the source apportionment tool in CAMx (Fann et al. 2012), run at a 12-km spatial resolution over the contiguous US and for the 2025 analysis year, using emissions projected from 2011. Two sets of BPTs were estimated using the CRFs from Krewski et al. (2009) and an extended follow-up of the Harvard Six Cities Study (Lepeule et al. 2012) for $PM_{2.5}$ and chronic exposure mortality. Both sets of BPTs also accounted for morbidity endpoints. Wolfe et al. applied an economic valuation of \$10.4 million to premature mortality (inflated to 2025 USD and adjusted for income growth). A cessation lag was applied, and rates of 3% and 7% were used to discount future mortalities. National-average and region-specific BPTs were estimated (for the western and eastern parts of the country). Overall, national-average BPTs varied substantially across on-road and off-road sub-sectors, particularly for $PM_{2.5}$ BPTs from off-road sources, where the geographic distribution of emissions can vary widely across a nation. BPTs from the study are listed in Table 7 based on the CRF by Krewski et al. (2009). BPTs based on Lepeule et al.'s (2012) CRF were roughly 2.3 times greater than those listed in Table 7.

Comparison with BPTs derived using reduced-complexity models

Reduced-form or reduced-complexity models have been developed as screening tools that are computationally efficient in estimating health benefits for a large number of emissions reduction scenarios/sources but rely on simplifications about atmospheric transport and transformation processes. While such tools exist for the US and Europe, reduced-complexity models have so far not been developed for the Canadian context. A recent study that applies a number of reduced-complexity models for BPT estimation for the US is summarized below. This study is included here not to serve as a benchmark against which to evaluate Health Canada's BPT estimates, but rather to provide an example of the types of BPT analyses that exist in the literature.

In a comparison across three reduced-complexity models, Gilmore et al. (2019) estimated location-specific BPTs across the US. BPTs were estimated for ground-level and elevated sources separately. The reduced-complexity models included AP2 (the predecessor of the Air Pollution Emission Experiments and Policy analysis model or APEEP), the Intervention Model for Air Pollution (InMAP), and the Estimating Air Pollution Social Impact Using Regression model (EASIUR). These tools differ from chemical transport models in that they entail simplified representations of atmospheric processes in order to reduce the resources required to delineate health impacts from a large number of sources/sectors/locations. Gilmore et al. applied these models for the 2005 US emissions inventory and population. BPTs were estimated using the CRF from the ACS CPS II (Krewski et al. 2009) for mortality from $PM_{2.5}$ exposure. An economic valuation of \$7.4 million (2006 USD) was applied to premature mortality. Morbidity effects were not included in the estimates.

Consistent with Fann et al. (2009, 2012) and with this report, Gilmore et al. (2019) found that BPTs for primary $PM_{2.5}$ were larger than for precursor pollutants. Nationwide emission-weighted average BPTs for ground-level sources were largest for primary $PM_{2.5}$ and ranged from $70,000 to $120,000 per tonne across the three reduced-complexity models. National average BPTs for precursor pollutants ranged from $6,400 to $13,000 per tonne of NO_x, $21,000 to $45,000 per tonne of SO_2, and $38,000 to $49,000 per tonne of NH_3. Location-specific BPTs generally differed by up to a factor of 4 across pairs of reduced-complexity models for most counties. For a limited number of counties, BPTs differed by a factor of 10. BPTs predicted by the reduced-complexity models differed substantially for NO_x and NH_3 as compared with $PM_{2.5}$ and SO_2, indicating larger uncertainties associated with those BPTs.

The US EPA also commissioned a project to compare health impacts for the above reduced complexity models for major air pollution regulations in the US by Industrial Economics, Incorporated (IEC 2019), with a database published by Baker et al. (2020). BPTs were not directly reported so the study results are not included here.

Applying BPTs for evaluating emission mitigation measures

Step-by-step guide for estimating health impacts

BPTs provided in this report can be applied by users seeking to evaluate the health benefits of emission mitigation measures that apply to emissions in SWBC or the WQCC. Mitigation options involving a small-to-moderate **reduction in emissions** or a similarly sized **increase in emissions** can be considered. For most emitted pollutants (except NO_x in some cases), an emissions increase would result in an overall deterioration in air quality and would incur a **negative benefit (disbenefit)**, or cost, to society. To estimate health impacts via BPTs, users must estimate the quantity of emissions change from the sector in question and multiply this by the BPT (e.g., Ramboll 2019). If the mitigation measure in question affects multiple pollutant emissions, it is assumed that the total health benefits of the measure are equal to the sum of health benefits from reducing each emitted pollutant separately. For example, for a mitigation measure that affects both NO_x and VOC emissions, a small reduction in NO_x emissions is assumed not to change the chemical regime of the atmosphere, such that the BPT for VOC emissions still holds. Over large changes in emissions (e.g., 50%), BPTs are more likely to interact. The expedited process of estimating health impacts via BPTs can be used when timelines and resources do not allow for full-form air quality modelling. BPTs are also useful in cases where the expected change in emissions is too small for a chemical transport model to accurately model.

Steps for estimating health impacts

Users can estimate the health impacts of an emission mitigation measure using the following steps:

1. Evaluate the appropriateness of BPTs for assessing the emission mitigation option in question. If users answer “yes” to all questions below, then BPTs are suitable for their application:
   - ✓ Does the mitigation measure affect emissions sources that reside within SWBC or the WQCC?
   - ✓ Is the magnitude of emissions change within −10% to +10% from baseline levels?

- ✓ Will the mitigation measure affect emissions throughout the year, or is it a seasonal measure?
  - o If seasonal, BPTs may be used but may have increased uncertainty.
- ✓ Are BPTs listed for the sectors/sub-sectors and pollutants of interest (Table 4)?
  - o If the emissions sector is a sub-sector of the "on-road," "off-road," "manufacturing and ore and mineral industries," or "agriculture" sectors, users should evaluate how the geographical distribution of emissions differs from the aggregate sectors before applying BPTs.
- ✓ Is the health impact estimate sought for a current or recent analysis year?
  - o BPTs were derived for 2015 emissions levels with 2019 meteorology. If emissions or meteorology differ substantially for the analysis year for the mitigation measure, this will be a source of uncertainty.
  - o If users wish to convert to a different currency year (e.g., 2020 CAD), the CPI can be used (Statistics Canada 2022).

2. Select BPT estimate(s) from Table 4 for the pollutant(s), sector, and region of interest. Uncertainty estimates can be drawn from Tables 5 and 6, and represent uncertainty in the health impacts modelling, not the air quality modelling. If within-region or within-province BPTs are sought instead of total BPTs, users can extract estimates from Figures 3–7.
3. Estimate the change in air pollutant emissions resulting from the mitigation measure for each emitted pollutant. Units should be converted to tonnes per year.
4. Multiply change in emissions by BPT for each emitted pollutant, and sum over all emitted pollutants, via Equation 6:

$$\Delta HI_{policy} = \sum_{p} BPT_{p,s,r} \times \Delta E_{p,s,r} \qquad \text{Equation 6}$$

where

ΔHI_{policy} represents the total value of the health impacts associated with the mitigation measure ($ per year);
p refers to the emitted pollutant (primary $PM_{2.5}$, NO_x, SO_x, VOCs, or NH_3);
$BPT_{p,s,r}$ is the pollutant-sector-region-specific BPT estimate from Table 4 (or Figures 3–7 if within-region or within-province BPTs are sought) ($ per tonne), equivalent to the BPT in Equation 5; and
$\Delta E_{p,s,r}$ is the emissions change for the pollutant in question, for the sector and region under the mitigation measure, with a positive value defined as a reduction in emissions (tonnes per year).

The summation of $BPT_{p,s,r} \times \Delta E_{p,s,r}$ is taken across all pollutants whose emissions are reduced as a result of the mitigation measure. This assumes that health impacts are additive. For mitigation measures resulting in an **increase in emissions**, $\Delta E_{p,s,r}$ should be input as a negative number.

How to interpret health impacts estimated via BPTs

Interpretation of health impacts estimated via these BPTs differs based on the BPT metric used. Health impacts based on total BPTs (Tables 4–6) can be interpreted as the total economic value of health impacts resulting from the emission mitigation measure in question, encompassing health impacts both within and outside the region (SWBC or the WQCC), in 2015 CAD per year. A positive value of health impacts is interpreted as a health benefit, while a negative value is interpreted as a cost or disbenefit. It is noted that in deriving these BPTs, health impacts outside the region were found to be relatively small compared to impacts within the region (Figures 3–7). Health impacts based on within-region BPTs (Figures 3–7) are interpreted as the economic value of health impacts occurring within the same region as the emissions mitigation (and not outside of the region) in 2015 CAD per year (Statistics Canada 2022). These health impacts are referred to as regional health impacts. Health impacts based on within-province BPTs (Figures 3–7) should be interpreted as the economic value of health impacts within British Columbia, Ontario, or Quebec only (and not outside of the province), as a result of the mitigation measure, in 2015 CAD per year. These health impacts are referred to as provincial health impacts.

Limitations and uncertainties associated with these BPT estimates also apply to health impact estimates derived using them. Users are referred to the section “Limitations of Health Canada’s BPT estimates” for a detailed discussion of factors that must be considered when working with these BPTs, including the temporal period of health impacts, consideration of negative BPTs, and uncertainty bounds. Users reporting health impacts estimated via these BPTs should adequately describe how the BPTs were derived and what they represent.

Conclusion

A wide body of scientific literature provides evidence that exposure to ambient air pollution is associated with adverse health effects. In Canada, an estimated 15,300 premature deaths are associated with the above-background component of ambient air pollution exposure, with a monetized value of $120 billion per year in 2019 (2016 CAD; Health Canada 2021). BPTs, or the health benefits per unit tonne of emissions reduction, are a tool that enables decision makers to estimate the health benefits of air pollution mitigation strategies. In this analysis, Health Canada estimated region-, sector-, and -pollutant-specific BPTs for two Canadian regions (southwestern British Columbia [SWBC] and the Windsor–Quebec City corridor [WQCC]); four source sectors (on-road mobile, off-road mobile, manufacturing and ore and mineral industries, and agriculture); and five emitted pollutants (primary $PM_{2.5}$, NO_x, VOCs, SO_x, and NH_3).

A number of broad findings are noteworthy from this analysis. Primary $PM_{2.5}$ BPTs are the largest in magnitude of all emitted pollutants due to reductions in ambient $PM_{2.5}$ concentrations. NH_3 has the largest BPTs of the gas-phase precursor pollutants (NO_x, SO_x, VOCs, and NH_3), particularly for the on-road sector. BPTs for NO_x, SO_x, and VOCs were smaller. An analysis of health impacts by location reveals that BPTs are due largely to health impacts occurring within the same region as the emissions reduction. These BPT estimates are suitable for analyzing the health impacts of small-to-moderate size, regional-scale emissions reduction policies within these regions.

It is strongly recommended that users wishing to apply Health Canada's BPT estimates to assess the health impacts of emission mitigation options follow the step-by-step process provided in this report, and carefully evaluate the limitations and uncertainties of these BPTs.

Should users have any questions not addressed in this report, they are encouraged to contact Health Canada for clarification. Health Canada may expand its analysis of BPTs in the future to include additional sectors and/or regions of broad interest to air pollution risk management in Canada.

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Appendix A. Summary of AQBAT 3.0 Concentration Response Functions.

Pollutant	Endpoint	Source(s)	Details	Averaging period	Regression type	Form	Mean beta	SE beta
NO_2	Acute exposure mortality	Burnett et al. (2004) Results from model with four gases provided by R.T. Burnett, in addition to published results	Analysis of air pollution and mortality in 12 Canadian cities. The lead author provided results from additional multi-pollutant models not reported in the paper; the four-gas model was selected based on the overall *t*-value among the candidate models. Percent excess mortality (associated with the mean pollutant concentration) from Poisson regression models for CO, NO_2, O_3 and SO_2, respectively, was 0.19% (t = 0.73, 1.0 ppm), 1.69% (t = 3.00, 22.4 ppb), 2.60% (t = 6.16, 30.6 ppb) and 0.23% (t = 2.09, 5.0 ppb). These results translate into regression coefficients (SE) of 0.00190 (0.00260), 0.000748 (0.000249), 0.000839 (0.000136) and 0.000459 (0.000220) for the same four pollutants, respectively. Although this multi-pollutant model excluded PM, it was selected as the model that best reflected the impact of the overall air pollution mix. Because of multi-co-linearity among pollutants, this model should nonetheless still reflect impacts of PM. In any case, the effects of PM in this study were reduced substantially when it was modelled together with NO_2, the effect of which predominated in this analysis. The AQBAT CRF is applied to all members of all age groups.	24 h	Log(RR) or Log(OR)	Normal	7.48E-04	2.49E-04
O_3				1 h			8.39E-04	1.36E-04
O_3 (May–Sept.)	Respiratory mortality	Jerrett et al. (2009)	Jerrett et al. (2009) analyzed data from the American Cancer Society cohort study. The relative risk of death from respiratory causes was 1.040 (95% CI 1.010–1.067) per 10 ppb O_3 in a model with $PM_{2.5}$; exposure was based on average of quarterly averages with ≥ 75% of daily values. This translates into a coefficient of 0.00392 with SE 0.00132. The AQBAT CRF is applied to the Canadian population ≥ 25 years of age.	1 h	Log(RR) or Log(OR)	Normal	3.92E-03	1.32E-03

Pollutant	Endpoint	Source(s)	Details	Averaging period	Regression type	Form	Mean beta	SE beta
O_3 (May–Sept.)	Acute respiratory symptom days	Krupnick et al. (1990)	The authors reported on the association between O_3 and the occurrence of acute respiratory symptoms in a panel of California families. They employed a Markov model that accounted for the occurrence of symptoms on the previous day and adjusted for CoH, NO_2 and SO_2 as co-pollutants. The incremental change in frequency of symptoms was calculated by substituting the coefficient from table V, column 3, divided by 10 to convert from pphm to ppb, together with the transitional probabilities, $p_1 = 0.775$ and $p_2 = 0.0468$ (provided by the authors), into equation 3 on page 12 of the paper. The baseline frequency of symptoms was calculated by substituting p_1 and p_0 into equation 2. Thus, the proportional change per 1 ppb O_3 is the output from equation 3 divided by that of equation 2, 0.000786 (SE 0.000386). The AQBAT CRF is applied to adults and non-asthmatic (85.7%) children aged 5–19 years.	1 h	Linear	Normal	7.86E-04	3.86E-04

Pollutant	Endpoint	Source(s)	Details	Averaging period	Regression type	Form	Mean beta	SE beta
O_3 (May–Sept.)	Asthma symptom days	Mortimer et al. (2002) Schildcrout et al. (2006)	Numerous panel studies have been conducted on the association between O_3 and asthma exacerbations in children. Several of these were carried out in summer camps, which may not reflect typical exposure conditions, in that campers would be expected to spend more time outdoors compared with non-campers. Others have been conducted in locations such as Mexico City and Los Angeles, which experience very high O_3 concentrations not representative of conditions in Canada. We therefore selected two large multicentre North American panel studies as the source of the CRF. Mortimer et al. (2002) analyzed data collected in summer 1993 for 846 inner-city children aged 4–9 years from eight American cities. The average 8 h maximum O_3 concentration among all cities was 48 ppb. The odds ratio for morning asthma symptoms was 1.16 (95% CI 1.02–1.30) in relation to a 15 ppb increment in average of lag 1–5 day O_3. This was reduced to 1.07 (0.92–1.26) in a joint model with NO_2 in seven cities and to 1.04 (0.70–1.55) in a joint model with PM_{10} based on three cities (table 4). Schildcrout et al. (2006) analyzed data collected from 1993 to 1995 for 990 children aged 5–13 years, also from eight cities and including Toronto, and only with Baltimore in common with the Mortimer et al. (2002) analysis. Median 1 h maximum O_3 concentrations ranged from 43 to 65.8 ppb. The odds ratio for asthma symptoms was 1.06 (95% CI 0.92–1.23) in relation to a 30 ppb increment in lag 0 O_3 (the largest effect among lags considered; figure 1). Joint models with other pollutants were not run. The log odds ratio from Mortimer et al. (2002) based on the 8 h maximum (joint model with NO_2) was multiplied by 1.13 (the ratio of 1 h maximum to 8 h maximum in Canadian cities) and pooled with the Schildcrout et al. (2006) result to obtain an odds ratio of 1.05 (95% CI 0.96–1.14) per 20 ppb. The same baseline frequency of asthma symptoms and prevalence of current wheeze as for $PM_{2.5}$ was applied to 14.3% of children aged 5–19 years.	1 h	Log(RR) or Log(OR)	Normal	2.38E-03	2.19E-04

Pollutant	Endpoint	Source(s)	Details	Averaging period	Regression type	Form	Mean beta	SE beta
O_3 (May–Sept.)	Minor restricted activity days	Ostro and Rothschild (1989)	Ostro and Rothschild (1989) reported an association between O_3 and minor reduced activity days based on an analysis of data from the US Health Interview Survey. They reported results by year for 1976–1981 based on a Poisson regression model including both O_3 and $PM_{2.5}$ (table 4, column 2). Coefficients were pooled using a random effect model, and a pooled estimate of 0.000530 (SE 0.00291) per 1 ppb daily 1 h maximum O_3 was obtained. The baseline daily rate of minor reduced activity days per person was 7.8/365 = 0.0214. The AQBAT CRF is applied to adults and non-asthmatic (85.7%) children aged 5–19 years.	1 h	Log(RR) or Log(OR)	Normal	5.30E-04	2.91E-03
O_3 (May–Sept.)	Respiratory emergency room visits	Burnett et al. (1997) Stieb et al. (2000)	Substantially more data are available pertaining to air pollution and hospital admissions in Canada relative to emergency department visits. We therefore elected to represent the effects of air pollution on respiratory emergency department visits using the results for hospital admissions scaled up in number based on the relative frequency of hospital admissions and emergency visits for these conditions. Thus, the coefficient per unit air pollution was the same as for hospital admissions based on Burnett et al. (1997), i.e., 0.000791 (SE 0.000355) per 1 ppb. The baseline rate of emergency visits is equal to the baseline rate of hospital admissions divided by 0.198, the proportion of visits resulting in hospital admission as reported by Stieb et al. (2000). The AQBAT CRF is applied to all members of all age groups.	1 h	Log(RR) or Log(OR)	Normal	7.91E-04	3.55E-04
O_3 (May–Sept.)	Respiratory hospital admissions	Burnett et al. (1997)	Burnett et al. (1997) reported the results of a study on O_3 and respiratory hospital admissions in 16 Canadian cities. Based on results from a Poisson regression model, which simultaneously adjusted for dew point temperature, CO and CoH, they reported a relative risk of 1.024 (p = 0.0258) per 30 ppb daily 1 h maximum O_3. Taking the natural logarithm of the relative risk and dividing by 30 yields a coefficient of 0.000791 (SE 0.000355) per 1 ppb. The AQBAT CRF is applied to all members of all age groups.	1 h	Log(RR) or Log(OR)	Normal	7.91E-04	3.55E-04

Pollutant	Endpoint	Source(s)	Details	Averaging period	Regression type	Form	Mean beta	SE beta
$PM_{2.5}$	Acute respiratory symptom days	Krupnick et al. (1990)	The authors reported on the association between CoH and the occurrence of acute respiratory symptoms in a panel of California families. They employed a Markov model that accounted for the occurrence of symptoms on the previous day and adjusted for O_3, NO_2 and SO_2 as co-pollutants. The incremental change in frequency of symptoms was calculated by substituting the coefficient from table V, column 3, multiplied by 0.211 to convert from CoH to $PM_{2.5}$, together with the transitional probabilities, p_1 = 0.775 and p_2 = 0.0468 (provided by the authors), into equation 3 on page 12 of the paper. The conversion from CoH to $PM_{2.5}$ was calculated by dividing the ratio of CoH to TSP (0.116) provided by the authors by the ratio of PM_{10} to TSP (0.55) provided by Environment Canada. This assumes that the toxicity of $PM_{2.5}$ per 1 µg/m³ is the same as that of PM_{10}. The baseline frequency of symptoms was calculated by substituting p_1 and p_0 into equation 2. Thus, the proportional change per 1 µg/m³ $PM_{2.5}$ is the output from equation 3 divided by that of equation 2, 0.00266 (SE 0.00139). The AQBAT CRF is applied to adults and non-asthmatic (85.7%) children aged 5–19 years.	24 h	Linear	Normal	2.66E-03	1.39E-03
$PM_{2.5}$	Adult chronic bronchitis cases	Abbey et al. (1995)	Abbey et al. (1995) reported the results of a cohort study of air pollution and the development of chronic lung disease among non-smoking Seventh Day Adventists living in California. Based on a logistic regression model, which also included personal characteristics, they reported an odds ratio of 1.81 (95% CI 0.98–3.25) for the development of chronic bronchitis per 45 µg/m³ $PM_{2.5}$ (table 2, row 2). Taking the natural log of the odds ratio and dividing by 45 yields a coefficient of 0.0132 (SE 0.006 80) per 1 µg/m³ $PM_{2.5}$. They reported that the 10-year incidence of chronic bronchitis was 6.26% (117 new cases occurred among 1868 subjects for whom $PM_{2.5}$ exposures could be estimated). We calculate the annual incidence, p_1, from the expression: $0.0626 = 1 - (1 - p_1)^{10}$, so that p_1 = 0.006 44. The AQBAT CRF is applied to the Canadian population ≥ 25 years of age.	24 h	Log(RR) or Log(OR)	Normal	1.32E-02	6.80E-03

Pollutant	Endpoint	Source(s)	Details	Averaging period	Regression type	Form	Mean beta	SE beta
$PM_{2.5}$	Asthma symptom days	Weinmayr et al. (2010) Ward and Ayres (2004) Dell et al. (2010)	These parameters are derived using the same approach as described in the Health Risk of Air Pollution in Europe project of the WHO European Centre for Environment and Health. Weinmayr et al. (2010) conducted a systematic review and meta-analysis based on 36 studies of the association between air pollution and asthma symptoms in children. The pooled odds ratio was 1.028 (95% CI 1.006–1.051) per 10 μg/m^3 PM_{10} (table 2) based on a random effect model including all studies. This is based on single pollutant models, as results from multi-pollutant models were not consistently available. However, the derived effect size is nonetheless much smaller than that observed in a multi-pollutant model for North American cities in Mortimer et al. (2002). In order to derive an odds ratio for $PM_{2.5}$, we multiplied the log odds ratio for PM_{10} by 2.37, which is the average of the ratio of log pooled odds ratios for $PM_{2.5}$ vs. PM_{10} for cough and other respiratory symptoms reported by Ward and Ayres (2004; tables 3 and 4) in their earlier meta-analysis. The result is an odds ratio for $PM_{2.5}$ of 1.07 (95% CI 1.01–1.12). The baseline daily frequency of asthma symptoms in asthmatic children varies widely in panel studies. We have conservatively estimated it at 20%. The population to which this is applicable is based on the prevalence of current wheeze in Canada from the National Longitudinal Survey of Children and Youth (14.3%; Dell et al. 2010). This is applied to asthmatic children (14.3%) aged 5–19 years.	24 h	Log(RR) or Log(OR)	Normal	6.545E-03	2.646E-03
$PM_{2.5}$	Cardiac emergency room visits	Burnett et al. (1995) Stieb et al. (2000)	Substantially more data are available pertaining to air pollution and hospital admissions in Canada relative to emergency department visits. We therefore elected to represent the effects of air pollution on cardiac emergency department visits using the results for hospital admissions scaled up in number based on the relative frequency of hospital admissions and emergency visits for these conditions. Thus, the change in frequency per unit air pollution was the same as for hospital admissions based on Burnett et al. (1995) – i.e., 0.0711% (SE 0.0170) increase per 1 μg/m^3. The baseline rate of emergency visits is equal to the baseline rate of hospital admissions divided by 0.760, the proportion of visits resulting in hospital admission as reported by Stieb et al. (2000). The AQBAT CRF is applied to all members of all age groups.	24 h	Linear	Normal	7.11E-04	1.70E-04

Pollutant	Endpoint	Source(s)	Details	Averaging period	Regression type	Form	Mean beta	SE beta
$PM_{2.5}$	Cardiac hospital admissions	Burnett et al. (1995)	Burnett et al. (1995) reported a 3.3% (95% CI 1.7–4.8) increase in cardiac hospital admissions per 13 μg/m³ sulphate based on a linear regression model that also included O_3 and temperature (table 5, row 2). Multiplying by the average ratio of sulphate to $PM_{2.5}$ of 0.28 (Environment Canada), this equates to a 0.0711% (SE 0.0170) increase per 1 μg/m³ $PM_{2.5}$. The AQBAT CRF is applied to all members of all age groups.	24 h	Linear	Normal	7.11E-04	1.70E-04
$PM_{2.5}$	Child acute bronchitis episodes	Hoek et al. (2012) Dockery et al. (1996)	These parameters are derived using the same approach as described in the Health Risk of Air Pollution in Europe project of the WHO European Centre for Environment and Health. Hoek et al. (2012) conducted a meta-analysis of eight cross-sectional studies from Europe and North America, including the 24 cities study, which included data from several Canadian communities. The random effect pooled estimate of the odds ratio was 1.08 (95% CI 0.98–1.19) per 10 μg/m³ PM_{10} (table 3), adjusted for age, sex, maternal education, paternal education, household crowding, current parental smoking, mother smoking during pregnancy, gas cooking, unvented gas/oil/kerosene heater, mould, nationality, birth order and "ever had a pet." The effect size was reduced based on joint models with SO_2, but this was based on only three studies (table 4). The average prevalence of bronchitis among the studies was 18.6% (table 2). In the 24 cities study, the odds ratio for bronchitis for $PM_{2.5}$ was identical to that for PM_{10} across the exposure difference between highest- and lowest-exposure communities, 17.3 and 14.9 μg/m³ for PM_{10} and $PM_{2.5}$, respectively (tables 1 and 4). We therefore multiply the log of the pooled odds ratios for PM_{10} by this ratio (1.16) in order to derive a log odds ratio per 10 μg/m³ $PM_{2.5}$, resulting in an odds ratio of 1.09 (95% CI 0.98–1.22). This is applied to the population of children 5–19 years of age.	24 h	Log(RR) or Log(OR)	Normal	8.927E-03	5.745E-03

Pollutant	Endpoint	Source(s)	Details	Averaging period	Regression type	Form	Mean beta	SE beta
$PM_{2.5}$	Chronic Exposure Internal Cause Mortality	Crouse et al. (2012)	Crouse et al. (2012) examined the association between PM2.5 derived from satellite observations and mortality during ten years of follow-up of a cohort of 2.1 million Canadians based on the 1991 long form census. Using a spatial random-effects Cox model including individual and ecological covariates and an urban/rural indicator, and accounting for spatial autocorrelation among cohort members, they reported a hazard ratio of 1.10 (95% CI 1.05-1.15) per 10 µg/m3 PM2.5. This translates to a β of 0.00953 with standard error 0.00232. [Note: choose either 4 specific causes or internal causes, not both.]	24 h	Log(RR) or Log(OR)	Normal	9.53E-3	2.32E-03
$PM_{2.5}$	Respiratory emergency room visits	Burnett et al. (1995) Stieb et al. (2000)	Substantially more data are available pertaining to air pollution and hospital admissions in Canada relative to emergency department visits. We therefore elected to represent the effects of air pollution on respiratory emergency department visits using the results for respiratory hospital admissions scaled up in number based on the relative frequency of hospital admissions and emergency visits for these conditions. Thus, the change in frequency per unit air pollution was the same as for hospital admissions based on Burnett et al. (1995) – i.e. 0.0754% (SE 0.0132) increase per 1 µg/m^3. The baseline rate of respiratory emergency visits is equal to the baseline rate of hospital admissions divided by 0.198, the proportion of visits resulting in hospital admission as reported by Stieb et al. (2000). The AQBAT CRF is applied to all members of all age groups.	24 h	Linear	Normal	7.54E-04	1.32E-04
$PM_{2.5}$	Respiratory hospital admissions	Burnett et al. (1995)	Burnett et al. (1995) reported a 3.5% (95% CI 2.3–4.7) increase in respiratory hospital admissions per 13 µg/m^3 sulphate based on a linear regression model that also included O_3 and temperature (table 4, row 2). Multiplying by the average ratio of sulphate to $PM_{2.5}$ of 0.28 (Environment Canada), this equates to a 0.0754% (SE 0.0132) increase per 1 µg/m^3 $PM_{2.5}$. The AQBAT CRF is applied to all members of all age groups.	24 h	Linear	Normal	7.54E-04	1.32E-04

Pollutant	Endpoint	Source(s)	Details	Averaging period	Regression type	Form	Mean beta	SE beta
$PM_{2.5}$	Restricted activity days	Ostro (1987) Ostro and Rothschild (1989) Chestnut et al. (1999)	Ostro (1987) reported an association between $PM_{2.5}$ and reduced activity days based on an analysis of data from the US Health Interview Survey. They reported results by year for 1976–1981 based on a Poisson regression model (table III, column 2). We pooled these coefficients using a random effect model and obtained a pooled estimate of 0.00481 (SE 0.00101) per 1 µg/m^3 $PM_{2.5}$. The baseline daily rate of reduced activity days per person was 0.052 (Chestnut et al. 1999). Ostro and Rothschild (1989) also reported an analysis of $PM_{2.5}$ and respiratory reduced activity days, in which they adjusted for the simultaneous effects of ozone. The effects of $PM_{2.5}$ were unaffected by this adjustment; thus, we opted to use the results from their earlier analysis on the grounds that reduced activity days are a more global outcome than the more narrowly defined respiratory reduced activity days. The AQBAT CRF is applied to adults and non-asthmatic (85.7%) children aged 5–19 years.	24 h	Log(RR) or Log(OR)	Normal	4.81E-03	1.01E-03

[a] Distribution forms: Normal, Gamma, Discrete, and Triangular

[b] Mean beta: mean of the pollutant coefficient (regression parameter)

[c] SE beta: standard error of the pollutant coefficient

[d] Coefficient of haze is a measure of the atmospheric impedance of light caused by suspended atmospheric particles or aerosols.

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Appendix B. Health Endpoint Baseline Rates in 2015 for Canada (annual events per million).

Health endpoint name	Age group	Baseline rate (annual events per million)	Notes
Acute Exposure Mortality	all ages	6,980	CD-specific rate available
Chronic Exposure Mortality	25+	9,680	CD-specific rate available
Chronic Exposure Respiratory Mortality	30+	1,040	CD-specific rate available
Acute Respiratory Symptom Days	100% of all adults(20+) and 82.3% (non-asthmatic) of children aged 5–19	64,000,000	The single rate is applied to all geo areas
Adult Chronic Bronchitis Cases	25+	6,400	The single rate is applied to all geo areas
Asthma Symptom Days	17.7% of children aged 5–19	73,000,000	The single rate is applied to all geo areas
Cardiac Emergency Room Visits	all ages	9,820	CD-specific rate available
Cardiac Hospital Admissions	all ages	7,460	CD-specific rate available
Child Acute Bronchitis Episodes	children aged 5–19	186,000	The single rate is applied to all geo areas
Minor Restricted Activity Days	100% of all adults(20+) and 82.3% (non-asthmatic) of children aged 5–19	8,000,000	The single rate is applied to all geo areas
Respiratory Emergency Room Visits	all ages	25,700	CD-specific rate available
Respiratory Hospital Admissions	all ages	5,080	CD-specific rate available
Restricted Activity Days	100% of all adults(20+) and 82.3% (non-asthmatic) of children aged 5–19	19,000,000	The single rate is applied to all geo areas