# Review and Assessment of the Canada-U.S. Air Quality Agreement (AQA)

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# Executive Summary

The Canada-U.S. Air Quality Agreement (AQA), signed in 1991, was originally designed to address transboundary contributions to acid rain caused by emissions of sulfur dioxide ($SO_2$) and oxides of nitrogen ($NO_X$). In 2000, the Agreement was amended to address the problem of transboundary ground-level ozone with the addition of commitments on volatile organic compounds (VOCs) and additional measures on $NO_X$.

By 2007, both countries had met their respective commitments under the Agreement. While the AQA is a remarkable example of what can be achieved through bilateral cooperation, commitments are now over 20 years old. Thus, under Article X, which calls for a comprehensive review and assessment of the Agreement and its implementation every five years unless otherwise agreed, Canada and the U.S. have jointly undertaken the work presented herein.

The objectives of this review and assessment are to:

- Review what the AQA has accomplished to date, including whether it is meeting its current objectives and whether emission reductions mandated by the Agreement have met the AQA objectives to: reduce the transboundary flow of air pollution, reduce acid deposition, reduce concentrations of ground-level ozone, and improve air quality in Canada and the U.S.;
- Assess whether the emissions reduction targets and measures included in Annex 1 (acid rain) and in Annex 3 (ozone) and the commitments in Annex 2 (scientific cooperation) remain appropriate for Canada/U.S. policy and science needs; and
- Examine whether new actions such as commitments and/or measures would be appropriate (*e.g.*, for pollutants included under the Agreement and those not currently addressed, such as $PM_{2.5}$).

In the context of this review and assessment, the Parties shall consider such action as may be appropriate, including the modification of the AQA and/or the modification of existing policies, programs, and measures.

## Acid Rain

The Acid Rain Annex (Annex 1) to the AQA sets out objectives for Canada and the U.S. to reduce emissions of $SO_2$ and $NO_X$ that cause acid rain. Both countries have met their commitments to reduce $SO_2$ and $NO_X$ emissions under the Agreement since 2007.

In 2020, Canada's total $SO_2$ emissions were approximately 651,000 metric tons, a 78% reduction from Canada's total $SO_2$ emissions of 3.0 million metric tons in 1990. Between 1990 and 2020, Canada's total $NO_X$ emissions also decreased by 36% (826 thousand metric tons). These emissions reductions have been achieved through programs including the 1985 Eastern Canada Acid Rain Control Program and Canada-wide Acid Rain Strategy for Post-2000.

In the U.S., between 1990 and 2020, $SO_2$ emissions have decreased by 92% from 23.1 million metric tons to 1.9 million metric tons, and $NO_X$ emissions have decreased by 69%, from 25.5 million metric tons to 7.8 million metric tons. The Acid Rain Program (ARP) in the U.S. has dramatically cut power plant emissions of $SO_2$ and $NO_X$, reducing acid rain. Regulatory actions in the U.S. such as the Cross-State Air Pollution Rule (CSAPR) and its subsequent updates have also achieved large reductions in annual $SO_2$ and annual and summertime $NO_X$ emissions from the power sector.

Continued and remarkable success in both countries in reducing pollutants contributing to acid deposition ($SO_2$ and $NO_X$) has led to recent signs of recovery. There are areas in both countries, most notably in eastern Canada, that are still recovering from the historic pollutant loadings and receiving acid deposition that may be in exceedance of current critical loads. Modeling suggests transboundary influence on total deposition, particularly in the less populated parts of northern Montana and the northern parts of the province of Ontario, where deposition is lower than in the northeastern U.S. Furthermore, deposition of reduced nitrogen (including $NH_3$ and $NH_4^+$) has not decreased in recent decades, and increased deposition of reduced nitrogen has been observed in some areas.

## Ozone

In 2000, the Ozone Annex (Annex 3) to the AQA set out commitments by Canada and the U.S. to reduce emissions of $NO_X$ and VOCs that contribute to transboundary ozone pollution. These commitments apply to a defined region in both countries known as the Pollutant Emission Management Area (PEMA), which includes central and southern Ontario, Southern Quebec, 18 U.S. States and the District of Columbia. At the time of the signing of the Annex, this PEMA was the area deemed the most critical for reducing transboundary ozone. These commitments aimed to help both countries attain their respective air quality goals, and to protect human health and the environment.

Canada and the U.S. have met their commitments in the Ozone Annex to reduce emissions of $NO_X$ and VOCs from stationary and mobile sources and from solvents, paints, and consumer products. Canada's national emissions of $NO_X$ and VOCs have decreased by 36% and 49%, respectively, between 1990 and 2020. The U.S. national air emissions of $NO_X$ and VOCs decreased by 70% and 48% respectively between 1990 and 2020.

Ambient ozone concentrations have also declined within the Canada-U.S. border region since the establishment of the Ozone Annex. Annual 4th highest MDA8 ozone concentrations have decreased by more than 10 parts per billion (ppb) at many monitoring stations across Ontario, Quebec, and the Maritimes, and by as much as 20 ppb at some stations in the Great Lakes states and Ohio Valley, where ozone concentrations are highest.

Ozone also continues to have significant impacts on public health and agricultural production in the U.S. and Canada, despite progress under Annex 3 of the Agreement. Transport from the U.S. continues to contribute a large fraction of anthropogenic ozone in Canada, with the largest influence in the Windsor-Quebec corridor, as well as southwestern British Columbia, in the greater Vancouver and Victoria area, southern Alberta, the Greater Toronto-Hamilton area, and the Montreal area. Air monitoring stations in southern Ontario and southern Quebec continue to measure ozone concentrations which approach or exceed the CAAQS, and modeling projections suggest continued CAAQS exceedances in 2035. Modeling suggests that transboundary flow of ozone and its precursors from the U.S. to Canada contributes to a significant portion of health impacts in central and Atlantic Canada and is the dominant source of health impacts in Ontario, Quebec, Nova Scotia, New Brunswick, Newfoundland and Labrador, and Prince Edward Island. Transboundary flow from the U.S. into Canada is also estimated to contribute to reduced crop yield, particularly along the Windsor-Quebec City corridor.

## Fine Particulate Matter

Although the Agreement does not include $PM_{2.5}$, emissions of some of the precursors of secondary $PM_{2.5}$ are addressed via actions to reduce $NO_X$, $SO_2$, and VOCs. However, direct emissions of primary $PM_{2.5}$ and

$NH_3$ (a $PM_{2.5}$ precursor) are not addressed under the Agreement. From 1990 to 2020, Canada's emissions of primary $PM_{2.5}$ decreased by 15%, having plateaued at approximately 1.5 million metric tons per year. U.S. national emissions of primary $PM_{2.5}$ decreased by 38% between 1990 and 2020, having gradually decreased until 2015, and then plateaued in recent years. The regional and multi-state programs that led to decreased ozone concentrations also reduced emissions of several chemical precursors to secondary $PM_{2.5}$ ($NO_X$, $SO_2$, and VOCs). However, emissions of $NH_3$ (another $PM_{2.5}$ precursor) have increased by 24% in Canada and by 25% in the U.S. from 1990 to 2020.

Adverse health impacts of $PM_{2.5}$ exposure are well documented and both countries continue to take action to address their respective emissions. In recent years, $PM_{2.5}$ concentrations are largest near urban areas and particularly in the Ohio Valley, Atlantic coast, and the Windsor-Quebec corridor, with observed concentrations for several stations in Canada approaching or exceeding the CAAQS. Although modeling projects that $PM_{2.5}$ concentrations will decrease by 2035, they are also projected to continue to exceed the CAAQS in some of Canada's largest cities. The analysis presented in this review and assessment finds greater transport of $PM_{2.5}$ from the U.S. to Canada. Recent modeling and analysis also indicate that transboundary $PM_{2.5}$ increases morbidity and mortality in Canada and has a larger health impact than transboundary ozone. Modeling results support the conclusion that the majority of transboundary $PM_{2.5}$ impacts are within several hundred kilometers of the border and felt predominantly in the Michigan-Ontario and Quebec regions – with the largest impacts in the Detroit-Windsor area.

## Scientific and Technical Cooperation

The Parties have strengthened their relationship through collaboration and science exchanges under the AQA. Since 1994, the Parties have notified each other of specific sources of air pollution within 100 miles of the Canada-U.S. border. The Canadian and U.S. governments share data through a range of programs and tools such as AirNow and the National Atmospheric Deposition Program (NADP), and maintain ongoing informal dialogue across a range of topics related to monitoring networks and measurement methods. Canada and the U.S. collaborate and share emission inventories, summaries, and analyses under several agreements and reports such as the AQA bi-annual Progress Report, Arctic Council, and the Convention on Long-Range Transboundary Air Pollution (LRTAP). In addition to meeting their respective pollution commitments, the Parties completed multiple pilot projects and joint modeling and analysis under the Ozone Annex. These efforts have expanded information sharing and knowledge of transboundary transport, to the benefit of each country.

Looking to the future, Subcommittee 2 (SC2) co-chairs held a series of science exchange workshops to share information, continue to build connections, and inform policy-related dialogue on current and emerging transboundary issues of concern. The Parties have accomplished a great deal under the AQA, continue to collaborate through a variety of projects and look for ways to continue working together in pursuit of shared goals to improve air quality conditions.

## Overall Conclusion and Recommendation

The Canada-U.S. AQA is a model of successful bilateral cooperation resulting in significant improvement in the environment over its three-decade history. Overall findings of the AQA review and assessment indicate that important results have been achieved under the current AQA and both countries have fully met their obligations. However, transboundary air pollution continues to impact both countries from a health and environmental perspective. To continue to meet the objective of the AQA "to control transboundary air pollution between the two countries”, it is recommended that the Agreement be updated, including exploring new strategies to address emerging issues of concern not currently covered under the AQA.

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## List of Abbreviations/Acronyms

| | |
|---|---|
| ADAGIO | Atmospheric Deposition Analysis Generated from optimal Interpolation from Observations |
| ANC | Acid neutralizing capacity |
| APCA | Anthropogenic Precursor Culpability Assessment |
| APEI | Air Pollutant Emissions Inventory (Canada) |
| ARP | Acid Rain Program (U.S.) |
| AQA | Canada-United States Air Quality Agreement |
| AQBAT | Air Quality Benefits Assessment Tool (Health Canada) |
| AQC | Canada-United States Air Quality Committee |
| AQMEII4 | Air Quality Model Evaluation International Initiative Phase 4 |
| AQMS | Air Quality Management System (Canada) |
| AQS | Air Quality System (U.S.) |
| AQVM2 | Air Quality Valuation Model |
| BAU | Business as usual |
| BLIER | Base Level Industrial Emissions Requirements (Canada) |
| CAAQS | Canadian Ambient Air Quality Standards |
| CAIR | Clean Air Interstate Rule (U.S.) |
| CAMx | Comprehensive Air Quality Modeling with Extensions |
| CAPMoN | Canadian Air and Precipitation Monitoring Network |
| CEM | Continuous Emissions Monitoring |
| CIESIN | Center for International Earth Science Information Network |
| CMAQ | Community Multiscale Air Quality Modeling System |
| Comb | Combustion |
| COPD | Chronic Obstructive Pulmonary Disease |
| CSAPR | Cross-State Air Pollution Rule (U.S.) |
| DOC | Dissolved organic carbon |
| ECCC | Environment and Climate Change Canada |
| EQUATES | EPA Air Quality Time Series Project |
| Elec | Electric |
| EVE | Electric Vehicles and the Environment |
| GEM-MACH | Global Environmental Multiscale Model – Modeling Air Quality and Chemistry |
| GRPE | UNECE Working Party on Pollution and Energy |
| ICAO | International Civil Aviation Organization |
| IMO | International Maritime Organization |
| IRA | Inflation Reduction Act (U.S.) |

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|---|---|
| kg $ha^{-1}$ $yr^{-1}$ | Kilograms per Hectare per Year |
| LRTAP | Convention on Long Range Transboundary Air Pollution |
| LTM | Long-term monitoring |
| MATS | Mercury and Air Toxics Standards (U.S.) |
| MDA8 | Maximum daily 8-hour average |
| MFG | Manufacturing |
| $Mm^{-1}$ | Inverse megameters |
| MOOSE | Michigan-Ontario Ozone Source Experiment |
| MOVES3 | Motor Vehicle Emissions Simulator |
| MSAPR | Multi-Sector Air Pollutants Regulations (Canada) |
| μeq/L | Micro-equivalents per liter |
| $\mu g/m^3$ | Micrograms per cubic meter |
| $\mu mol_c\ L^{-1}$ | Micro-mol charge equivalent per liter |
| MWe | Megawatts electrical |
| NAAMC | National Ambient Air Monitoring Conference (U.S.) |
| NAAQS | National Ambient Air Quality Standards (U.S.) |
| NADP | National Atmospheric Deposition Program (U.S.) |
| NAPS | National Air Pollution Surveillance (Canada) |
| NASA | National Aeronautics and Space Administration (U.S.) |
| NEI | National Emission Inventory (U.S.) |
| $NH_3$ | Ammonia |
| $NH_4^+$ | Ammonium |
| $NO_3^-$ | Nitrate |
| $NO_X$ | Nitrogen oxides |
| NOAA | National Oceanic and Atmospheric Administration (U.S.) |
| NSPS | New source performance standards (U.S.) |
| $nssSO_4^{-2}$ | Non-sea-salt sulfate |
| OSAT | Ozone Source Apportionment Technology |
| PA | Policy Assessment (U.S.) |
| PA16 | PA nominal present for base year 2016 (U.S.) |
| PEMA | Pollutant Emission Management Area (AQA Ozone Annex) |
| pH | Potential hydrogen |
| PM | Particulate matter |
| $PM_{2.5}$ | Fine particulate matter; inhalable particles generally 2.5 micrometers in diameter or smaller |
| $PM_{10}$ | Inhalable particles generally 10 micrometers in diameter or smaller |
| ppb | Parts per billion by volume |
| PSAT | Particulate matter source attribution technology |
| RAQDRS | Regional Air Quality Deterministic Reforecast System (Canada) |
| RCU | Final Revised Cross State Air Pollution Rule Update (U.S.) |
| RCU16 | RCU modeling for year 2016 (U.S.) |
| RCU23 | RCU projections for year 2023 (U.S.) |
| RCU28 | RCU projections for year 2028 (U.S.) |
| RHR | Regional Haze Rule (U.S.) |

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|---|---|
| RHR16 | Regional Haze Rule modeling for 2016 (U.S.) |
| RHR28 | Regional Haze Rule modeling for 2028 (U.S.) |
| RWC | Residential Wood Combustion |
| SC1 | AQC Subcommittee on Program Monitoring and Reporting |
| SC2 | AQC Subcommittee on Scientific Cooperation |
| SIP | State Implementation Plan (U.S.) |
| $SO_2$ | Sulfur Dioxide |
| $SO_4^{2-}$ | Sulfate |
| SSWC | Steady-State Water Chemistry model |
| TDEP | Total deposition |
| TEMPO | Tropospheric Emissions: Monitoring of Pollution |
| TSD | Technical Support Documentation |
| UNECE | United National Economic Commission for Europe |
| UOG | Upstream Oil and Gas |
| U.S. | United States of America |
| U.S. EPA | United States Environmental Protection Agency |
| Util | Utility |
| VOC | Volatile Organic Compound |
| WHO | World Health Organization |
| WMO | World Meteorological Organization |

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# 1 Introduction

## 1.1 Historical Context

In the 1970s, the forests and lakes of North America began to show damage from acid rain caused by emissions of sulfur dioxide ($SO_2$) and oxides of nitrogen ($NO_X$). Recognizing that the transboundary flow of these pollutants was an important contributor to acid rain, on March 13, 1991, Canada and the United States (U.S.) (also referred to as 'the Parties') signed the Canada-United States Air Quality Agreement (AQA; herein referred to as 'the Agreement' or 'AQA'), a treaty-level agreement that commits both countries to reducing emissions and impacts of transboundary air pollution. The purpose of the Agreement is to establish "a practical and effective instrument to address shared concerns regarding transboundary air pollution" with the objective "to control transboundary air pollution between the two countries"(Canada-United States Air Quality Agreement, 1991).

The main body of the Agreement addresses issues such as general objectives, roles and responsibilities of the Parties, methods for exchanging information, and undertaking assessment and consultation. It establishes the framework for the obligations of both countries to deal with transboundary air pollution.

The original AQA included two annexes:

(1) *Specific Objectives Concerning $SO_2$ and $NO_X$* (Annex 1) contains specific commitments for Canada and the U.S. to reduce nationwide emissions of the precursors of acid rain – $SO_2$ and $NO_X$.

(2) *Scientific and Technical Activities and Economic Research* (Annex 2) contains guidelines for collaboration in scientific, technical activities and economic research, monitoring activities, and the exchange of information related to air quality, acid deposition, and other areas of mutual interest.

In 2000, the AQA was amended to include *Specific Objectives Concerning Ground-level Ozone Precursors* (Annex 3) (Canada-US Air Quality Agreement: Ozone Annex, 2000), aiming to address the problem of transboundary ground-level ozone, a key component of smog. Smog is a term that describes a fog or haze combined with smoke and other atmospheric pollutants. Poor air quality due to smog is often associated with reduced visibility and increased incidences of respiratory-related illnesses such as asthma. Smog is composed of a mixture of air pollutants, but its two main components are ground-level ozone and particulate matter (PM).

In 2000, Canada and the U.S. also updated Annex 2 to include further guidelines on cooperation and information exchange related to emissions trading, outreach activities, and data sharing on ground-level ozone and its precursors.

## 1.2 Environmental Context

Transboundary air pollution refers to emissions of air pollutants that are released in one jurisdiction and then transported or moved by winds and weather systems into another. Transboundary flows include pollutants directly emitted into the air (*i.e.*, primary pollutants) and those that transform into different substances via a chemical reaction in the air (*i.e.*, secondary pollutants). Many countries are both sources and receptors for transboundary air pollution (Kauffmann & Saffirio, 2020). The AQA was established to address transboundary air pollution between the two countries. Due to prevailing winds and large emissions sources, U.S. emissions of air pollutants can affect air quality in certain regions of Canada as shown by Canadian modeling scenarios. Transboundary transport of pollutants occurs across the length of the Canada-U.S. border but has a greater impact on air quality over southern Ontario, Quebec, and Atlantic Canada (AMC, 2021a). However, recent episodes of wildfires in both countries have highlighted the potential for wildfire smoke to exert adverse effects across the length of the border (Albores et al., 2023; NOAA, 2023).

Air pollution is the most important environmental contributor to the global burden of disease, leading to an estimated 6 to 7 million premature deaths annually and large economic losses ($5.1 trillion U.S. dollars or 6.6% of the global world product) (UNEP, 2019). In 2020 the Health Effects Institute reported that air pollution was the fourth leading risk factor for early death worldwide in 2019. The Institute for Health Metrics Evaluation has calculated that in 2019, air pollution worldwide contributed to an estimated 6.67 million premature deaths each year, and 213 million disability adjusted life years lost. In this analysis ambient fine particulate matter ($PM_{2.5}$) accounts for 4.14 million premature deaths, household (indoor) air pollution accounts for 2.31 million premature deaths, and ozone accounts for an estimated 365,000 premature deaths (Health Effects Institute, 2020).

Although Canada and the U.S.'s overall air quality is relatively good compared to that of other developed nations, several recent studies indicate that air pollution increases the risk of mortality even at low ambient concentrations (Brunekreef et al., 2021; Crouse et al., 2015; Pappin et al., 2019; Pinault et al., 2017). Researchers looked at the effects of low ambient concentrations of $PM_{2.5}$ in 68.5 million older Americans, finding an estimated 6% to 8% increased risk of mortality per 10 $\mu g/m^3$ (annual average) of $PM_{2.5}$ in a low exposure sub-group (Dominici et al., 2022). Health Canada estimates that air pollution from human and natural sources in North America contributes to 15,300 premature deaths per year in Canada, as well as 2.7 million asthma symptom days, and 35 million acute respiratory symptom days per year, with a total economic cost approximately $120 billion Canadian dollars (Health Canada, 2021). The Global Burden of Disease Study estimates that air pollution contributes to 602,000 premature deaths in the U.S. Ten percent of those estimated deaths are caused by chronic obstructive pulmonary disease, 4% from lung cancer, and 3% from lower respiratory infections (Global Burden of Disease Collaborative Network, 2021).

Air emissions also have impact on the environment through deposition. Acid deposition (wet or dry deposition of acidic compounds) removes essential nutrients from soils via leaching and mobilizes toxic aluminum. This loss of nutrients negatively affects the health and growth of trees and depletes the capacity of soils to neutralize future loadings of acid deposition. As such, acid deposition can contribute to declining growth rates and increased death rates in trees and a reduction in biodiversity (e.g., Clark et al., 2019). Detailed descriptions of the impacts of acid deposition on aquatic and terrestrial ecosystems in North America are available in assessment documents from the U.S. (Burns et al., 2011) and Canada (Environment Canada, 2005).

Since establishing the AQA, Canada and the U.S. have both achieved significant reductions in emissions of $SO_2$ and $NO_X$, the two major pollutants leading to acidic deposition. Since 2000, the Parties also have made further progress in reducing emissions of $NO_X$ and volatile organic compounds (VOCs) to address ground-level ozone in the Canada-U.S. border region[1]. In 2007, Canada and the U.S. achieved the emissions reduction targets laid out in both the acid rain and ozone annexes, and these emissions have continued to decrease in the subsequent years. These emissions reductions have led to lower levels of

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[1] See Table 2-1 and Table 3-1. for details on the Annex 1 and Annex 3 commitments, and areas covered by the "border region."

acid rain and ambient ground-level ozone, as well as decreased levels of ambient PM (ECCC & US EPA, 2023).

Over the past three decades, the global environmental context, including that specific to North America, has shifted substantially. For instance, the impacts of climate change are increasingly apparent worldwide, our understanding of climate change and its impacts has evolved, and climate change itself has accelerated. Climate change is linked to changes in air quality – including changes in ozone and PM concentrations – through higher temperatures, increasingly common slow-moving high-pressure weather systems, and more frequent extreme events related to rising temperatures, like wildfires (Health Canada, 2022b). In Canada and the U.S., the impacts of climate change are already being felt in many communities through more frequent and intense extreme weather and climate events. These events are expected to cause disruption and damage to infrastructure and property, and impede the rate of economic growth. Additionally, impacts on the health and well-being of the public, specifically vulnerable populations, remain a concern (USGCRP, 2018). There have also been significant changes to emissions sources. For example, there is evidence that the rapid growth in oil and gas extraction in the Bakken Formation is leading to an increase in transboundary transport of air pollutants in both directions between North Dakota, Montana, Alberta, and Saskatchewan (Prenni et al., 2016).

Both the U.S. and Canada are taking action to address new environmental challenges through new policy initiatives designed to decrease air pollution and mitigate climate change. In 2022, the U.S. announced its largest investment in combating climate change, the Inflation Reduction Act (IRA). IRA investments – along with additional investments within the 2021 Bipartisan Infrastructure Law and 2022 CHIPS and Science Act – accelerate the transition to a clean energy economy, address environmental injustice, reduce renewable energy costs, spur innovation, and are expected to reduce carbon emissions by roughly 40% by 2030. The U.S. is also advancing new regulatory efforts to address a myriad of air pollutants such as new requirements designed to address ozone transport through the Good Neighbor Rule (US EPA, 2023b). In 2022, the Government of Canada released the 2030 Emissions Reduction Plan, which provides a roadmap to reach its climate commitments, such as reducing national greenhouse gas emissions by 40 to 45% below 2005 levels by 2030 under the Paris Agreement, and achieving net-zero emissions by 2050. The Government of Canada is currently developing the *Clean Electricity Regulations* that will help drive progress towards a net-zero electricity grid by 2035. Further, the Government of Canada has proposed amendments to regulations[2] which require manufacturers and importers to meet specified annual targets of zero-emission vehicles. Canada has also amended the *Canadian Environmental Protection Act*, 1999 (Canada, 2023) to recognize that every individual in Canada has a right to a healthy environment, which includes consideration of the principle of environmental justice. Many of these landmark policies are not captured in the projections included in this report, but will, directly or indirectly, impact emissions across the economies of both countries.

## 1.3 Objectives of the Review and Assessment

Article X of the AQA calls for Canada and the U.S. to conduct a comprehensive review and assessment of the Agreement and its implementation every five years unless otherwise agreed upon. The last review and assessment was completed in 2012, and was included as a section in the 2012 biennial AQA

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[2] The proposed *Regulations Amending the Passenger Automobile and Light Truck Greenhouse Gas Emission* new requirements for manufacturers and importers to ensure that their fleet of new light-duty vehicles offered for sale in Canada meets specified annual targets of zero-emission vehicles.

progress report (Environment Canada & US EPA, 2012). The 2012 review recommended that consideration be given to: streamlining the reporting process under the AQA; expanding the scope of the Agreement to address transboundary PM; and addressing transboundary air quality issues in the western border area, if the science demonstrated that there were issues of concern in this area. The ongoing threat to human health, the environment, and the economy posed by air quality issues, particularly in the context of ongoing global warming with its potential to intensify air quality issues, further motivated the need for a review and assessment at the time.

In November 2020, the Canada-U.S. Air Quality Committee (AQC), which oversees implementation of the AQA, agreed to undertake an exercise to define the scope and content of a potential review and assessment of the AQA. This exercise focused on reviewing current Annexes covering acid rain and ground-level ozone and evaluating transboundary impacts from $PM_{2.5}$, which is not currently part of the AQA. After defining the scope of an AQA review and assessment in the spring of 2021, the AQC Subcommittee on Program Monitoring and Reporting (SC1), in consultation with the AQC Subcommittee on Scientific Cooperation (SC2), finalized plans to undertake a new review of the AQA, its effectiveness, and potential gaps.

The objectives of this review and assessment are to:

- Review what the AQA has accomplished to date, including whether it is meeting its current objectives and whether emission reductions mandated by the Agreement have met the AQA objectives to: reduce the transboundary flow of air pollution, reduce acid deposition, reduce concentrations of ground-level ozone, and improve air quality in Canada and the U.S.;
- Assess whether the emissions reduction targets and measures included in Annex 1 and in Annex 3 and the commitments in Annex 2 remain appropriate for Canada/U.S. policy and science needs; and
- Examine whether new actions such as commitments and/or measures would be appropriate (*e.g.*, for pollutants included under the Agreement and those not currently addressed, such as $PM_{2.5}$).

## 1.4 Approach for the Review and Assessment

To assess the effectiveness of actions under Annex 1 and Annex 3, and to inform future work under the AQA, emissions, monitoring, and modeling data were evaluated. Emissions data from the Canadian Air Pollutant Emissions Inventory (APEI) and the U.S. National Emissions Inventory (NEI) were used to evaluate historical changes since the AQA came into effect. Note that the most recent year for which data are available for this report, the year 2020, was marked by the COVID-19 pandemic, which coincided with changes in emissions for many pollutants. Wet and dry deposition were measured by the Canadian Air and Precipitation Monitoring Network (CAPMoN), the Alberta Precipitation Quality Monitoring Program, and the U. S. National Atmospheric Deposition Program (NADP). Ozone and $PM_{2.5}$ were measured by the Canadian National Air Pollution Surveillance Program (NAPS) and U.S. air monitoring networks included in the Air Quality System (AQS). The monitoring data were used to evaluate historical trends and current concentrations and deposition. Air quality and deposition modeling were conducted to estimate future concentrations and deposition, and to attribute the relative influence of Canadian and U.S. emissions at a given location. Four modeling datasets were considered, including new modeling performed by Environment and Climate Change Canada (ECCC) and modeling from previous U.S. Environmental Protection Agency (U.S. EPA) studies[3], as described in Appendix A. Instead of performing new joint modeling for this report, both countries agreed to use existing modeling if appropriate, and to perform new separate modeling runs as needed. The U.S. and ECCC model runs are not directly comparable, as they consider different emissions inventories and time periods. However, the modeling runs can be considered together qualitatively to gain confidence in the results. The modeling runs do not include wildfire emissions, as the focus of the current AQA is on the transboundary impact of emission sources that can be directly addressed through targeted measures by each country. The ECCC air quality modeling output was also used with health and agricultural models to estimate the impacts of transboundary pollution.

The results of these evaluations are presented for Annex 1 commitments on acid rain (Section 2), Annex 3 commitments on ozone (Section 3), as well as for $PM_{2.5}$ (Section 4). A review of Annex 2 on scientific and technical cooperation (Section 5) was also conducted. Per Article X of the Agreement, in the context of this review and assessment, the Parties shall consider such action as may be appropriate, including the modification of the AQA and/or the modification of existing policies, programs, and measures. Key findings (Section 6) and recommendations for further collaboration (Section 6.3) are also outlined.

# 2 Acid Rain

*What is acid rain:* Acid deposition is the removal of acidic compounds from the atmosphere by the process of wet deposition (precipitation and fog) and dry deposition (transfer of gases and particles to the earth's surface). Wet deposition is more commonly known as acid rain. Emissions of $SO_2$ and $NO_X$ from power plants, transportation, industries, and other sources, react in the atmosphere with oxidants to form various acidic compounds, notably sulfuric acid and nitric acid. These acidic compounds can react with ammonia ($NH_3$) to form secondary inorganic species, such as particle sulfate ($SO_4^{2-}$) and particle nitrate ($NO_3^-$), which are also major components of $PM_{2.5}$ (Section 4). Once deposited (in either gas or particle form) to surfaces, the acidic compounds harm aquatic and terrestrial ecosystems (particularly forests) and damage surfaces of buildings or other man-made structures.

*Acid rain in the AQA:* Annex 1, the Acid Rain Annex to the AQA (see Table 2-1) established commitments by Canada and the U.S. to reduce emissions of $SO_2$ and $NO_X$, the primary precursors to acid rain, from stationary and mobile sources. Both Canada and the U.S. have met their commitments under the Acid Rain Annex, as described in Section 2.1. Specific details on when commitments were achieved over the past three decades can be found in the biennial progress reports. Between 1990 and 2020, Canada's total emissions of $SO_2$ and $NO_X$ decreased by 78% and 36%, respectively. The U.S. total emissions reductions for the same timeframe for $SO_2$ and $NO_X$ were 93% and 70%, respectively. Reductions in $SO_2$ and $NO_X$ emissions in both Canada and the U.S. since 1990 have led to major decreases in the wet deposition of $SO_4^{2-}$ and $NO_3^-$ over the eastern half of the two countries. Implementation of various regulatory and non-regulatory actions for more than two decades in Canada has significantly reduced emissions of $SO_2$ and $NO_X$ and ambient concentrations. Similar measures, especially regulatory programs in the electric power sector, have significantly reduced emissions of $SO_2$ and $NO_X$ and ambient concentrations in the U.S.

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[3] Notable recent relevant U.S. regulatory and legislative actions such as new Heavy-Duty Highway Standards, the Bipartisan Infrastructure Law, and the IRA are not included in U.S. modeling presented in this report.

*Table 2-1. Acid Rain (Annex 1): Specific objectives concerning $SO_2$ and $NO_X$*[4]

| Objectives | Canada | U.S. |
|---|---|---|
| **$SO_2$** | • By 1994, reduce annual $SO_2$ emissions in the seven easternmost provinces to 2.3 million metric tons.[5]<br>• From 1995 to 1999, an annual emissions cap in the seven easternmost provinces at 2.3 million metric tons of $SO_2$.<br>• By 2000, permanent national emissions cap of 3.2 million metric tons per year of $SO_2$. | • By 2000, reduce annual $SO_2$ emissions by approximately 10 million short tons[6] from 1980 levels, taking into account credits ('allowances") earned for reductions from 1995 to 1999.<br>• By 2010, permanent national emissions cap of 8.95 million short tons of $SO_2$ per year for electric utilities.<br>• Beginning in 1995, national $SO_2$ emissions cap of 5.6 million short tons for industrial sources. |
| **$NO_X$** | • By 2000, reduce annual national stationary source $NO_X$ emissions of 100,000 metric tons below the year 2000 forecast level of 970,000 metric tons.<br>• By 1995, develop further annual national emission reduction requirements from stationary sources to be achieved by 2000 and/or 2005.<br>• Implement a more stringent mobile source $NO_X$ control program. | • By 2000, reduction of total annual emissions of $NO_X$ by 2 million short tons.<br>• Implement stationary source control program for electric utility boilers.<br>• Implement a mobile source $NO_X$ control program. |
| **Compliance Monitoring** | • By 1995, estimate $SO_2$ and $NO_X$ emissions from new electric utility units and existing electric utility units greater than 25 MWe (megawatts electrical) using methodologies like continuous emissions monitoring (CEM) and investigate feasibility of using and implementing CEM, where appropriate.<br>• Work towards utilizing comparably effective methods of emission estimation for $SO_2$ and $NO_X$ emissions from all major industrial boilers and process sources, including smelters. | • By 1995, require new electric utility units and existing units greater than 25MWe operate CEM systems.<br>• Work towards utilizing comparably effective methods of emission estimation for $SO_2$ and $NO_X$ from all major industrial boilers and process sources, including smelters. |
| **Prevention of Air Quality** | • By 1995, develop and implement means (comparable to those in the | • Maintain means for preventing significant air quality deterioration and protecting |

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[4] The complete text of the AQA and Annexes 1 and 2 can be found at https://www.canada.ca/en/environment-climate-change/services/air-pollution/issues/transboundary/canada-united-states-air-quality-agreement.html.

[5] One metric ton is equal to approximately 1.1 short tons.

[6] One short ton is equal to approximately 0.91 metric tons.

| | | |
|---|---|---|
| **Deterioration and Visibility Protection** | U.S.) for achieving levels of prevention of significant air quality deterioration and protection of visibility with respect to sources that could cause significant transboundary air pollution. | visibility with respect to sources that could cause significant transboundary air pollution. |

## 2.1 Effect of Emissions Reduction Strategies on Acidifying Pollutants

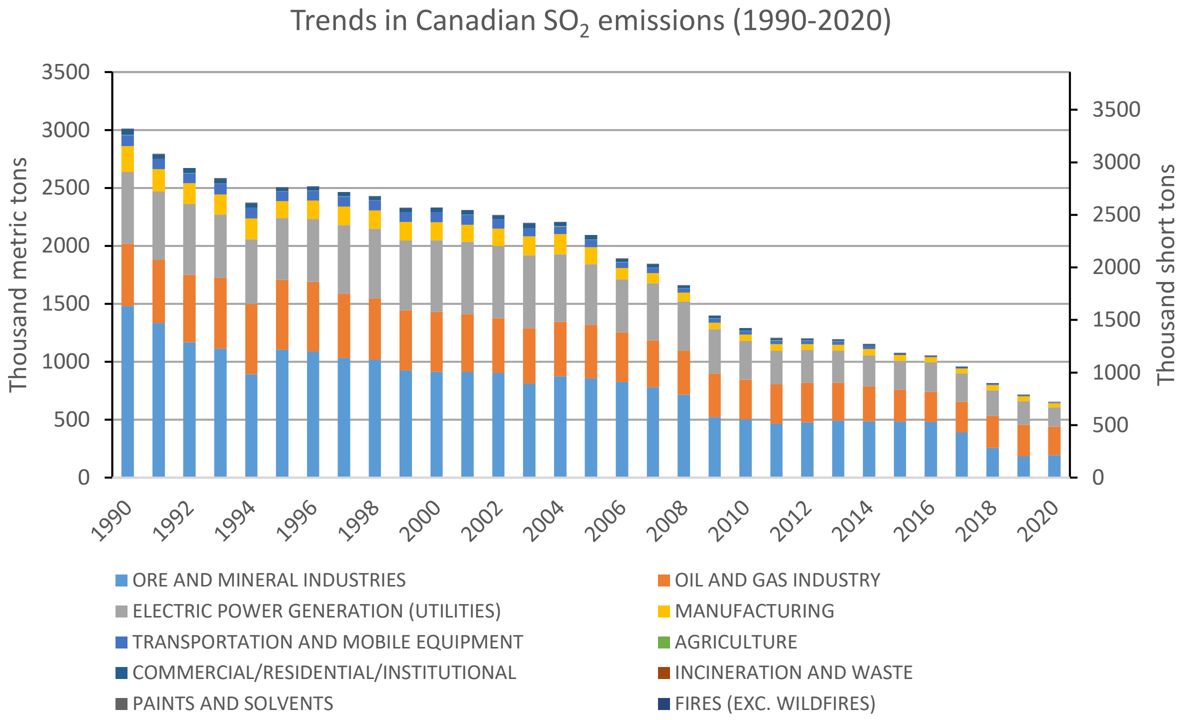
### 2.1.1 Canada

Under the Acid Rain Annex (Annex 1), Canada agreed to a permanent national $SO_2$ emissions cap of 3.2 million metric tons per year by 2000. Canada also agreed to reduce annual $NO_X$ emissions from power plants, major combustion sources, and metal smelting operations by 100 thousand metric tons below the forecasted level of 970 thousand metric tons by 2000. Canada met these commitments through efforts undertaken by the federal government and eastern provinces as part of the 1985 Eastern Canada Acid Rain Control Program and Canada-wide Acid Rain Strategy for Post-2000, which was adopted in 1998. The 1985 Eastern Canada Acid Rain Control Program committed Canada to cap total $SO_2$ emissions in the seven provinces from Manitoba eastward at 2.3 million metric tons by 1994. Canada met this cap in 1993. Under the AQA, this cap was extended to cover the period 1994-1999.

The Canada-wide Acid Rain Strategy for Post-2000 put in place a framework for addressing the issues related to acid rain with the goal of ensuring that the deposition of acidifying pollutants does not further deteriorate the environment in eastern Canada and that new acid rain problems do not occur elsewhere in Canada. As part of this Strategy, the provinces of Ontario, Quebec, New Brunswick, and Nova Scotia committed to an additional 50% reduction in their $SO_2$ emissions beyond their 1985 Eastern Canada Acid Rain Program targets by 2010 (by 2015 for Ontario). All four provinces have met the stricter $SO_2$ emissions targets established under the Strategy. Canada continues to develop measures to reduce emissions that contribute to acid rain and smog. In 2016, Canada published the Multi-sector Air Pollutants Regulations (MSAPR), which includes limits on $SO_2$ emissions from cement manufacturing facilities. These regulations established Canada's first mandatory national air pollutant emission standards for major industrial facilities, and are a key element of Canada's Air Quality Management System (AQMS).

As of 2020, Canada's total $SO_2$ emissions were approximately 651 thousand metric tons, about 80% below the national cap of 3.2 million metric tons. Overall, $SO_2$ emissions decreased by 78% (2.4 million metric tons) between 1990 and 2020 (Figure 2-1). Reductions in emissions from the Ore and Mineral industries, and in particular the Non-Ferrous Refining and Smelting Industry sector, were the largest driver of this downward trend, particularly in the early 1990s, and again from 2008 to 2020. The decrease in $SO_2$ emissions since 2008 can be attributed to the preparation and implementation of pollution prevention plans by facilities, the installation of new technology or processes at facilities, the closure of four major smelters in Manitoba, Ontario, Quebec and New Brunswick, and facilities achieving Base Level Industrial Emissions Requirements (BLIERs) through environmental performance agreements (ECCC, 2017, 2018). Emissions from Electric Power Generation (Utilities) decreased significantly from 2005 to 2020, primarily owing to the closure of, or improvements to, generating stations burning heavy fuel oil. Improvements consisted of installing pollution control equipment or switching to low sulfur heavy fuel oil. Furthermore, Coal-fired electric power generation saw an important $SO_2$ emission decrease of 19% (37 thousand metric tons) between 2019 and 2020, attributed to a decrease in coal consumption.

*Figure 2-1. Canadian $SO_2$ emission trends for 1990-2020.*

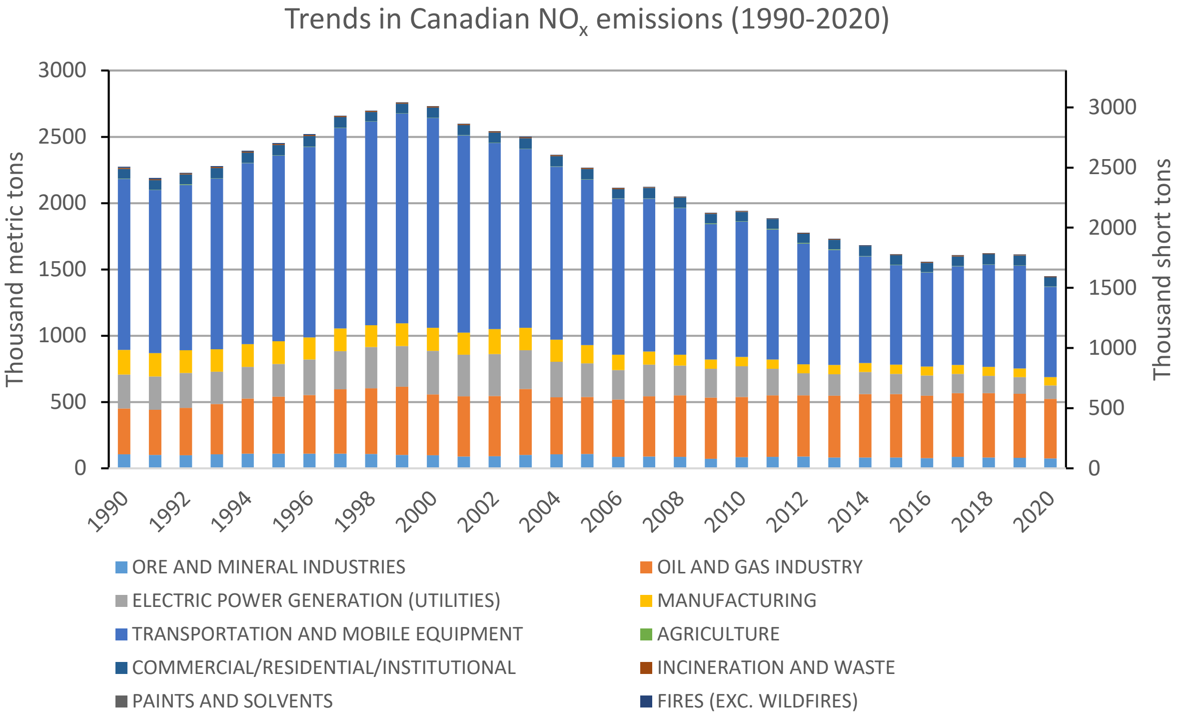


Data source: Canada's Air Pollutant Emissions Inventory 1990-2020 *(ECCC, 2022)*

Canada has also met its 2000 commitment to reduce $NO_X$ emissions from power plants, major combustion sources, and metal smelting operations by 100 thousand metric tons below the forecasted level of 970 thousand metric tons (*i.e.*, cap set at 870 thousand metric tons). Recent measures in Canada's 2016 MSAPRs further limit $NO_X$ emissions from industrial boilers, heaters, stationary gaseous fuel-fired engines, and cement manufacturing facilities.

Between 1990 and 2020, Canada's total $NO_X$ emissions decreased by 36% (826 thousand metric tons) (Figure 2-2) (ECCC, 2022). The most significant changes in $NO_X$ emissions include a decrease of 47% (607 thousand metric tons) from Transportation and Mobile Equipment, a decrease of 61% (156 thousand metric tons) from Electric Power Generation, and an increase of 30% (103 thousand metric tons) from the Oil and Gas industry. A more comprehensive discussion of $SO_2$ and $NO_X$ emissions in Canada can be found in Canada's APEI Report for 2022 (ECCC, 2022).

*Figure 2-2. Canadian $NO_X$ emission trends for 1990-2020.*



Data source: Canada's Air Pollutant Emissions Inventory 1990-2020 *(ECCC, 2022)*

### 2.1.2 United States

The U.S. met its commitments to reduce $SO_2$ and $NO_X$ under the Acid Rain Annex (Annex 1). The national Acid Rain Program (ARP) has dramatically cut power plant emissions of $SO_2$ and $NO_X$, reducing acid rain as well as secondary formation of $PM_{2.5}$. Further reductions in power plant pollution have been achieved by state and U.S. EPA efforts to cut interstate air pollution, which also helped downwind states meet health-based air quality standards for fine particles and ozone. The Clean Air Interstate Rule (CAIR) achieved large reductions in power plant annual $SO_2$ and $NO_X$ emissions, as well as additional summertime $NO_X$ reductions beyond those required by the 1998 $NO_X$ State Implementation Plan (SIP) Call. In 2015, CAIR was replaced by the Cross-State Air Pollution Rule (CSAPR). In addition, the Mercury and Air Toxics Standards (MATS) rule, which went into effect in April 2015, achieved substantial $SO_2$ emissions reductions as an additional benefit to air toxics emissions reductions from the power sector. These reductions occurred while the demand for electricity increased and were the result of continued increases in efficiency, installation of state-of-the-art pollution controls, and the switch to lower emitting fuels. These regulatory programs along with economic forces, contributed to a decrease in $SO_2$ and $NO_X$ emissions by 93% and 70%, respectively, between 1990 and 2020. In addition to control programs mentioned above, the achievement of greater efficiency in energy production and the use of lower emitting fuels have reduced emissions.

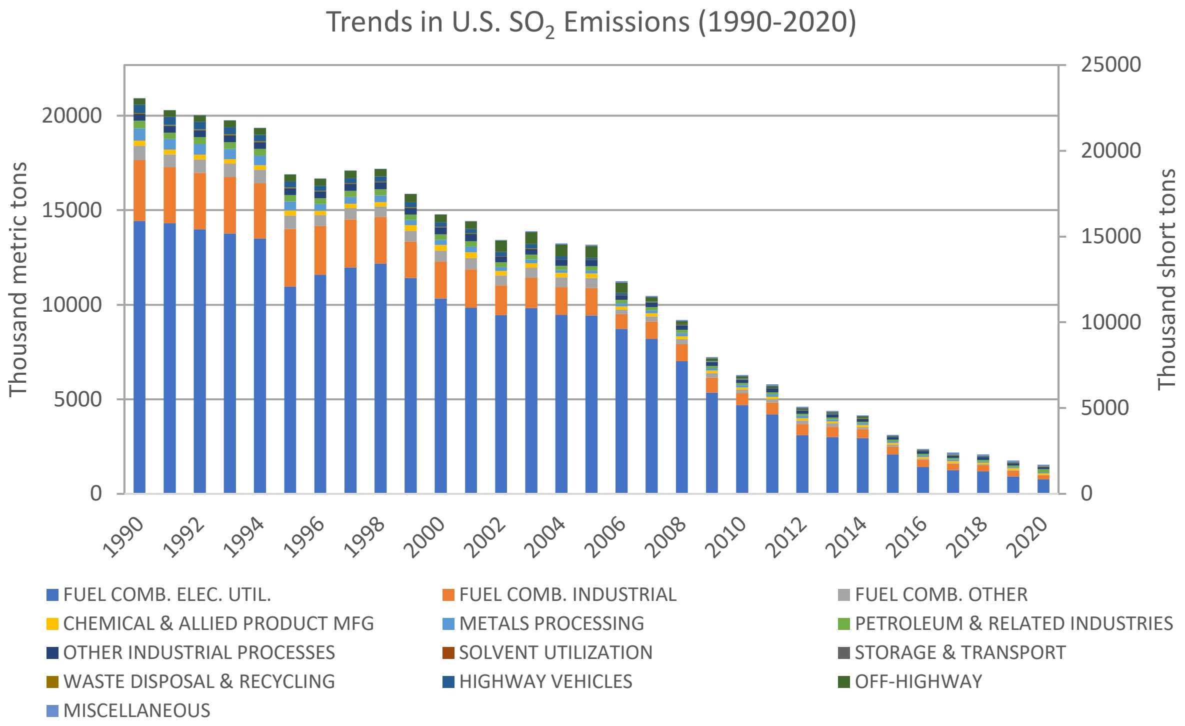
The Clean Air Act requires that when new industrial facilities are designed and built, good pollution control must be part of the design. In areas not meeting the NAAQS, to avoid making pollution worse, new and modified large plants and factories must meet the lowest achievable emission rate and obtain offsetting emissions reductions from other sources. In areas that meet the NAAQS, new and modified large plants and factories must apply the best available control technology, considering cost and other factors, and avoid causing significant degradation of air quality or visibility impairment in national parks. For example, natural gas fired units reduce $NO_X$ emissions through flue gas recirculation and low $NO_X$ burners, reducing $NO_X$ by 60 to 90%.

In the transportation sector, which includes highway and non-road vehicles, U.S. EPA has required significant reductions over the years in emissions from new motor vehicles and non-road engines through standards that require a combination of cleaner engine technologies and cleaner fuels. Dating back to the mid-1970s/early 1980s when U.S. EPA established the very first emission standards for new motor vehicles, each decade since has brought forward new more stringent emission standards and cleaner fuels for the fleet. To highlight a few accomplishments, during the 1980s U.S. EPA established Inspection and Maintenance programs for motor vehicles and finalized regulations to remove lead from gasoline. During the 1990s U.S. EPA imposed limits on diesel fuel sulfur content and finalized new emission standards for diesel engines used in construction and agricultural equipment. In the early 2000s U.S. EPA finalized regulations for small non-road handheld engines such as trimmers and chainsaws and also finalized regulations to reduce air toxics from mobile sources. As the most recent example, in December 2021, U.S. EPA finalized standards for light-duty vehicles (US EPA, 2021b) that will reduce emissions, while bolstering energy security and encouraging manufacturing innovation. The U.S. EPA also recently finalized new heavy-duty engine and vehicle standards (US EPA, 2022b) in December 2022. The rule sets stronger emission standards to further reduce air pollution from heavy-duty vehicles and engines starting in model year 2027. The final program includes new, more stringent emissions standards that cover a wider range of heavy-duty engine operating conditions compared to today's standards, and it requires these more stringent emissions standards to be met for a longer period of time when these engines operate on the road.

Figure 2-3 shows trends in U.S. $SO_2$ emissions. The greatest reduction in $SO_2$ emissions between 1990 and 2020 is from stationary fuel combustion (95%). In 2020, electric utility fuel combustion contributed 49% of the total $SO_2$ emissions (1.7 million short tons). The transportation sector experienced a 97% decrease in total $SO_2$ emissions (42,000 short tons).

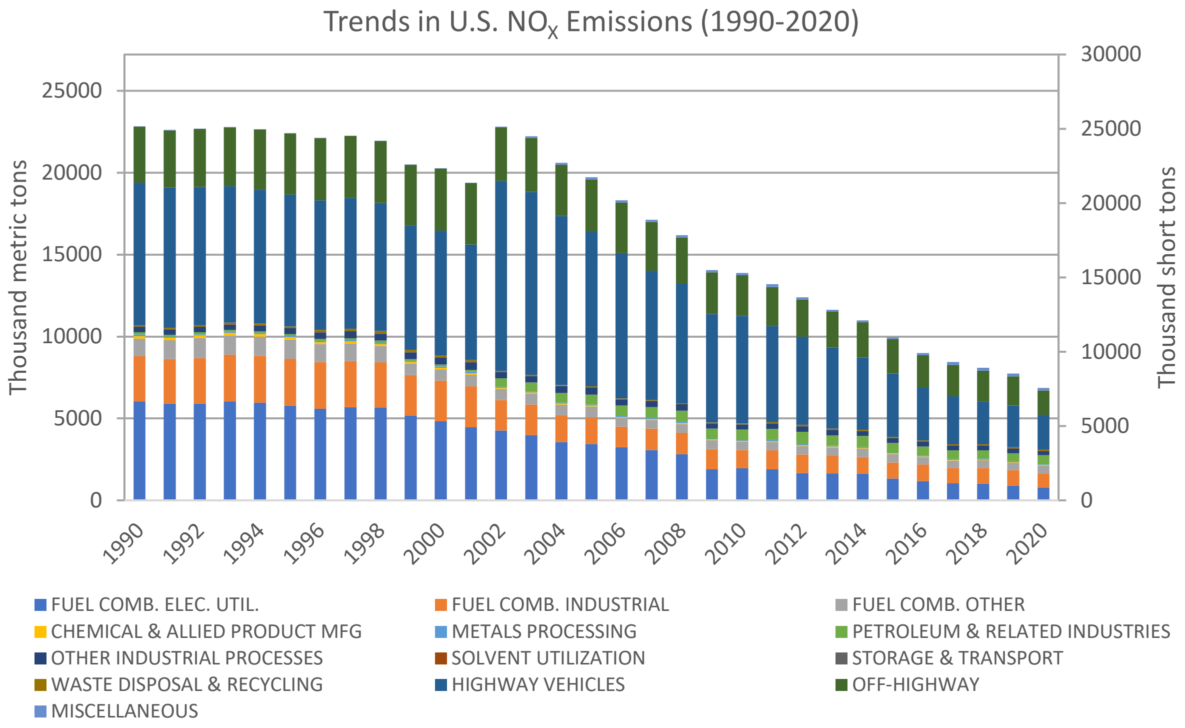
Figure 2-4 shows U.S. $NO_X$ emission trends. Total $NO_X$ emissions from electric utility fuel combustion decreased 87% from 1990 to 2020. Petroleum and Related Industries noted an increase in $NO_X$ emissions to a peak in 2012 but have decreased 28% since then. Transportation $NO_X$ emissions have decreased 70% from 1990 to 2020. The methods used to compute all emissions categories were updated starting with the year 2002. A different version of U.S. EPA's Motor Vehicle Emissions Simulator (MOVES3) was used starting 2001 for highway vehicles, emission factors, and activity data accounts for the increase in emissions from 2001 to 2002.

*Figure 2-3. U.S. $SO_2$ emission trends for 1990-2020.*



Data source: 2020 U.S. National Emissions Inventory (US EPA, 2023a).

*Figure 2-4. U.S. $NO_X$ emission trends for 1990-2020.*



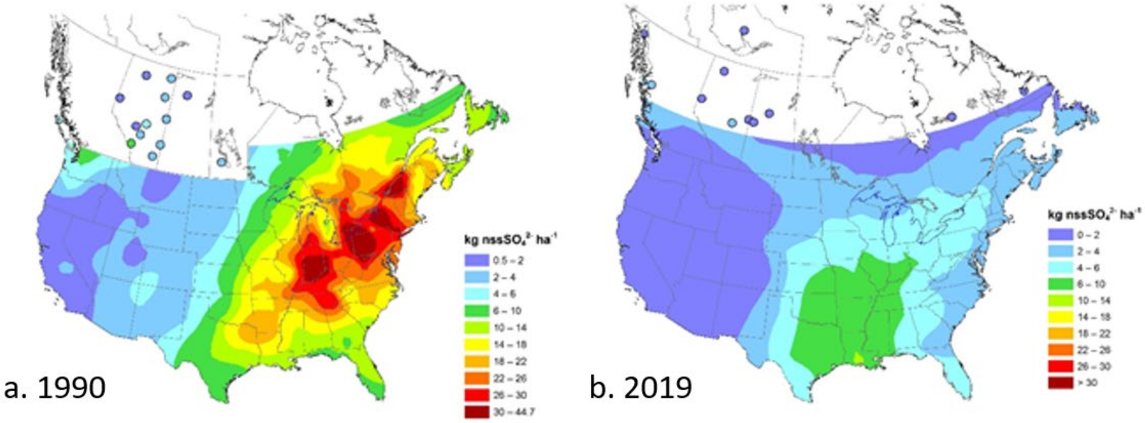
Data source: 2020 U.S. National Emissions Inventory (US EPA, 2023a).

## 2.2 Current Levels and Trends in Acid Deposition

Wet deposition of $SO_4^{2-}$ and $NO_3^-$ is measured by precipitation chemistry monitoring networks in Canada and the U.S. The measurement data, presented in kilograms per hectare per year (kg $ha^{-1}$ $yr^{-1}$), are the basis for binational spatial wet deposition maps.

Figure 2-5 shows the spatial patterns of annual wet $SO_4^{2-}$ deposition of non-sea-salt sulfate ($nssSO_4^{2-}$), in 1990 and 2019 along with point values at sites in less densely measured regions. Non-sea-salt sulfate is the measured $SO_4^{2-}$ with the contribution from sea salt $SO_4^{2-}$ removed (WMO, 2004) for sites within 100 km of an ocean. The interpolation for 2019 was extended over southwestern Canada because of the increased site density in this region compared to 1990. The lower Great Lakes region received the highest wet deposition of $SO_4^{2-}$ early in the 30-year period, while recently the maximum has shifted to the Mississippi valley. In 1990, $SO_4^{2-}$ deposition exceeded 26 kg $nssSO_4^{2-}$ $ha^{-1}$ over a large area of eastern North America. In 2019, only a small area in southern Louisiana exceeded 10 kg $nssSO_4^{2-}$ $ha^{-1}$. It is noteworthy that while $SO_4^{2-}$ deposition decreases were most pronounced in the east, all areas of the domain exhibited decreased impacts.

*Figure 2-5. Annual $nssSO_4^{2-}$ wet deposition for (a) 1990 and (b) 2019.*



a. 1990

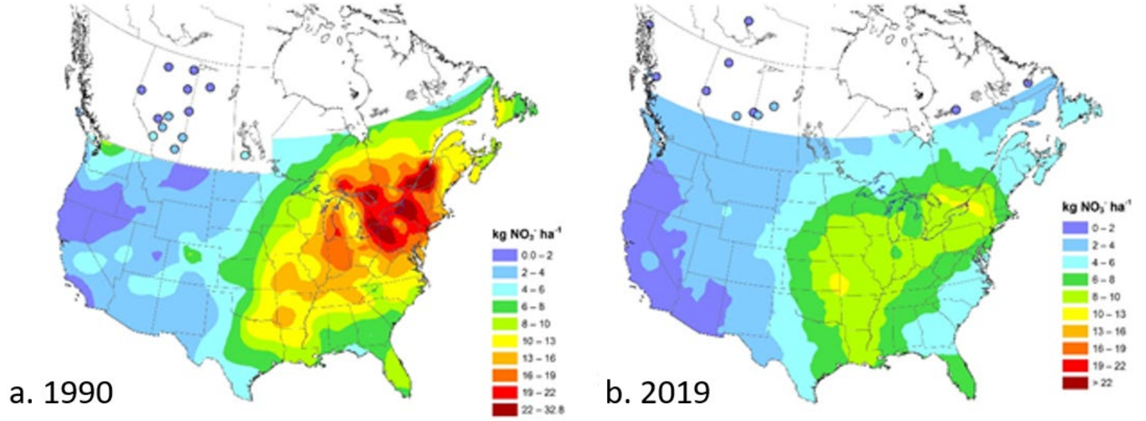
b. 2019

Data sources: CAPMoN, the Alberta Precipitation Quality Monitoring Program (both at https://doi.org/10.18164/72bef1bc-709a-4d57-99ea-6969b9728335), and the U.S. NADP (http://nadp.slh.wisc.edu/).

Figure 2-6 shows the patterns of wet $NO_3^-$ deposition in 1990 and 2019, along with point values at sites in less densely measured regions. Similar to $SO_4^{2-}$, the lower Great Lakes region consistently received the highest wet deposition of $NO_3^-$ early in the 30-year period. $NO_3^-$ deposition exceeded 19 kg $NO_3^-$ $ha^{-1}$ in many parts of the northeastern U.S. and southern Ontario and Quebec in 1990. In 2019, $NO_3^-$ deposition was less than 13 kg $NO_3^-$ $ha^{-1}$ throughout North America. The steep declines in $NO_3^-$ wet deposition after the year 2000 are due to major $NO_X$ emission reductions in both countries. Similarly to $SO_4^{2-}$, most areas of the continent experienced benefits with regards to deposition of $NO_3^-$.

These results are consistent with other recent studies, which have found that reductions in $SO_2$ and $NO_X$ emissions in both Canada and the U.S. between 1990 and 2019 have led to decreases in the wet deposition of $SO_4^{2-}$ and $NO_3^-$ over the eastern half of both countries. Wet deposition of $SO_4^{2-}$ and $NO_3^-$ at monitoring stations in eastern Canadian and eastern U.S. decreased by 69% and 46%, respectively, from 1989 to 2016, with the decline in $NO_3^-$ wet deposition occurring primarily after 2000 (Feng et al., 2021), corresponding to the substantial declines in $NO_X$ emissions commencing in 2000. Similar decreasing trends in wet $SO_4^{2-}$ and $NO_3^-$ were observed in the U.S. Mid-Atlantic, Midwest and Northeast based on precipitation samples collected as far back as 1981 to as recently as 2019 (Baldigo et al., 2021; Burns et al., 2021; Isil et al., 2022; Likens et al., 2021; McHale et al., 2021).

*Figure 2-6. Annual wet $NO_3^-$ deposition for (a) 1990 and (b) 2019.*



Data sources: CAPMoN, the Alberta Precipitation Quality Monitoring Program (both at https://doi.org/10.18164/72bef1bc-709a-4d57-99ea-6969b9728335), and the U.S. NADP(http://nadp.slh.wisc.edu/).

Total deposition is made up of wet deposition plus dry deposition. Dry deposition is typically cost-intensive to measure and typically is more subject to uncertainties than wet deposition (Walker, Bell, et al., 2019). In order to estimate dry and subsequently total deposition, measurements of ambient concentrations are paired with modeled dry deposition velocities in measurement model fusion techniques (Fu et al., 2022), including the Total Deposition (TDep) method (Schwede & Lear, 2014) for the U.S. and the Atmospheric Deposition Analysis Generated from optimal Interpolation from Observations (ADAGIO) method (Robichaud et al., 2020) for Canada.

Dry deposition in 2018-2020 comprised 33% of U.S. total sulfur deposition on average across the U.S., which decreased from 40% from 2000 to 2002. There are areas with notable high dry sulfur deposition (exceeding 60%) along the Canada-U.S. border (northwestern and northeastern Washington state, northeastern Montana and northwestern North Dakota). Based on data from 15 CAPMoN sites collected between 2000 and 2018 (Cheng et al., 2022), dry deposition contributes to 11 to 55% and wet deposition contributes  45 to 89% of total sulfur deposition in Canada.  Using the annual deposition data from Cheng et al., 2022, the reduction in total sulfur deposition at eastern Canadian sites from 2000–2002 to 2016-2018 was 70%. There are limited data on the total nitrogen deposition budget in Canada as CAPMoN does not routinely measure $NO_2$, other oxidized forms of nitrogen, or $NH_3$ in ambient air for subsequent estimation of dry deposition fluxes. Maps of total deposition (wet plus dry) are shown for the U.S. for sulfur deposition in Figure 2-7 and nitrogen deposition in Figure 2-8. Note that similar maps of total deposition are not yet available for Canada because of ongoing development and evaluation of a model-measurement fusion deposition product. The reduction in total sulfur deposition (wet plus dry) in the eastern U.S. has been of similar magnitude to that of wet deposition with an overall average reduction of 81% from 2000–2002 to 2018–2020 (Figure 2-7). All areas of the eastern U.S. have shown significant improvement in wet $SO_4^{2-}$ deposition, with an overall 70% reduction from 2000–2002 to 2018–2020. Between 2000–2002 and 2018–2020, the Northeast and Mid-Atlantic experienced the largest reductions in wet $SO_4^{2-}$ deposition, 77% and 74% reduction, respectively. Reductions in total nitrogen deposition recorded since the early 1990s have been less pronounced than those for sulfur. The most recent Canadian and U.S. studies report that the decline in total deposition since 2010 has been slower than that of the previous decade, particularly for nitrogen (Benish et al., 2022; Cheng et al., 2022).

*Figure 2-7. Three-year average of total sulfur deposition in the U.S. for (a) 2000-2002 and (b) 2018-2020[7].*

Data source: Maps generated using the Total Deposition (TDep) method (Schwede & Lear, 2014).

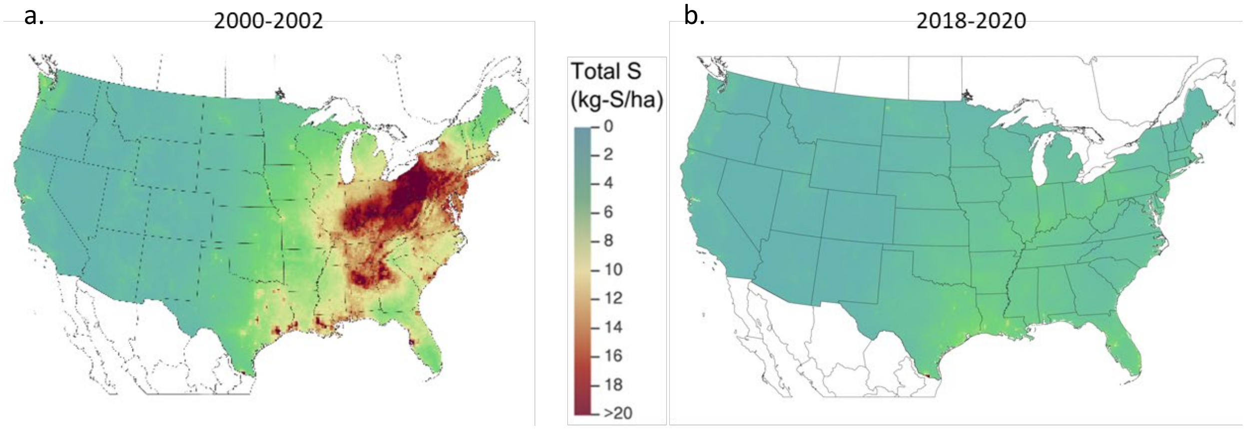
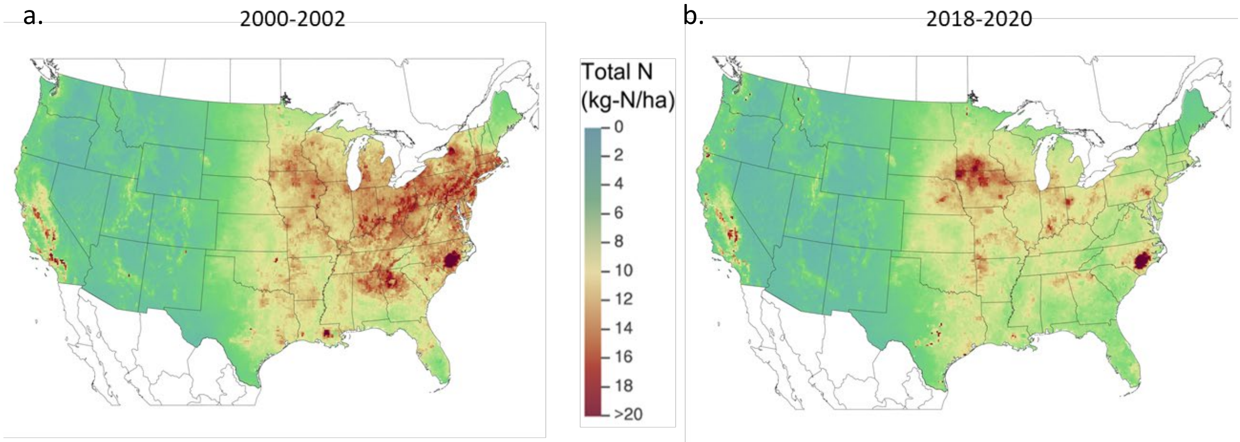


*Figure 2-8. Three-year average of total nitrogen deposition in the U.S. for (a) 2000-2002 and (b) 2018-2020[8].*



Data source: Maps generated using the Total Deposition (TDep) method (Schwede & Lear, 2014).

The increasing importance of reduced nitrogen ($NH_3$ and ammonium - $NH_4^+$) to total nitrogen deposition has been reported across both countries. In Canada, wet deposition of inorganic nitrogen ($NO_3^-$ plus $NH_4^+$) decreased on average by 25% at eastern Canadian sites from 2000–2002 to 2016–2018. However, this was entirely due to changes in $NO_3^-$, as no trends were observed in wet $NH_4^+$deposition at the majority of the sites from 2000 to 2018 (Cheng et al., 2022). In the U.S., wet deposition of inorganic nitrogen decreased an average of 19% in the Mid-Atlantic and 32% in the Northeast but increased by 17% and 9% in the North and South Central regions from 2000–2002 to 2018–2020. Increases in wet deposition of inorganic nitrogen in the North Central and South Central regions are attributed to 44% and 34% increases in wet deposition of reduced nitrogen ($NH_4^+$), respectively, between 2000 and 2020.

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[7] Estimates for dry and total deposition are provided by the TDep measurement model fusion method begin in 2000 due to availability of modeled data. In cases where measurement data is unavailable, modeled data is used.

[8] See Footnote 7

Considering total (wet + dry) deposition, decreases in oxidized nitrogen ($NO_X$) have generally been greater than increases in reduced nitrogen ($NH_X$) deposition. Total oxidized nitrogen deposition decreased 57 % in the east, which total deposition of reduced nitrogen increased by an average of 38 % in the east from 2000–2002 to 2018–2020 (NADP, 2023). Overall, long-term trends in atmospheric deposition of $NH_4^+$ and $NH_3$ in Canada and the U.S. were either observed to fluctuate with no statistically significant trend (Burns et al., 2021; Cheng et al., 2022; Feng et al., 2021; Likens et al., 2021; Zhang et al., 2018) or were increasing (Benish et al., 2022; McHale et al., 2021). The increasing contribution of reduced nitrogen to acidification, eutrophication, and air quality impacts is a topic of growing interest to scientists (Walker, Beachley, et al., 2019) and policy makers (Kanter et al., 2020).

## 2.3 Influences of Transboundary Flow

The U.S. 2020 Ozone Policy Assessment[9] (Ozone PA) (US EPA, 2020d) encompassed simulations that estimated nitrogen and sulfur deposition. These simulations used zero-out for attribution, which lends itself to a similar contribution analysis of deposition. However, the simulations were not evaluated against deposition observations and have not been fused with observations like the total deposition products in the maps shown above (US EPA, 2020d). Qualitatively, Figure 2-9 and Figure 2-10 show that contributions from each country are largest near their own population centers. In the less populated parts of northern Montana and the northern parts of the province of Ontario, there are larger contributions due to transboundary impacts. Compared to the Northeastern regions, these areas have relatively low total deposition in both the raw model results and, for Montana, in the fused maps above.

Note that Ozone PA modeling results utilized different datasets for the U.S. and Canada. There are sharp transitions in land-use classifications between grassland and farmland, particularly in the southern Alberta and northern Montana region. This sharp shift in classification could impact modeled levels of nitrogen deposition in particular, considering the relatively large levels of nitrogen associated with cropland. Further verification of land-use classifications and their representation in modeled outcomes could be pursued by both countries in future work.

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[9] The U.S. EPA 2020 Ozone Policy Assessment presents a staff analysis of the scientific basis for policy options for considerations and judgments required of the senior EPA management and Administrator in determining whether it is appropriate to retain or revise the NAAQS.

*Figure 2-9. Influence of Canadian and U.S. emissions on annual sulfur deposition in transboundary region estimated from the U.S. EPA 2020 Ozone Policy Assessment simulations. The panels show (a) the total deposition, (b) total deposition from U.S. emissions sources only, (c) total deposition from Canadian sources only, and (d) the ratio of total deposition from U.S. over Canadian emissions sources. Modeled deposition is shown for the region within 500-km of the Canada-U.S. border.*

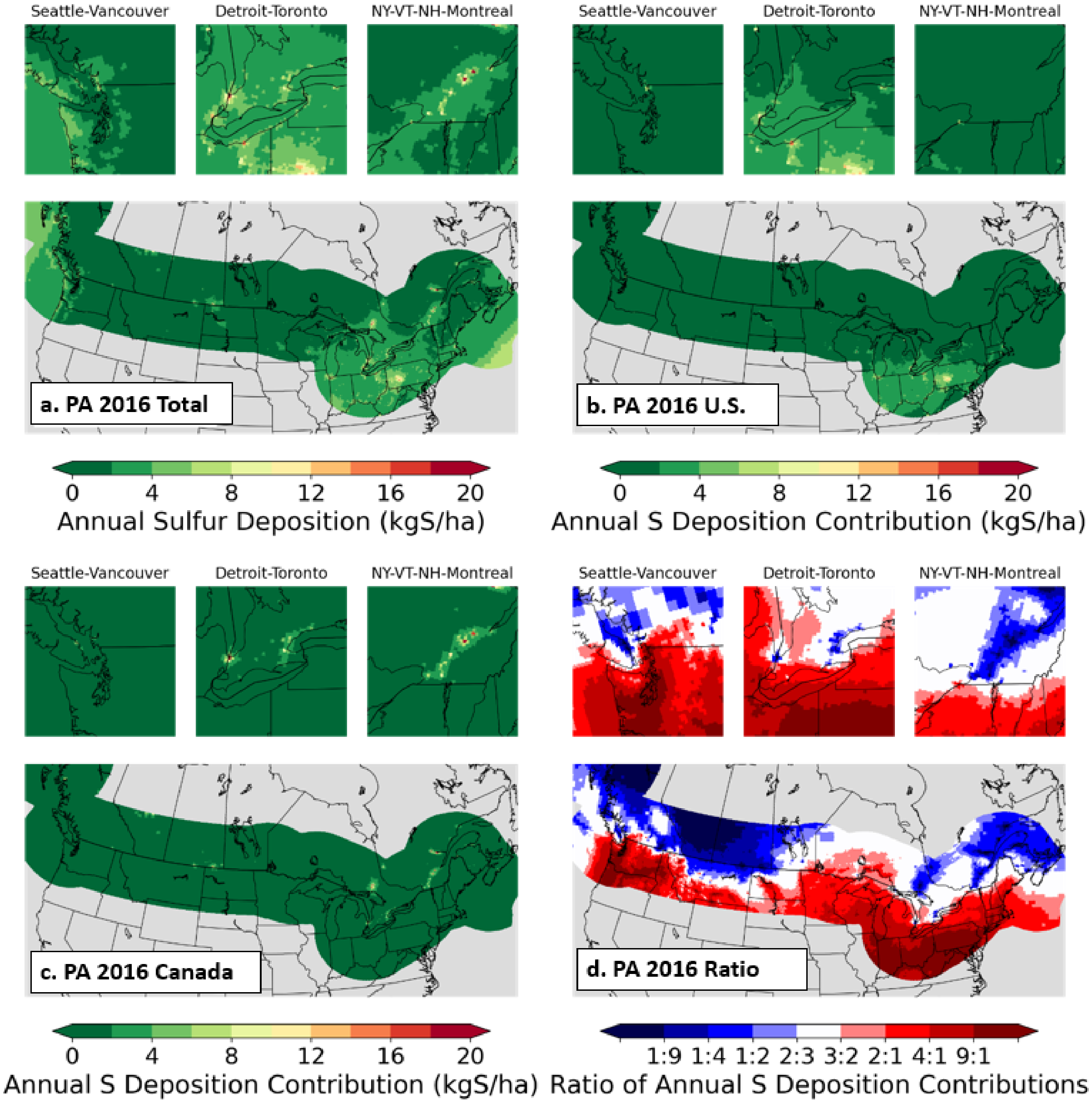
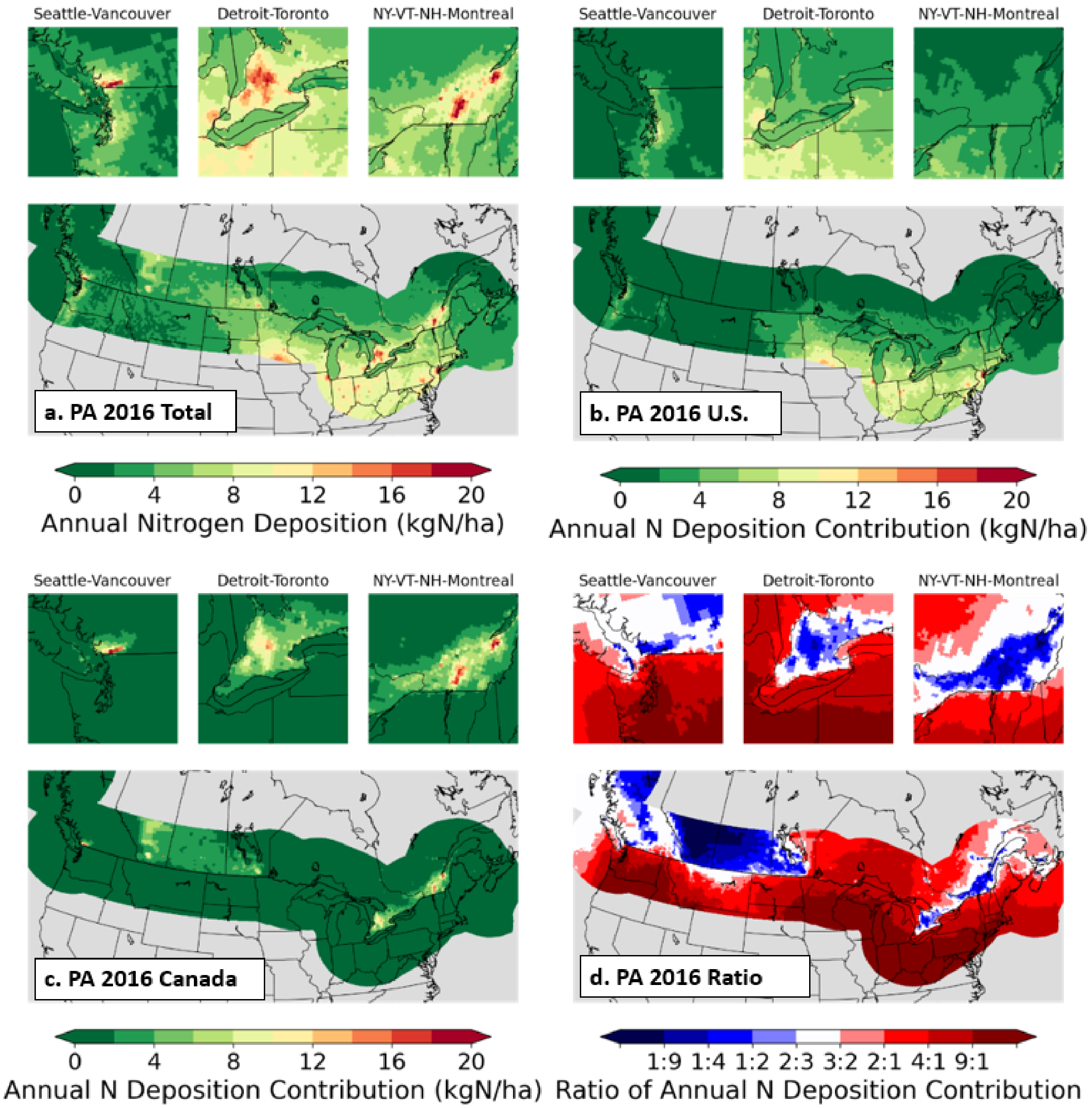


*Figure 2-10. Influence of Canadian and U.S. emissions on annual nitrogen deposition in transboundary region estimated from the U.S. EPA 2020 Ozone Policy Assessment simulations. The panels show (a) the total deposition, (b) total deposition from U.S. emissions sources only, (c) total deposition from Canadian sources only, and (d) the ratio of total deposition from U.S. over Canadian emissions sources. Modeled deposition is shown for the region within 500-km of the Canada-U.S. border.*



## 2.4 Aquatic Acidification and Eutrophication

Deposition of total nitrogen and sulfur to water bodies can cause or contribute to aquatic acidification and eutrophication which can affect the health of the aquatic ecosystem. The amount of a pollutant that is estimated to cause a biological effect or change in an ecological system is called a critical load. Critical loads for aquatic systems are generally developed using models that take into account historical pollution impacts and geological conditions, both of which affect leaching of total nitrogen and sulfur out of the aquatic system. Ecosystems that are less responsive to acidic pollution have high critical loads, while sensitive ecosystems have low critical loads. An ecosystem is said to be in exceedance of its critical load when acid deposition crosses the critical loads threshold. In both the U.S. and Canada, sustained emissions reductions and deposition rates outlined in Sections 2.1 and 2.2 have resulted in broad reductions in critical load exceedances. However, some persistent challenges remain in both countries where current sulfur and nitrogen loadings from 2019-2021 still exceed levels required for recovery of some lakes and streams. In the U.S., critical loads are used for analysis purpose only, whereas in Canada, critical loads are used to inform emissions policy development.

While deposition from air-related emissions sources is an important contributor of total nitrogen and total sulfur, it is generally not the dominant source of total nitrogen and total sulfur to water bodies that are located within or close to agricultural areas. For example, synthetic fertilizer is estimated to be the single largest source of human-caused total nitrogen inputs (US EPA, 2020b). Geologic sources in the soil of the watershed, generally in the form of $SO_4^{-2}$, release total sulfur into the water and can be important contributors, especially in areas that have experienced historically high total sulfur deposition (US EPA, 2020b).

### 2.4.1 Canada

Critical loads of acidity were established for lakes across Canada using the Steady-State Water Chemistry model (SSWC) (Henriksen & Posch, 2001) and setting a threshold based on the acid neutralizing capacity (ANC). The ANC threshold was usually set at a value related to the dissolved organic carbon (DOC) concentration in the lake water[10] or, in the absence of DOC data, at the commonly used value of 40 $\mu mol_c\ L^{-1}$ (Henriksen et al., 2002). The critical load for a region was set at a level to protect 95% of lakes in that area from the harmful effects of acid deposition. Critical loads of acidity for natural and semi-natural mineral soils (*i.e.*, excluding organic and agricultural soils) were recently mapped using the Simple Mass Balance model (Posch et al., 2015), which considers inputs from deposition and base cation weathering as well as outputs such as leaching and harvesting. A site-specific critical base cation to aluminum ratio representing a 5% root or biomass growth restriction was set to protect sensitive trees and vegetation.

The resulting aquatic and terrestrial critical loads of acidity can be used to estimate exceedances to identify areas of concern. For example, in a recent study (Cheng et al., 2022), terrestrial critical loads of acidity were estimated surrounding 14 stations and aquatic critical loads were estimated around five sites in Canada, which represent a range of soil and surface water conditions. In the early 2000s, acid deposition exceeded critical loads at approximately one third of the sites, but since 2012 has decreased below these thresholds.

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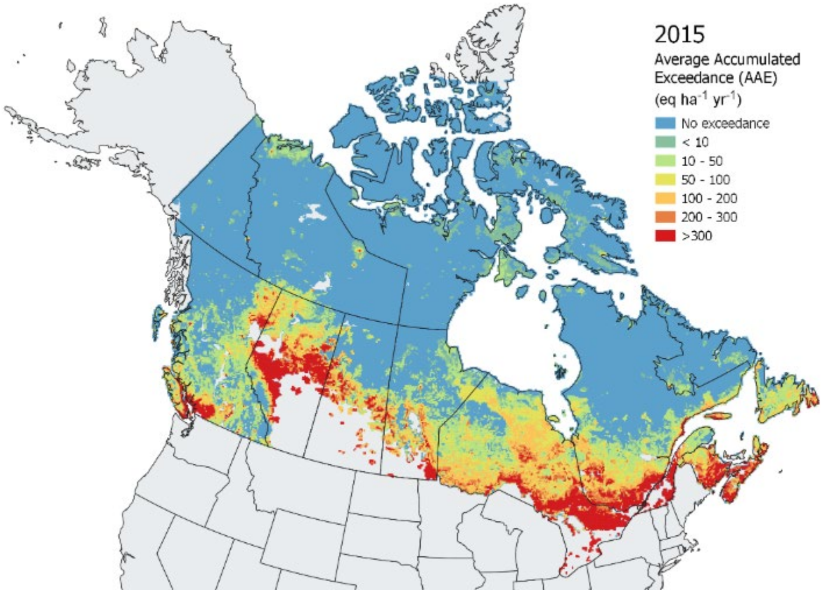
[10] The threshold is measured in micro-mol charge equivalent per liter ($\mu mol_c\ L^{-1}$)

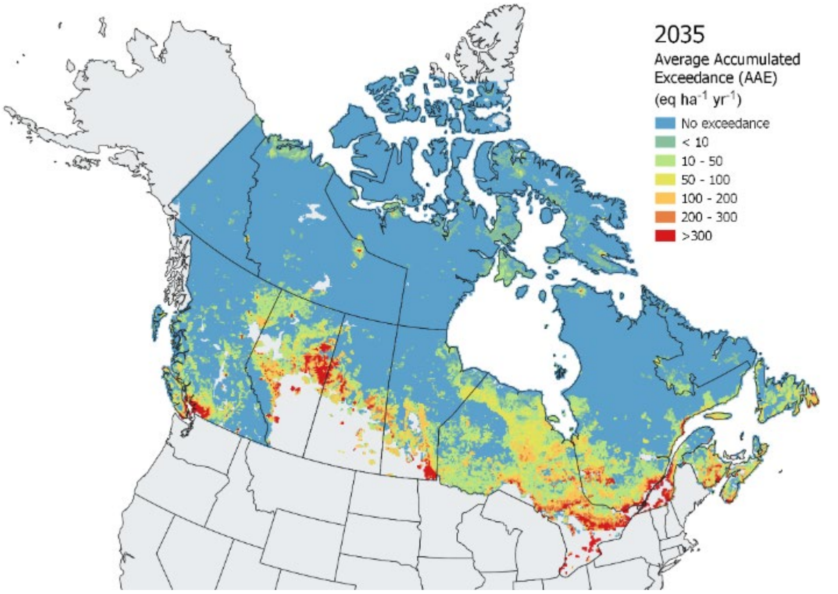
Model estimates of nitrogen and sulfur deposition provide improved spatial coverage and chemical coverage (*e.g.*, deposition of non-measured species) and the ability to look at the impact of future emission projections on acid deposition and resulting exceedances of critical loads. The model scenarios described in Appendix A present estimated deposition of total nitrogen and total sulfur. Figure 2-11 compares the terrestrial sulfur plus nitrogen critical load exceedances for the base year (2015) to the projected exceedances for the 2035 business as usual (BAU) scenario. Lake critical load exceedances (Figure 2-12) were estimated using sulfur deposition only, since the SSWC model does not take into account sinks of nitrogen in a catchment or lake and should be considered the "best case scenario" for surface waters.

As seen in Figure 2-11 and Figure 2-12, large areas of northern Alberta and Saskatchewan are particularly sensitive to acid deposition and are in exceedance of their critical loads both in the base year, 2015, and future scenario for 2035. Eastern Canada and southwestern British Columbia also show critical load exceedances for both soils and surface waters, though the mineral soil exceedances are significantly smaller in area under the projected future emissions. These latter regions are significantly impacted by contributions from transboundary transport of total nitrogen and total sulfur deposition, as shown in Figure 2-9 and Figure 2-10. Under the 2035 BAU scenario, exceedance of soil critical loads was broadly predicted to decline in magnitude and area except in a few regions near point sources; a smaller reduction in S exceedance in central and eastern Canada is seen in the lake maps.

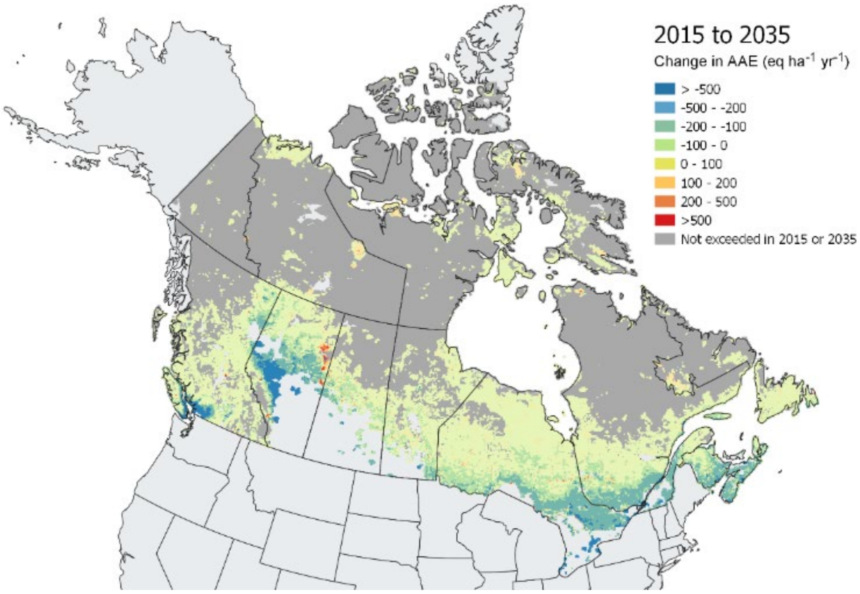
*Figure 2-11. Maps of Average Accumulated Exceedance (AAE) of natural and semi-mineral soils under the Global Environmental Multiscale Model – Modeling Air Quality and Chemistry (GEM-MACH) sulfur plus nitrogen deposition models for (a) baseline 2015 and (b) BAU 2035, along with (c) reductions (negative values) or increases (positive values) in AAE between 2015 and 2035. Critical loads of acidity were estimated using the Simple Mass Balance model (UNECE, 2004).*



a. 2015



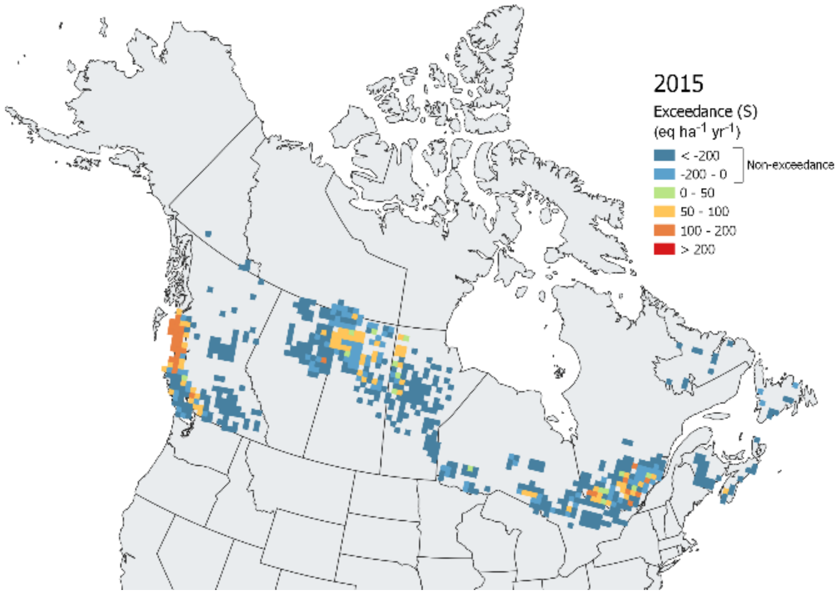
b. 2035



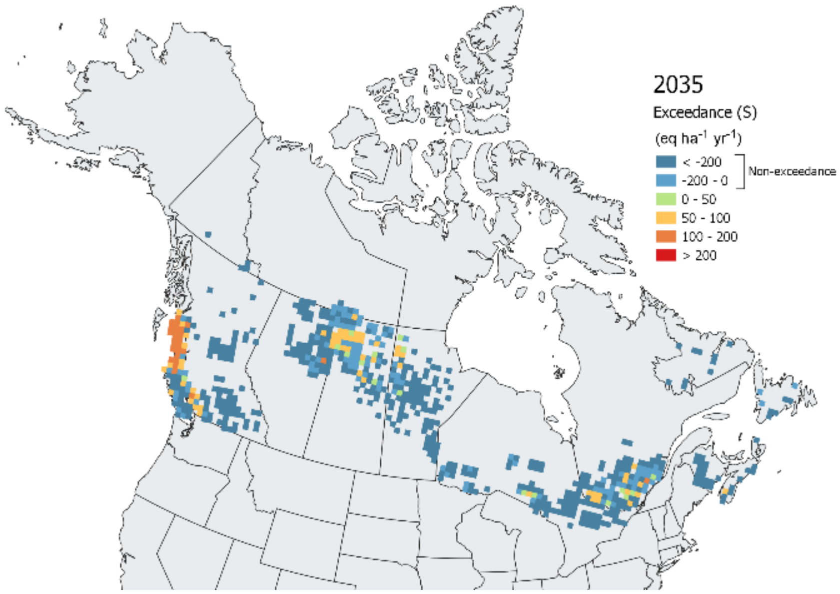
c. 2035 – 2015 difference map

*Data sources: Soil data sourced from the OpenLandMap project (openlandmap.org) as well as the Canadian Soil Information System (sis.agr.gc.ca/cansis). Runoff estimates from Reinds et al., 2015. Forest data sourced from the National Forest Inventory (nfi.nfis.org) and supporting information from Pardo et al., 2005; Paré et al., 2013. Landcover data from North American Land Cover Change Monitoring System (2010).*

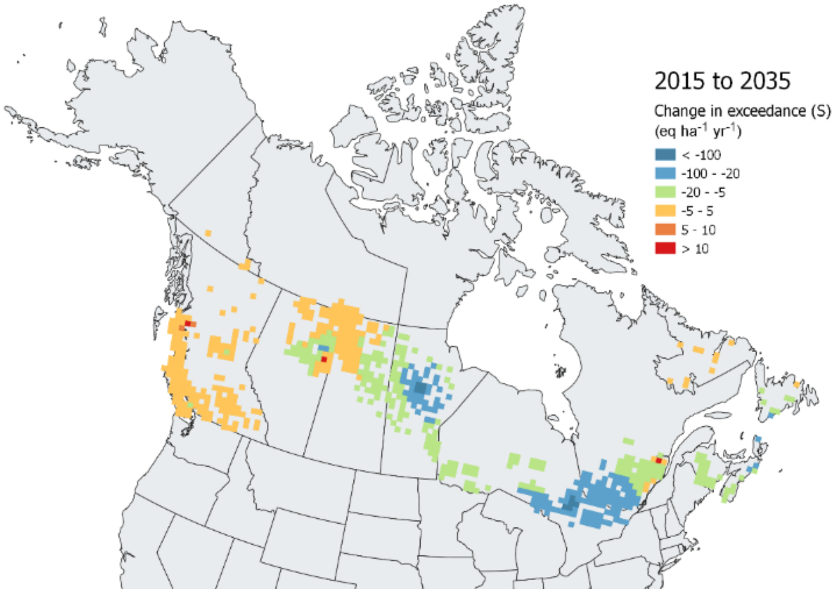
*Figure 2-12. Mapped lake exceedance under GEM-MACH modeled deposition for (a) baseline 2015 and (b) BAU 2035, with changes to the exceedance shown in (c). Lake critical loads of acidity were estimated with the SSWC model using sulfur deposition only. Values above 0 are considered in exceedance of their critical loads. Critical loads were summed to the 5<sup>th</sup> percentile, to provide protection for 95% of the ecosystems in each 42 km grid (Jeffries et al., 2010).*



a. 2015



b. 2035



c. 2035 – 2015 difference map

### 2.4.2 United States

Figure 2-13 shows that U.S. emissions reductions achieved between 2000 and 2021 have contributed to broad surface water improvements and increased aquatic ecosystem protection across the five Long-Term Monitoring (LTM) regions along the Appalachian Mountains. These emissions reductions are expected to continue to contribute to improvements in coming years.

In support of the analysis shown in Figure 2-13, the U.S. collected surface water samples from 7,869 lakes and streams along acid-sensitive regions of the Appalachian Mountains and some adjoining Northern coastal plain regions through a number of water quality monitoring programs. Critical loads information were obtained from the National Critical Load Database (NCLDv3.2.1), a repository of critical load data for the U.S. (CLAD, 2022; Lynch et al., 2022). Aquatic critical loads were determined using a host of methods from the SSWC model (Henriksen & Posch, 2001) to a regional regression model (McDonnell et al., 2012; Sullivan et al., 2012). Critical load exceedances were determined using total deposition estimate of total nitrogen and sulfur deposition for the period of 2000-2002 and 2019-2021 (US EPA, 2021a).

The analysis shown in Figure 2-13 focuses on aquatic biological resources in acid-sensitive regions in the eastern U.S. Lake and stream waters having an ANC[11] – a key indicator of aquatic ecosystem recovery from acidification – value greater than 50 µeq/L. An ANC of 50 is often used as a target in Critical Loads

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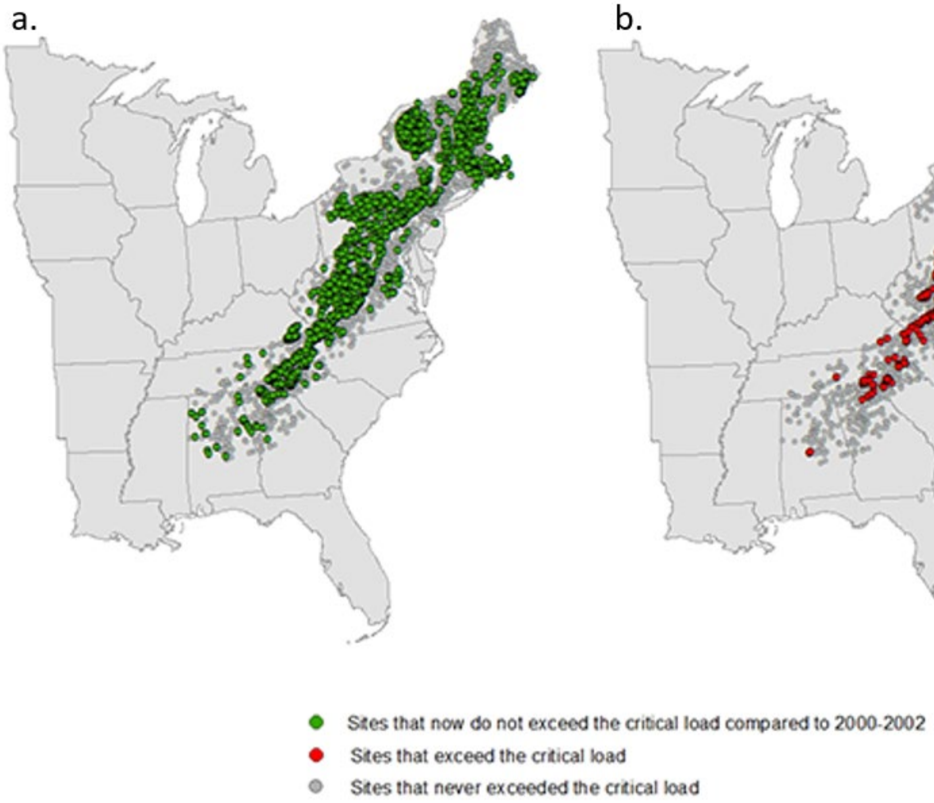
[11] Measured in micro-equivalents per liter (µeq/L)

*Figure 2-13. Lake and stream exceedances of estimated critical loads for total nitrogen and sulfur deposition, for (a) sites that exceeded the critical load in 2000-2002 but did not exceed the critical load in 2019–2021 and (b) sites that still exceeded the critical load in 2019-2021.*



Data Sources: National Surface Water Survey (NSWS), Environmental Monitoring and Assessment Program (EMAP), Wadeable Stream Assessment (WSA), National Lake Assessment (NLA), Temporally Integrated Monitoring of Ecosystems (TIME), LTM, and other water quality monitoring programs.

for freshwater waterbodies[12]. The aquatic critical load represents the amount of combined sulfur and nitrogen that could be deposited annually to a lake or stream and its watershed and still support a moderately healthy aquatic ecosystem. Critical loads exceedances showed a decreasing trend from 2000-2002 to 2019-2021. The percentage of lakes and streams exceeding critical loads fell from 38% to 5.8%.

## 2.5 Recovery

Recent studies of long-term trends in water and soil chemistry provide indicators of ecosystem recovery in both countries. Signs of acid deposition impacts on soils include the release of free aluminum, which is toxic to plants and animals, and increased potential hydrogen (pH). Forest soils in eastern Canada and the northeastern U.S. showed some evidence of decreased free aluminum and increased pH at most sites sampled (Lawrence et al., 2015), suggesting that forest soils are undergoing recovery from the effects of acidic deposition. In response to acid deposition reductions, $SO_4^{2-}$ concentrations significantly

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[12] Studies of acid-impacted waterbodies have reported reduced numbers of fish species with ANC below 0 to 20 µeq/L. The U.S. EPA's National Lakes Assessment has used an ANC threshold of approximately 50 µeq/L, above which there is less concern for acidification; a value of 20 µeq/L has been characterized as a lower bound for protection against chronic acidification-related adverse impacts on fish populations (US EPA, 2009, 2011, 2020b).

declined, and pH generally increased, in surface waters in Atlantic Canada, Ontario, Quebec, the northeastern U.S., New York, and Virginia (*e.g.*, Baldigo et al., 2021; Houle et al., 2022; Marty et al., 2021; Nelson et al., 2021; Redden et al., 2021; Scanlon et al., 2021; Watmough & Eimers, 2020; Webster et al., 2021). However, the recovery is incomplete. For example, very acid-sensitive lakes in Nova Scotia did not show evidence of recovery up to 2007 (Clair et al., 2011), have only recently started to recover (Redden et al., 2021), and pH and calcium values are still below recommended thresholds to protect aquatic life (Houle et al., 2022). In other locations, ANC, an indicator of recovery, has continued to decrease (*e.g.*, Scanlon et al., 2021), or toxic aluminum continued to increase (Redden et al., 2021). The recovery of lakes is impacted by cumulative effects of acidic deposition on catchment soils (*e.g.*, Eng & Scanlon, 2021; Hazlett et al., 2020; Scanlon et al., 2021), and may need decades to reach target pH and ANC (Whitfield et al., 2007). Sustained efforts to maintain or further reduce levels of acid deposition throughout the U.S. and Canada would allow these sensitive areas time to recover.

## 2.6 Summary

The Acid Rain Annex (Annex 1) to the AQA sets out objectives for Canada and the U.S. to reduce emissions of $SO_2$ and $NO_X$ that cause acid rain. Both countries have met their commitments to reduce $SO_2$ and $NO_X$ emissions under the Agreement since 2007.

In 2020, Canada's total $SO_2$ emissions were approximately 651,000 metric tons, a 78% reduction from Canada's total $SO_2$ emissions of 3.0 million metric tons in 1990. Between 1990 and 2020, Canada's total $NO_X$ emissions also decreased by 36% (826 thousand metric tons). These emissions reductions have been achieved through programs including the 1985 Eastern Canada Acid Rain Control Program and Canada-wide Acid Rain Strategy for Post-2000.

In the U.S., between 1990 and 2020, $SO_2$ emissions have decreased by 93% from 23.1 million metric tons to 1.9 million metric tons, and $NO_X$ emissions have decreased by 70%, from 25.5 million metric tons to 7.8 million metric tons. The ARP in the U.S. has dramatically cut power plant emissions of $SO_2$ and $NO_X$, reducing acid rain. Regulatory actions in the U.S. such as CSAPR and its subsequent updates have also achieved large reductions in annual $SO_2$ and annual and summertime $NO_X$ emissions from the power sector. Further reductions, not yet reflected in cited modeling, are expected following recent regulatory actions, such as the Good Neighbor Plan (US EPA, 2023b), and upcoming implementation of the provisions contained in the Inflation Reduction Act.

$SO_4^{2-}$ and $NO_3^-$ wet deposition have decreased by approximately 70% and 50%, respectively, in response to $SO_2$ and $NO_X$ emission reductions in both Canada and the U.S. Long-term water quality surveys show conditions improving from historical acidification for watersheds in some regions, while delayed recovery has been observed in others.

While important progress has been made, further reductions in total nitrogen and sulfur inputs to U.S. and Canadian water bodies, including those from air-related sources, could further reduce aquatic acidification and speed up the recovery of aquatic and terrestrial ecosystems. Zero-out modeling suggests transboundary influence on total deposition, particularly in the less populated parts of northern Montana and the northern parts of the province of Ontario, where deposition is lower than in the northeastern U.S. Many water bodies in Canada as well as some in the U.S. are still exposed to total nitrogen and sulfur that exceed the capacity of soils and surface waters to neutralize the acidic contributions. The continued exceedances and modeling projections indicate that aquatic acidification remains an environmental concern. Additionally, emissions of ammonia have not experienced the sharp decline that has been seen with $SO_2$ and $NO_X$ emissions. In fact, in some areas (*i.e.*, agricultural regions) reduced nitrogen emissions have increased. The role of nitrogen in contributing to acidification and eutrophication is an issue of great interest to scientists (Walker, Beachley, et al., 2019) and policy makers (Kanter et al., 2020).

# 3 Ground-level Ozone

*What is ground-level ozone:* Ground-level ozone is a colorless and highly irritating air pollutant and greenhouse gas that forms just above the earth's surface. As outlined in Section 1.2, ozone is harmful to human health and ecosystems and reduces crop yields. Ozone is not emitted directly into the air but rather is formed primarily by chemical reactions between $NO_X$ and VOCs in the presence of sunlight. $NO_X$ and VOCs are emitted by combustion sources such as vehicles and power plants. VOCs are also given off by solvents, cleaners, and paints, as well as natural (biogenic) sources.

*Ground-level ozone in the AQA:* Annex 3, the Ozone Annex, contains commitments by Canada and the U.S. to control and reduce emissions of $NO_X$ and VOCs, key precursors to ground-level ozone. Both Canada and the U.S. have met their commitments, as described in Section 3.2. These commitments apply to a defined region in both countries known as the Pollutant Emission Management Area (PEMA), which at the time of the signing of the Annex, was the area deemed the most critical for reducing transboundary ozone. The PEMA includes central and southern Ontario, southern Quebec, 18 U.S. states, and the District of Columbia (Figure 3-1). The objective of the Ozone Annex is to help both countries achieve their respective air quality standards for ozone, that is, the then-current Canada-wide Standard for ozone (later replaced by the Canadian Ambient Air Quality Standard (CAAQS) for Ozone in 2013) and the U.S. ozone NAAQS. See Table 3-1. for a summary of the commitments under the Ozone Annex.

*Figure 3-1. Map of ozone Annex PEMA*

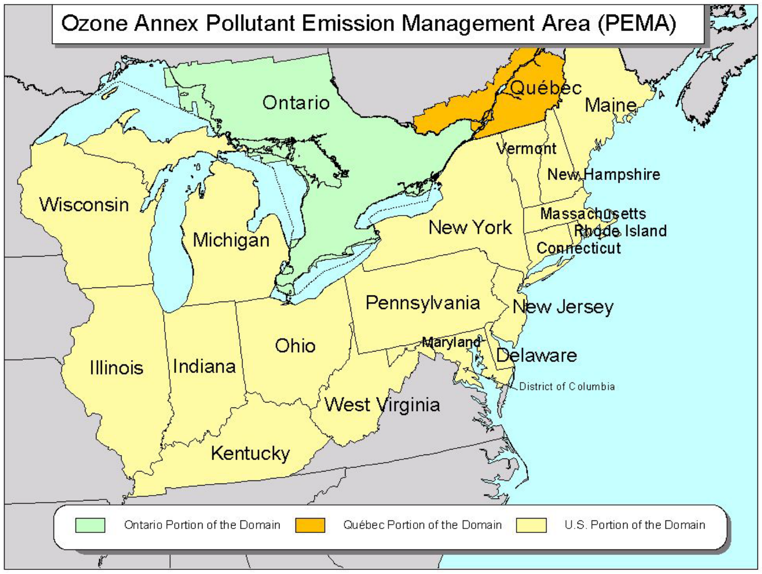

*Table 3-1. Ozone Annex (Annex 3): specific objectives concerning ground-level ozone precursors*[13]

| Objective | Canada | U.S. |
|---|---|---|
| **Mobile Sources of $NO_x$ and VOCs** | • Implementation of stringent $NO_x$ and VOC emissions reduction standards for vehicles, including cars, vans, light-duty trucks, off-road vehicles, small engines, and diesel engines, as well as fuels. | • Implementation of existing U.S. vehicle, nonroad engine, and fuel quality rules to achieve both VOC and $NO_x$ reductions. |
| **Stationary Sources of $NO_x$** | • By 2007, annual caps of 39 thousand metric tons of $NO_x$ emissions from fossil-fuel power plants in the Ontario portion of the PEMA in and 5 thousand metric tons of $NO_x$ in the Quebec portion of the PEMA<br>• Development of a proposed national Guideline for Renewable Low-Impact Electricity. | • Implementation of the $NO_x$ emissions reductions program, known as the $NO_x$ SIP Call, in the PEMA states that are subject to the rule. |
| **$NO_x$ and VOCs Emission Reduction Strategies** | • Implementation of measures to reduce $NO_x$ emissions from key industrial sectors, and VOC emissions from solvents, paints, and consumer products to attain the Canada-wide Standard for Ozone.<br>• Implementation of Ontario and Quebec specific measures to reduce emissions of $NO_x$ and VOCs in the PEMA region | • Implementation of existing U.S. rules for control of emissions from stationary sources of hazardous air pollutants and control of VOCs from consumer and commercial products, architectural coatings, and automobile repair coatings.<br>• Implementation of 36 existing U.S. new source performance standards, to achieve VOC and $NO_x$ reductions from new sources. |
| **Anticipated Additional Control Measures and Indicative Reductions** | • Implementation of anticipated additional control measures that are expected to contribute to overall reductions of $NO_x$ and VOC emissions. | |
| **Reporting** | • Beginning in 2004, Parties agreed to report annual and ozone season (typically January until May across the U.S.) emissions of $NO_x$ and VOCs for the PEMA;<br>• Beginning in 2002, Parties agree to report on ambient ozone, $NO_x$ and VOC concentrations, and 10-year trends within 500 km of the Canada-U.S. border. | |
| **Revisiting** | • In 2004, assess progress in implementing the annex with a view to negotiating further reductions;<br>• At the request of either Party, discuss the possibility of amending the annex to designate additional PEMAs or to revise annex commitments | |

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[13] The complete text of the Ozone Annex (Annex 3) can be found at https://www.canada.ca/en/environment-climate-change/services/air-pollution/publications/canada-united-states-quality-agreement-ozone-annex.html.

|  |  |
|---|---|
| **More Stringent Measures** | - Either Party may take more stringent measures to control and reduce $NO_x$ and VOC emissions than those specified in this Annex. |

## 3.1 Air Quality Standards and Guidelines

Both Canada and the U.S. have established air quality standards to protect human health and the environment.

For Canada, CAAQS were established as objectives under the *Canadian Environmental Protection Act*, 1999, and are a key component of Canada's AQMS. The CAAQS for ozone were selected on the basis of providing a specific level of improvement to population exposure (and the related improvement to population health) when the targets are met. Future more stringent targets recognize the non-threshold nature of the health effects of ozone (i.e., adverse effects occur even at low concentrations) and the AQMS principle of continuous improvement. Further, while the CAAQS are primarily health-based, they also explicitly recognize that at these levels, environmental effects may also be exerted by ozone. The CAAQS are underpinned by four air quality management levels, where each level requires progressively more rigorous management action by provincial and territorial jurisdictions as the air quality in a given air zone or area approaches or exceeds the CAAQS (CCME, 2021). Certain Canadian jurisdictions have also established air quality criteria/objectives of their own. Under AQMS, provincial and territorial governments are required to provide annual reports on air quality for each of their air zones. These reports include the actual metric values and achievement status of the CAAQS for each CAAQS reporting station and air zone, as well as associated management actions. The CAAQS are not legally binding or enforceable, meaning that there are no mechanisms (financial or otherwise) imparted by the federal government that require provinces to achieve the CAAQS.

The U.S. NAAQS (US EPA, 2023e) are set by the U.S. EPA in accordance with the U.S. Clean Air Act, which requires the U.S. EPA to establish standards for those pollutants for which air quality criteria have been issued under section 108 of the U.S. Clean Air Act, which are also referred to as criteria pollutants. The NAAQS are generally implemented through the development of state implementation plans, through which states are required to attain and maintain the level of the NAAQS, and which must be submitted to the U.S. EPA for review and approval. Section 109 of the U.S. Clean Air Act directs the EPA Administrator to set two types of NAAQS, "primary" and "secondary" standards, for criteria pollutants. Under section 109(b)(1) of the Act, the primary NAAQS are health-based, and are defined as those standards "the attainment and maintenance of which in the judgment of the Administrator, based on such [air quality] criteria and allowing an adequate margin of safety, are requisite to protect the public health." Under section 109(b)(2), secondary standards are welfare based, and must "specify a level of air quality the attainment and maintenance of which, in the judgment of the Administrator, based on such criteria, is requisite to protect the public welfare from any known or anticipated adverse effects associated with the presence of [the] pollutant in the ambient air." Under section 302(h) of the Act, effects on welfare include, but are not limited to "effects on soils, water, crops, vegetation, manmade materials, animals, wildlife, weather, visibility, and climate, damage to and deterioration of property, and hazards to transportation, as well as effects on economic values and on personal comfort and well-being." (US EPA, 2023e).

Table 3-2 summarizes the current CAAQS for ozone and the U.S. ozone NAAQS. Each standard is defined in terms of the chemical species or mixture to be measured, an averaging time period, a "numerical value" (level), and a "metric" (the statistical form of the numerical standard). For ozone, the CAAQS and U.S. NAAQS are the annual 4th-highest daily maximum 8-hour concentration (MDA8), averaged over three years. The CAAQS numerical values for ozone are becoming more stringent over time, with different standards for 2015, 2020, and 2025.

*Table 3-2. National ambient air quality standards for ozone*

| **Averaging Time** | **CAAQS Numerical Value** | | | **U.S. NAAQS Level** | **CAAQS and NAAQS Metric**[14,15, 16] |
|---|---|---|---|---|---|
| | **2015** | **2020** | **2025** | | |
| 8- hour | 63 ppb | 62 ppb | 60 ppb | 0.070 ppm (70 ppb) (primary and secondary) | Annual fourth-highest daily maximum 8-hour concentration, averaged over 3 years |

## 3.2 Effect of Emissions Reduction Strategies on Ground-Level Ozone Precursors

### 3.2.1 Canada

Under the Ozone Annex, Canada committed to establishing more stringent $NO_X$ and VOC emissions standards for vehicles and engines, limiting the sulfur content in fuels, establishing annual $NO_X$ emissions caps for fossil fuel power plants in southern Ontario and southern Quebec, and establishing regulations to reduce emissions of VOCs. Canada has met its Ozone Annex commitments. Canada has implemented and continues to implement a series of regulations to align Canadian emission standards for vehicles, engines, and fuels with corresponding standards in the U.S. These include regulations that set emission performance standards for on-road (On-Road Vehicle and Engine Emission Regulations, SOR/2003-2) and off-road vehicles (Off-Road Small Spark-Ignition Engine Emission Regulations, SOR/2003-355) and off-road mobile and stationary engines (see the most recent Progress Report for further details (ECCC & US EPA, 2023)), as well as regulations to limit sulfur and benzene in gasoline and sulfur in diesel. Both Ontario and Quebec have met their $NO_X$ limits for the electricity sector.

Canada has also put in place various regulations to address VOC emissions, such as regulations to reduce VOCs from dry cleaning, solvent degreasing, automotive refinishing products, and architectural coatings. Recent regulations establish maximum VOC concentrations and emissions for the manufacture and

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[14] The "form" of a standard defines the air quality statistic that is to be taken for the appropriate averaging time and compared to the level of the standard in determining whether an area attains the standard.

[15] Under the CAAQS, transboundary flows and exceptional events are influences on concentrations from sources over which the jurisdiction has little or no direct control. This includes, for example, wildfire influences. Provinces and territories have the option of demonstrating in air zone reports that a given CAAQS exceedance may have been influenced by transboundary flows (TF) or exceptional events (EE), such as wildfires. The metric values that could have been influenced by TF or EE can be re-calculated using the guidance provided in the Guidance Document on Transboundary Flows and Exceptional Events for Air Zone Management.

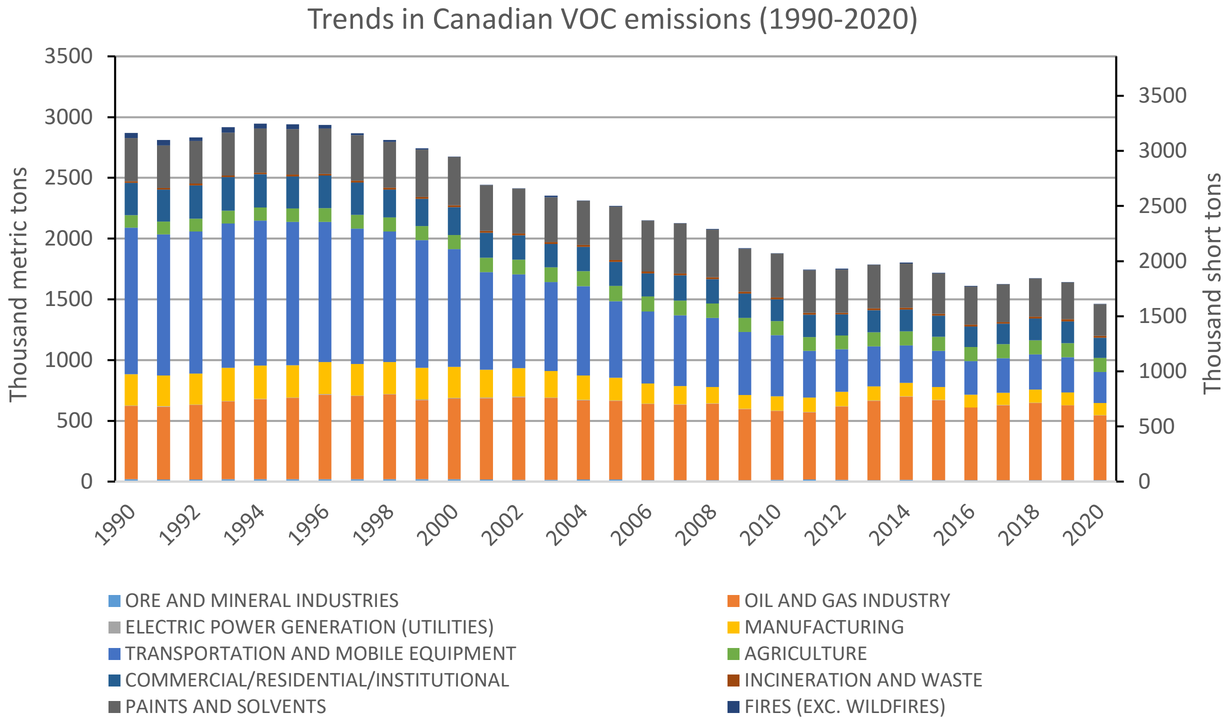
[16] Exceptional events are unusual or naturally occurring events that can affect air quality but are not reasonably controllable or preventable using techniques that tribal, state, or local air agencies may implement in order to attain and maintain the NAAQS. Exceptional events may include wildfires, high wind dust events, prescribed fires, stratospheric ozone intrusions, and volcanic and seismic activities. Local and Regional offices are able to flag their exceptional event data through the Air Quality System, and ultimately request that U.S. EPA exclude that data from certain regulatory decisions for the U.S. EPA to exclude all hours in an event-affected day. More information in U.S. exceptional events are in CAA 319(b), 42 USC 7619(b), and 2016 rule – 81 FR 68216 (Oct. 3, 2016).

import of over 130 categories and sub-categories of products. Canada has published a notice of intent to renew the federal agenda on the reduction of emissions of VOCs from consumer and commercial products for the period of 2022 to 2030. Canada has also established requirements to limit VOC emissions from industrial facilities, as well as regulations to reduce emissions of methane and certain VOCs from the upstream oil and gas sector.

The addition of the Ozone Annex in 2000 resulted in further decreases in Canada's $NO_X$ emissions, as shown in Figure 2-2. Strategies for reduction of $NO_X$ emissions were included under the Acid Rain Annex, but $NO_X$ emissions in Canada continued to increase by 485 thousand metric tons (21%) from 1990 to 1999. From 2000 to 2020, after the Ozone Annex was included in the AQA, $NO_X$ emissions in Canada decreased by 1.3 million metric tons, with the largest decreases in the transportation (900 thousand metric tons), electric power generation (226 thousand metric tons), and manufacturing (111 thousand metric tons) sectors.

Between 1990 and 2020, national VOC emissions in Canada decreased by 49% (1.4 million metric tons) (Figure 3-2). For example, increasingly stringent regulations on spark-ignition engines lead to a nation-wide decrease of 657 thousand metric tons in VOC emissions from off-road gasoline, liquefied petroleum gas, and natural gas vehicles and equipment.

*Figure 3-2. Canadian VOC emission trends for 1990-2020*



Data source: Canada's Air Pollutant Emissions Inventory 1990-2020 *(ECCC, 2022)*

### 3.2.2 United States

Under the Ozone Annex, the U.S. committed to implement existing U.S. vehicle, non-road engine, and fuel quality rules to achieve both VOC and $NO_X$ reductions. In addition, the U.S. committed to implementing existing rules for control of emissions from stationary sources of hazardous air pollutants and control of VOCs from consumer and commercial products, architectural coatings, and automobile repair coatings, and to continue existing U.S. new source performance standards (NSPS) to achieve VOC and $NO_X$ reductions from new sources. These actions have led to the U.S. meeting its commitments under the Ozone Annex

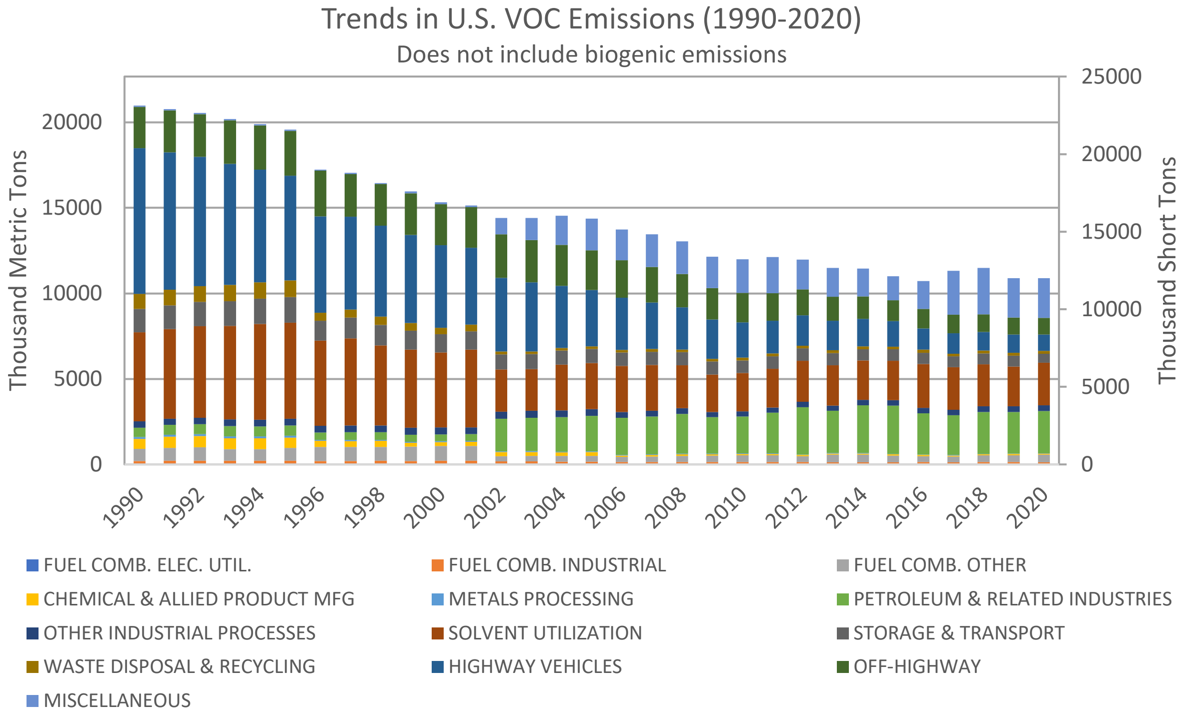
Emissions of ozone precursors have been reduced due to several programs. Reductions in power plant emissions have been achieved through multi-state $NO_X$ budget programs designed to address interstate transport of air pollution contributing to ozone, first by twelve New England and Mid-Atlantic states and the District of Columbia, and later for much of the eastern U.S. (the $NO_X$ SIP Call Regulations). The subsequent CAIR achieved large reductions in power plant annual $SO_2$ and $NO_X$ emissions beyond those required by the $NO_X$ SIP Call. In response to litigation, CAIR was replaced by the CSAPR. CSAPR was later updated and revised to further reduce summertime $NO_X$ emissions from power plants in the eastern U.S. and help downwind states to meet ozone standards. In March 2023, the U.S. adopted a new set of $NO_X$ control requirements for power plants and industrial sources under the "Good Neighbor Rule" (US EPA, 2023b).

The U.S. has required significant reductions over the years in $NO_X$ and VOC emissions from new motor vehicles and non-road engines – such as those used in construction, agriculture, industry, trains, and marine vessels – through standards that require a combination of cleaner engine technologies and cleaner fuels, such as those listed in Section 2.1.2.

U.S. EPA continues to implement and update NSPS to achieve VOC and $NO_X$ reductions from new and modified sources. Reductions of $NO_X$ emissions are also being achieved through rules on solid waste incineration units and guidelines that impact new and existing incineration units. Programs and rules such as those mentioned above have contributed significantly to $NO_X$ and VOC reductions, enhancing public health and environmental protections regionally and for local communities.

As shown in Figure 2-4, there has been an overall trend of emissions reduction for $NO_X$. A similar trend can be seen for VOCs, as shown in Figure 3-3. The decrease in emissions from 1990 to 2020 for $NO_X$ is 70% and for VOCs is 48%. The greatest $NO_X$ reductions are found in electric utility fuel combustion. Transportation recorded a greater than 80% reduction in VOC emissions from 1990 to 2020. While there is an overall decrease in VOC emissions, petroleum and related industries noted an increase in emissions from 611 thousand short tons in 1990 to a peak of 3.1 million short tons in 2015, a pattern similar to that of $NO_X$. In 2020, VOC emissions were 13% less than at the 2015 peak. Emissions estimation methods and reporting for sources in oil and gas production have also improved significantly in recent years.

*Figure 3-3. U.S. VOC emission trends for 1990-2020.*



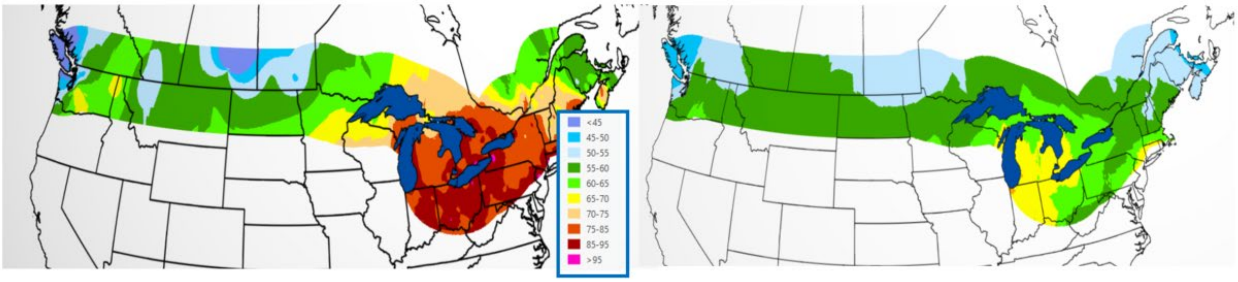
Data source: 2020 U.S. National Emissions Inventory (US EPA, 2023a).

## 3.3 Ambient Concentrations of Ozone

Ambient ozone concentrations were analyzed to assess the changes in ozone over the course of the AQA and to evaluate current ambient concentrations. Figure 3-4 shows interpolated (by kriging) maps of average annual 4th highest MDA8 ozone concentrations within 500 km of the Canada-U.S. border for 2000-2002 and 2018-2020. Figure 3-4a shows ozone concentrations in the border region for 2000-2002, around the time of signing of the Ozone Annex. The highest ozone concentrations were generally located in the Great Lakes-St. Lawrence region, the upper midwestern and northeastern U.S., southern Ontario, southern Quebec, and the southern Maritimes. Parts of southern Ontario and southern Quebec had annual 4th highest MDA8 ozone concentration over 80 ppb and 70 ppb, respectively, exceeding the 2000 Canada-wide standard for ozone of 65 ppb. Various areas of nonattainment in U.S. Great Lakes States also exceeded the 1997 NAAQS of 80 ppb, reaching 3-year averages of annual 4th highest MDA8 ozone concentration of 90 ppb in some locations. The lowest ozone concentrations were observed in the western side of the U.S.-Canadian border region.

Ozone concentrations have decreased significantly throughout the border region since the Ozone Annex was signed. Figure 3-4b, illustrating ozone concentrations in the border region for 2018-2020, shows that ozone concentrations are much decreased compared to 2000-2002. However, higher ozone concentrations still occur near the Great Lakes and along the U.S. eastern coast. The lowest values are generally found in western and eastern Canada. Concentrations are generally higher in, and downwind of, urban areas. Although ozone concentrations have decreased in southern Ontario, southern Quebec, and Great Lake States, areas continue to have ozone concentrations exceeding national standards in both the U.S. and Canada in these regions at the time of this report.

*Figure 3-4. Ozone concentrations (three-year average of the annual 4th highest MDA8 ozone concentration) along the Canada-U.S. border for (a) 2000-2002 and (b) 2018-2020. Concentrations are shown for the region within 500-km of the Canada-U.S. border.*



a. 2000-2002

b. 2018-2020

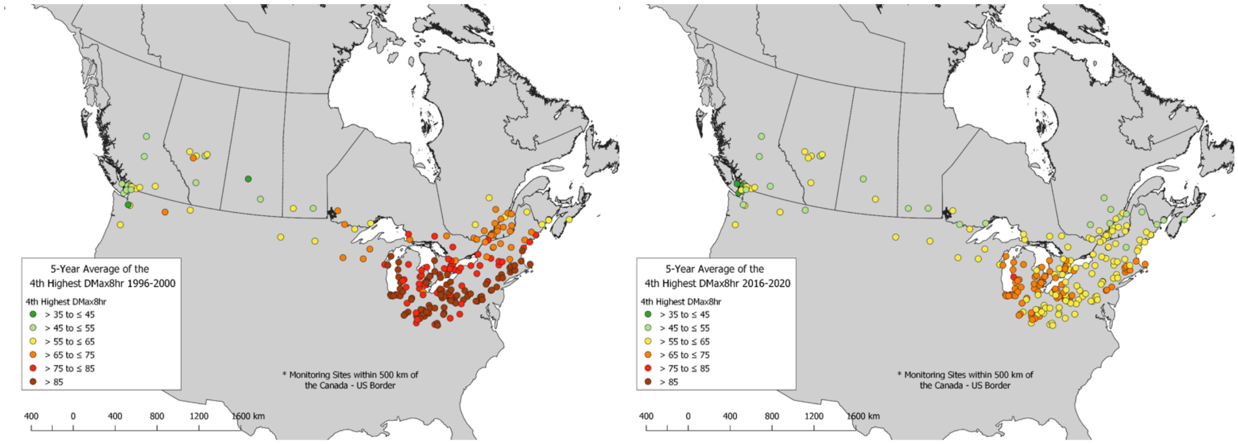
Data Source: Data are the averages of annual 4th-highest daily values, where the daily value is the highest running 8-hour average for the day, for (a) 2000-2002 and (b) 2018-2020. Point data interpolated by kriging. Only sites that met data completeness requirements (based upon 75 % or more of all possible daily values during the U.S. EPA-designated ozone monitoring seasons) were used to develop this map.

Ozone monitoring data for individual stations located within 500 km of the Canada-U.S. border for the period just prior to when the Ozone Annex was signed, 1996-2000 (Figure 3-5a), and for the most recent data years, 2016-2020 (Figure 3-5b), are shown in Figure 3-5. Ozone concentrations are shown as a 5-year average of the annual 4th-highest MDA8 ozone concentration. This statistical form is similar to metrics for both the CAAQS and U.S. NAAQS, but it is extended over a longer averaging period to account for variability introduced by wildfires and other factors. The concentrations were not adjusted for transboundary flow or exceptional events, including wildfires. Wildfires have a significant influence on ozone at some stations, as described below.

Nearly all the Canadian and U.S. stations east of the Manitoba-Ontario border have recorded lower ozone concentrations since 2000, by more than 10 ppb for many stations across Ontario, Quebec, and the Maritimes, and by as much as 20 ppb at some stations in the Great Lakes states and Ohio Valley. While some ozone concentrations measured over the past two decades have been associated with cool, rainy summers, the decreases from high MDA8 ozone concentrations are mainly due to air quality regulatory programs undertaken by both countries. A slight rise in ozone concentrations from 1996-2000 to 2016-2020 is observed at some stations in Alberta, Saskatchewan and Washington State. This may be related to impacts from a number of larger wildland fires (*Canadian Interagency Forest Fire Centre*, 2023; *National Interagency Fire Center*, 2023).

Overall, net ozone concentration decreases highlight the success of air quality regulations in the transport-relevant analysis region. However, despite decreasing trends, 2016-2020 ozone concentrations continue to be elevated in southeastern and southern Ontario, as well as some stations near the Mid-Atlantic and New England coasts. Some locations downwind of the Greater Toronto Area recorded higher ozone concentrations, likely a result of increased $NO_X$ emissions from sources in this large metropolitan area. In Canada, many of the stations within the PEMA continue to measure ozone concentrations that approach or exceed the CAAQS.

*Figure 3-5. Ozone concentrations (five-year average of the annual 4th highest MDA8 ozone concentration) along the Canada-U.S. border for (a) 1996-2000 and (b) 2016-2020.*



a. 1996-2000

b. 2016-2020

Data Source: Data are the averages of annual 4th-highest daily values, where the daily value is the highest running 8-hour average for the day, for (a) 1996-2000 and (b) 2016-2020. Only sites that met data completeness requirements (based upon 75 % or more of all possible daily values available between April 1 and September 30) were used to develop this map.

## 3.4 Projected Changes in Ozone

Current and projected ozone concentrations from recent ECCC and U.S. EPA modeling efforts were used to estimate how ozone concentrations are projected to change in the future. The modeling is described in further detail in Appendix A. The ECCC modeling was performed using GEM-MACH, a chemical transport model. Modeling scenarios are shown for base year 2015 as well as for projected BAU scenarios for 2025 and 2035. The projected years include regulations that will be enforced in the future. The U.S. EPA modeling data include results from the 2020 Ozone PA (US EPA, 2020d) and the Final Revised Cross-State Air Pollution Rule Update (RCU) (US EPA, 2020a).[17] The EPA modeling uses the Community Multiscale Air Quality Model (CMAQ) in the PA modeling and Comprehensive Air Quality Model with Extensions (CAMx) for the RCU modeling. Future year scenarios include activity growth and emission controls associated with "on the books" regulations in the U.S. – for example the effect of fleet turnover. In the RCU modeling Canadian emissions were developed from the 2015 base year and 2023 and 2028 future projection data provided to U.S. EPA by ECCC. More details on the inventories in the RCU project can be found in the emissions modeling technical support documentation (TSD) (US EPA, 2021c).

The ECCC and U.S. EPA modeling results are shown for the seasonal average of the MDA8 ozone concentration for May through September during the future year simulations. The modeling was performed using 2019 meteorology (ECCC) or 2016 meteorology (U.S. EPA), and by varying the projected emissions for different future years. Hence, the projected changes in the future are due entirely to changes in emissions of precursors, and the subsequent transportation, transformation, and fate of chemical constituents. Note that the ECCC and U.S. EPA modeled different reference and future years and the modeling uses different years for their meteorology and different emissions datasets, so the

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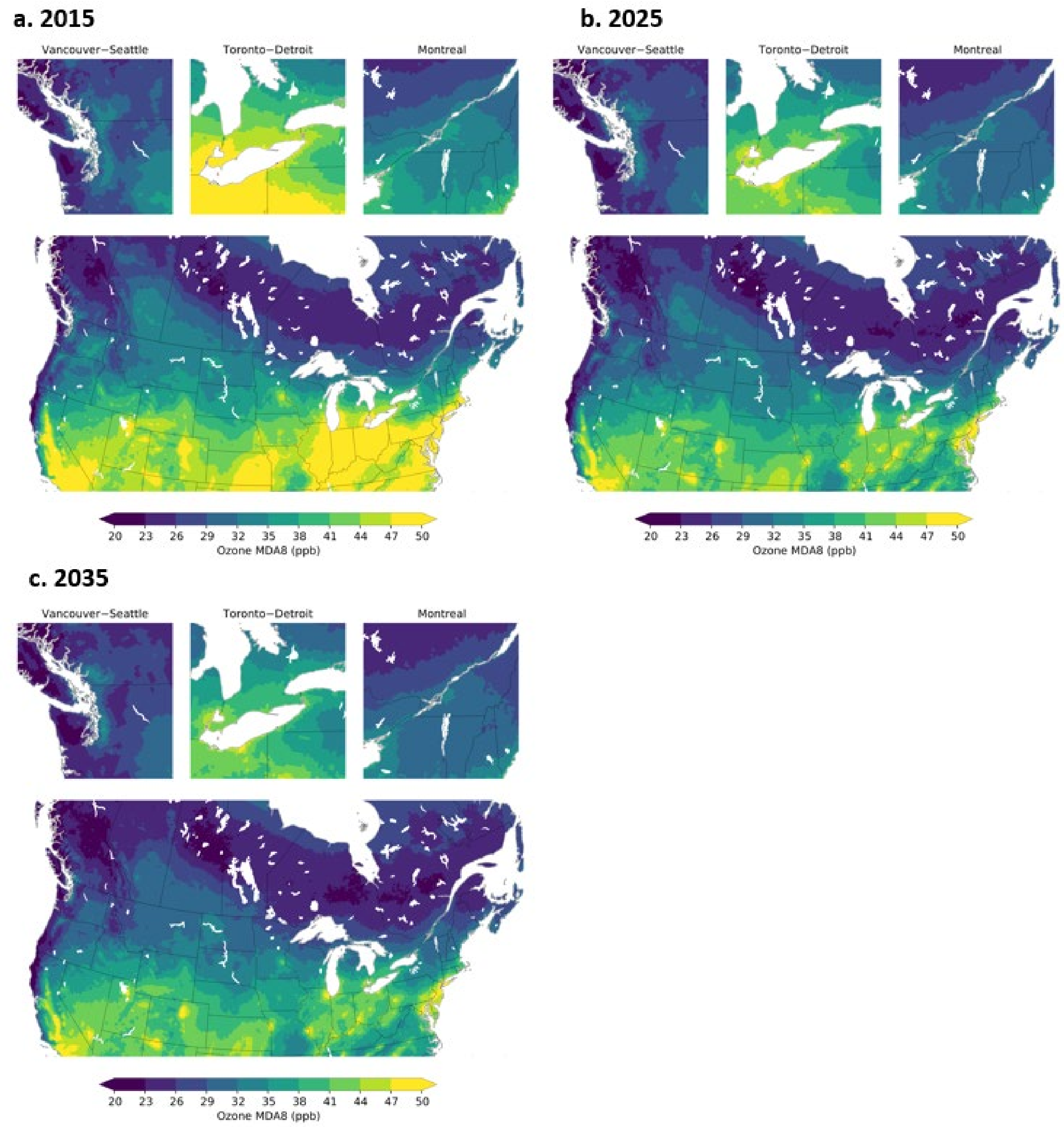
[17] Due to time and resource constraints, EPA's most current ozone modeling could not be included in this report. The new modeling for 2016 in the 2015 Ozone NAAQS Good Neighbor Plan (88 R 36654). https://www.epa.gov/system/files/documents/2023-03/AQ%20Modeling%20Final%20Rule%20TSD.pdf

outputs are not directly comparable. However, the modeling projections can be considered together qualitatively to gain more confidence in projected changes in ozone concentrations.

The ECCC modeling results are shown in Figure 3-6 for the base year 2015, and for two future scenarios – 2025 and 2035. The modeling for the 2015 base year is broadly consistent with the monitoring data shown in Section 3.3, with the highest ozone concentrations in the Great Lakes – St. Lawrence region and southern Maritimes. Similarly in the U.S., the highest ozone concentrations were estimated to the

*Figure 3-6. Seasonal average of the MDA8 ozone concentrations from ECCC modeling scenarios for May-September for the base year (a) 2015 and for future projections (b) 2025 and (c) 2035.*

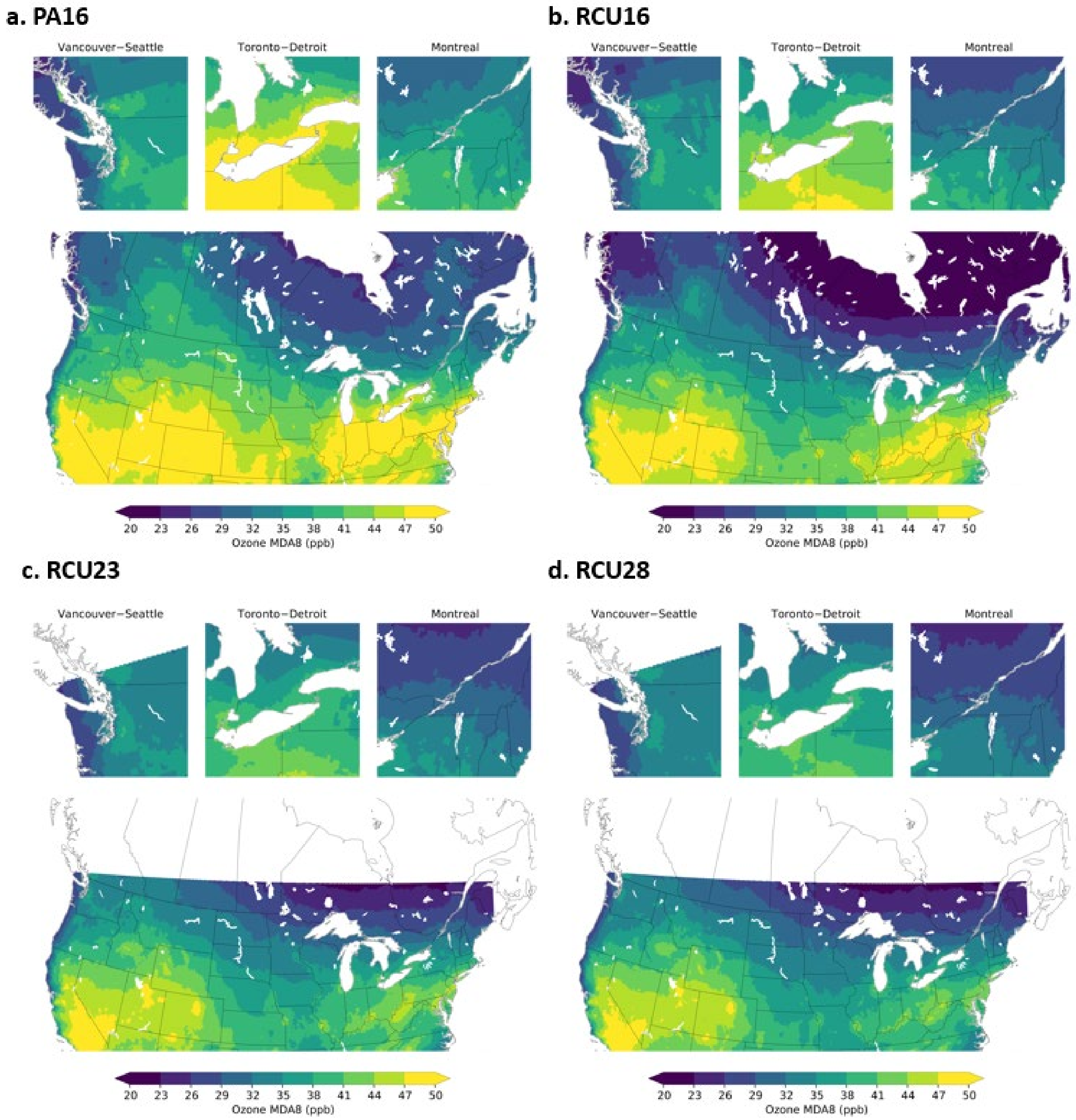


northwest and south of Lake Erie, corresponding to the urban centers of Detroit (Michigan) and Cleveland (Ohio). Somewhat elevated ozone concentrations are also captured by the model for parts of Alberta and southern British Columbia. Between 2015 and 2025, there is a substantial decrease in the modeled ozone concentrations, with the biggest changes in the U.S. and southern Ontario. These results are consistent with projected decreases in emissions of precursors ($NO_X$ and VOCs) due to continued actions in Canada and the U.S. (see Section 3.2). Between 2025 and 2035, there is little projected change in emissions, and ozone concentrations remain fairly stable. Despite the projected decreases in emissions and ozone concentrations between 2015 and 2035, projected annual 4th highest value of the MDA8 ozone concentrations for 2035 (not shown here) exceed the 2025 CAAQS of 60 ppb, in areas in southern Ontario, with the highest concentrations along the Ontario-Michigan border region.

The U.S. EPA modeling results are shown in Figure 3-7 for the 2016 base year, and for projections to 2023 and 2028. The PA and RCU baseline modeling for 2016 are broadly consistent with the monitoring data shown in Section 3.3 and the ECCC 2015 baseline modeling results. Modeled ozone concentrations are higher in the Ohio Valley region (Illinois, Indiana, Ohio, West Virginia) and southern Ontario, and lower in other areas. Notably, the south-to-north gradient of ozone is most pronounced along the eastern U.S. and Canada. The PA nominal present for base year 2016 (PA16) modeling has a less pronounced gradient than the RCU16. This is likely due to a combination of factors. The PA16 used an earlier version of emissions provided by ECCC, which had large emissions in Alberta associated with oil production. The RCU used a newer version where the Alberta emissions were revised downward. PA16 used the Community Multiscale Air Quality Modeling System (CMAQ) and RCU16 used the Comprehensive Air Quality Model with Extensions (CAMx). Though many fundamentals are shared, each model has its own implementation of chemistry, aerosols, transport, and deposition. In addition, the U.S. emission inventories are different versions of the 2016 modeling platform (*i.e.*, versions 2016fe and 2016fh). The magnitude of these differences is useful for understanding the range of reasonable results.

By comparing the three RCU plots in Figure 3-7, projected changes in ozone concentrations can be inferred. Ozone concentrations are projected to decrease between 2016 and 2023, with the largest decreases within the Ohio Valley and northeastern U.S. Between 2023 and 2028, ozone concentrations remain fairly steady. This is broadly consistent with the ECCC modeling results. These plots highlight the influence of projected decreases in emissions from both the U.S. and Canada, but more simulations are necessary to quantify and attribute the total contributions. Based on the RCU modeling, it is expected that locations in the U.S. that are within 500 km of the Canadian border will likely attain the ozone NAAQS threshold of 70 ppb before 2028. Note that the modeling does not consider the impact of climate change and increasing temperatures, likely to have an impact on ozone formation.

*Figure 3-7. Seasonal average of the MDA8 ozone concentrations from U.S. EPA modeling scenarios for May-September for the base year (a) PA16 and (b) RCU16, and for projections (c) RCU23 and (d) RCU28.*



## 3.5 Influence of Transboundary Flow

Ozone is known to have a long atmospheric lifetime ranging from 8 days to 120 days (Seinfeld & Pandis, 2006) that allows for long-distance transport. Previous air quality modeling by the Ontario Ministry of Environment and Climate Change[18] (MOECC, 2018) found large transboundary contributions to ozone near the Canada-U.S. border in southwestern Ontario.

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[18] Currently Ontario Ministry of Environment, Conservation and Parks

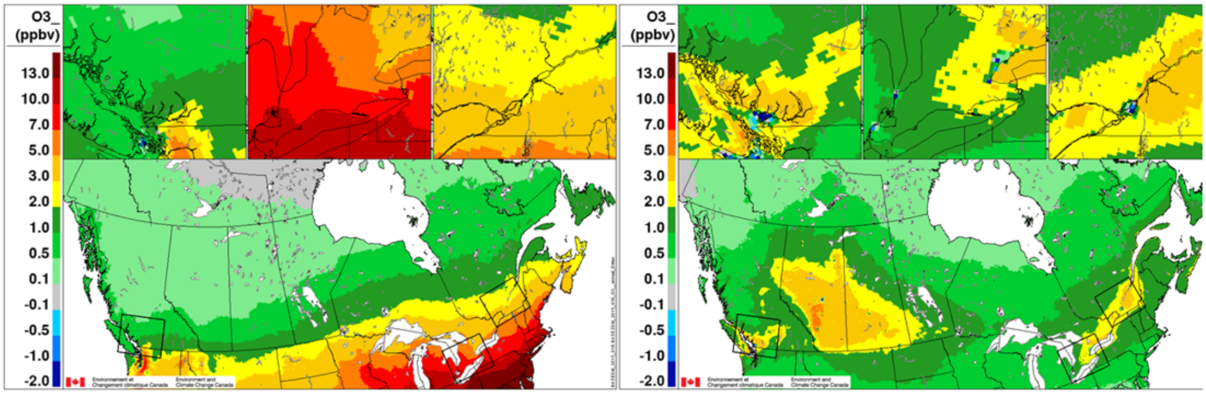
Appendix B shows a meteorological analysis of winds along the Canada-U.S. border. For ozone, we focus on the summertime when ozone production is high. During this time, aloft winds are generally west-to-east. Near the surface, the flows take on more of a south-to-north component particularly between Detroit and Windsor. This change in flow leads to transport from the U.S. to Canada being favorable during the ozone season.

To estimate the influence of emissions from Canada and the U.S., additional modeling runs were performed. The ECCC and U.S. EPA Ozone PA modeling runs were repeated, with anthropogenic emissions for each country set to zero (*i.e.*, zeroed out). To visualize the influence of the emissions from each country, a difference map can be calculated by subtracting the zeroed-out result from the base case (which includes the emissions from both countries). Where a difference map is positive, the base case has a higher value, meaning that the zeroed-out country has an influence in those regions. Note that for the U.S. EPA modeling, a zero-out of Canada only was not performed. Instead, both Canada and Mexico were simultaneously removed. The two countries are far enough apart and transport patterns are sufficiently different that the Canada (or Mexico) contribution can generally be distinguished by location. The zero-out modeling runs are a first order method of determining the impact of one set of emissions against another. As an intermediate step, 20% reduction runs for the 2015 emissions year were also performed with the ECCC model to better understand these results. The U.S. EPA RCU modeling uses the Ozone Source Apportionment Technology (OSAT) method to infer contributions from Canada and the U.S.

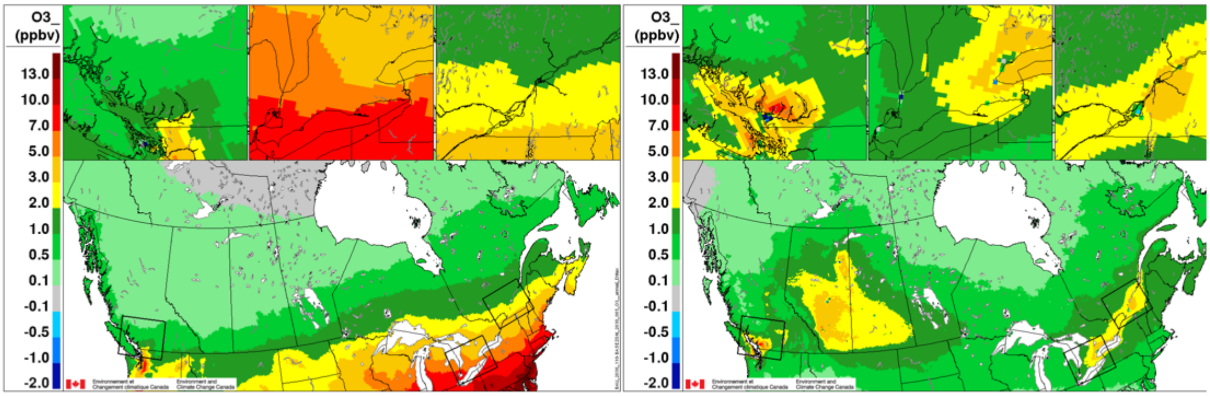
These modeling results are interpreted qualitatively due to the non-linear nature of ozone production. While $NO_X$ is a precursor for ozone under many conditions, in highly polluted regions it can titrate ozone leading to very low ozone concentrations, for example, in city centers or regions of low ventilation. In such cases, local ozone concentrations can be lower than background, and a reduction in $NO_X$ can lead to an increase in ozone since it is no longer being titrated. An example can be found in southern Quebec near the Canada-U.S. border, for which a decrease of 20% of either Canadian or U.S. emissions in the ECCC modeling (see Appendix A) results in a slight increase in ozone. In addition, ozone can be formed by either $NO_X$ or VOCs, such that in regions with lower $NO_x$ levels, ozone production can be controlled by $NO_X$ and vice versa. This factor adds another level of nonlinearity into ozone production. Since the zero-out scenarios are extreme cases and not realistic, quantitative interpretation of the zero-out modeling results is not presented here. The zero-out modeling can however be used to broadly demonstrate the relative influence of transboundary flow, such that qualitative interpretation of the results is presented here, with an indication of whether transboundary flow makes a large or small contribution to total concentrations.

Figure 3-8a shows the ECCC modeled annual average of daily maximum ozone concentration differences between the 2015 base case and the corresponding scenario without U.S. emissions, which can be used to estimate the influence of the U.S. emissions. The region that is most affected in Canada is southern Ontario, where there is a large gradient with concentration differences increasing from north to south. Smaller influences of U.S. emissions are also observed in southern Quebec, New Brunswick, Nova Scotia, and southern British Columbia. The influence of U.S. anthropogenic emissions on ambient ozone concentrations over the U.S. is significant, with the largest influence seen in the northeastern U.S. extending toward the south of Lake Superior, as well as around the Seattle area.

*Figure 3-8. Influence of Canadian and U.S. emissions on daily maximum ozone concentrations from the ECCC model. Differences in modeled annual average daily maximum ozone concentrations for 2015 when (a) U.S. and (b) Canadian emissions are zeroed out, and for 2035 when (c) U.S. and (d) Canadian emissions are zeroed out.*


**a. 2015 – Influence of U.S. Emissions**

**b. 2015 – Influence of Canadian Emissions**


**c. 2035 – Influence of U.S. Emissions**

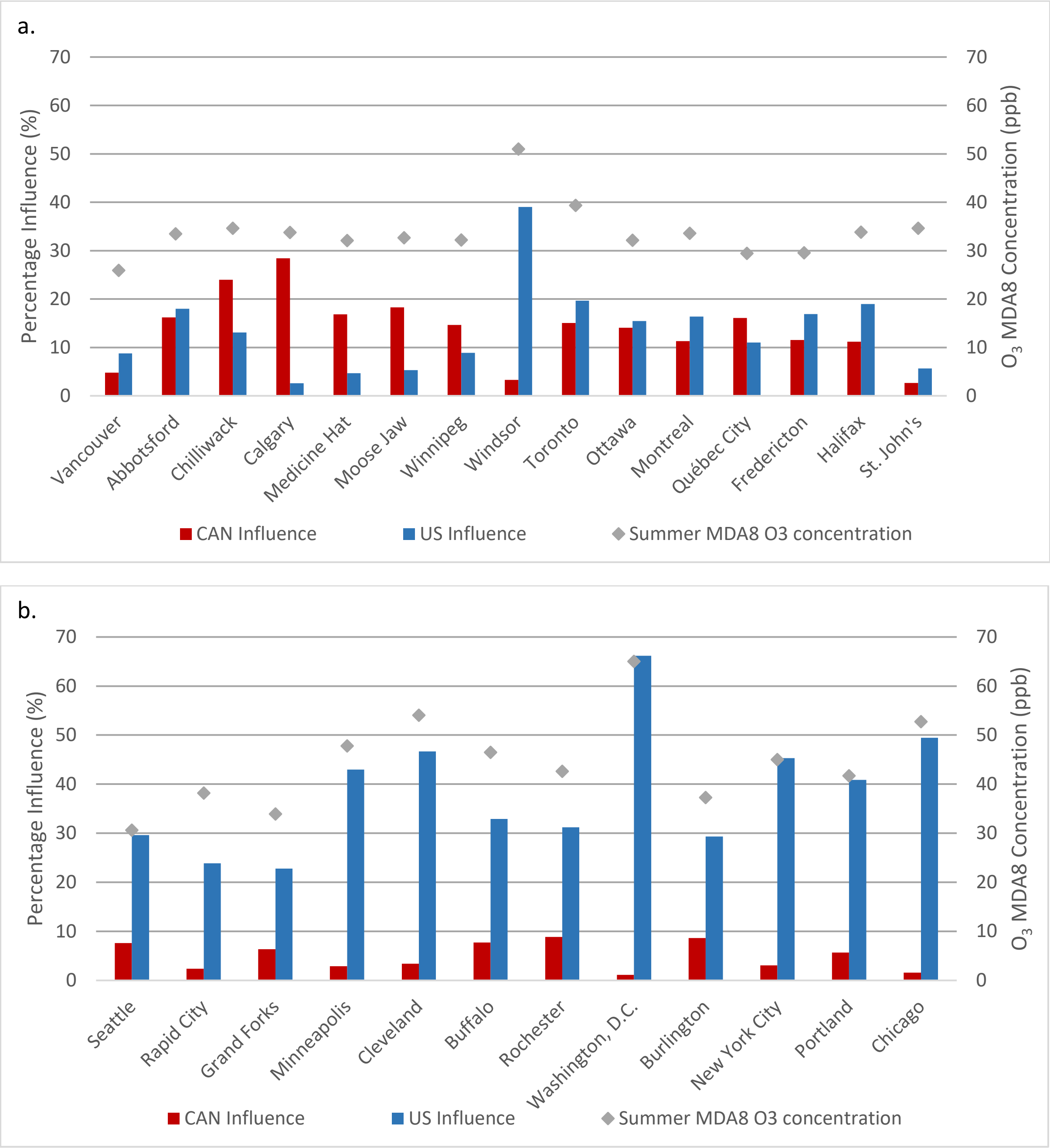
**d. 2035 – Influence of Canadian Emissions**

Figure 3-8b shows the ECCC modeled annual average of daily maximum ozone concentration differences between the 2015 base case and the corresponding scenario without Canadian emissions, which can be used to estimate the influence of Canadian emissions on both the U.S. and Canada. The positive values indicate a contribution from Canadian anthropogenic emissions. There is a strong influence of Canadian emissions on ozone concentrations in Canada, with the largest influence in southwestern British Columbia, in the greater Vancouver and Victoria area, southern Alberta, the Greater Toronto and Hamilton Area, and the Montreal area. In southern Ontario, the influence of U.S. emissions on ambient ozone concentrations (Figure 3-8a) is larger than the influence of Canadian emissions (Figure 3-8b).

ECCC modeling projections to 2035 were used to estimate future influences of emissions from the U.S. (Figure 3-8c) and Canada (Figure 3-8d). Concentrations of ozone are expected to decrease between 2015 and 2035, as described in Section 3.4, due to projected decreases in emissions. However, because the same 2019 meteorology is used for the 2015 and 2035 modeling, the impact of transboundary flow is similar in 2035 compared to the 2015 base year, and projected decreases are due to changes in emission sources. The modeling projections do not include the impact of climate change on meteorology.

Figure 3-9 shows the influence of Canadian and U.S. emissions on the seasonal average MDA8 ozone concentrations for May to September in 27 Canadian (Figure 3-9a) and U.S. (Figure 3-9b) cities located within 500 km of the Canada/U.S. border for the 2015 base year. For a given city, modeling data from the grid cell containing the latitude and longitude of the city were used. Note that the influences from Canada and the U.S. do not add to 100% because ozone concentrations are also affected by other factors, such as global background concentrations. The response in Canadian cities' ozone

*Figure 3-9. Influence of Canadian and U.S. emissions (left y-axis) and ozone concentration (right y-axis) for the 2015 summer average of daily maximum 8-hr ozone concentrations in (a) Canadian and (b) U.S. cities from ECCC modeling.*



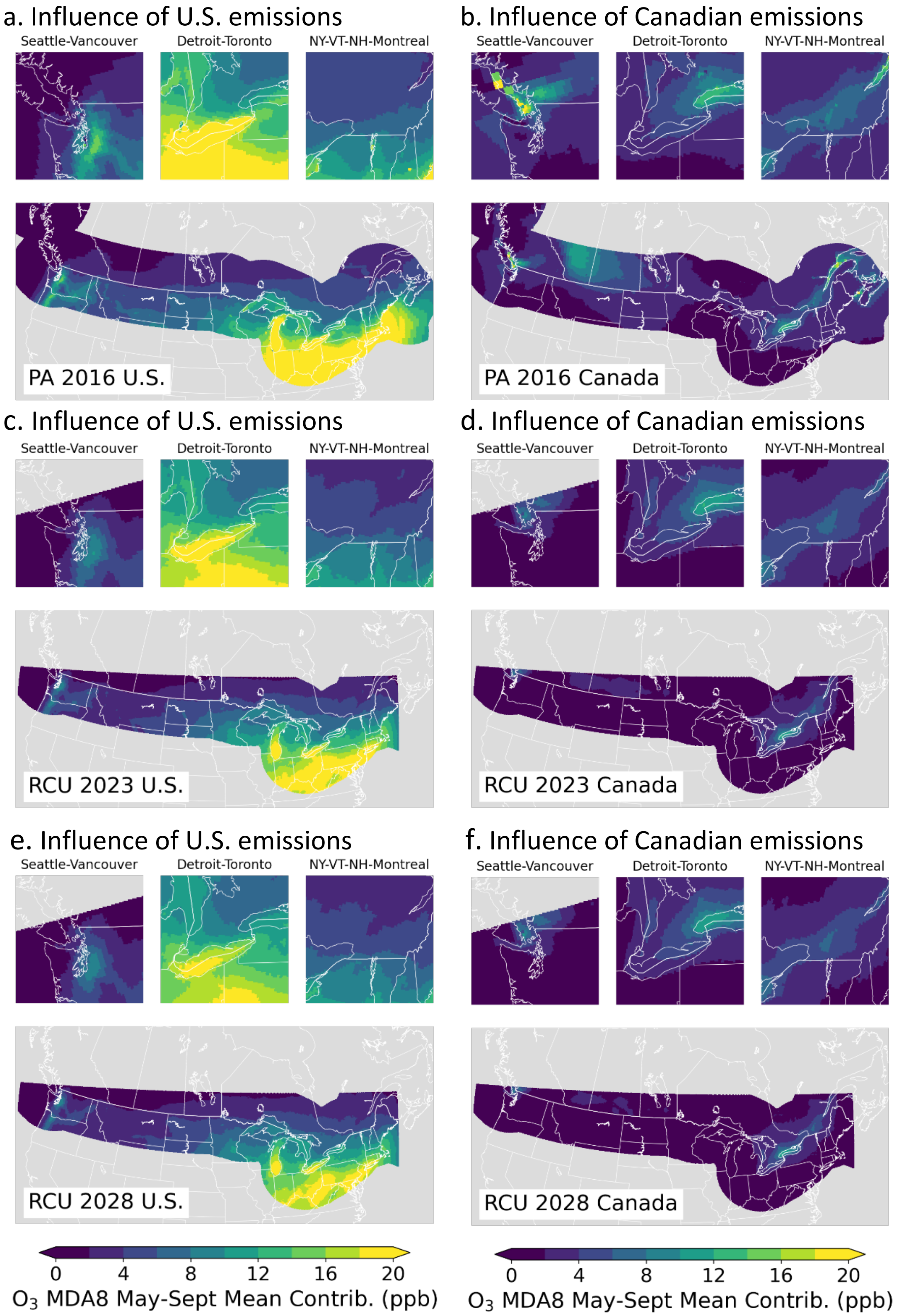
concentrations to U.S. emissions varies from less than 5% of the summer MDA8 ozone concentration in Calgary, where the modeled ozone concentration is approximately 35 ppb, to approximately 40% of Windsor's summer MDA8 ozone concentration, where the modeled ozone concentration is approximately 50 ppb. For more than half of the Canadian cities shown, the response to the U.S. emissions is larger than the response to the Canadian emissions, particularly for the cities located closer to the Canada-U.S. border. Across all Canadian cities listed, the majority of ozone concentrations are influenced by factors beyond U.S. and Canadian emissions, such as global background ozone levels. The response in U.S. cities' ozone concentrations to Canadian emissions varies from less than 2% in Washington D.C., where the summer MDA8 ozone concentration is approximately 65 ppb to up to approximately 10% in Rochester, New York where the summer MDA8 ozone concentration is approximately 40-45 ppb. For all the U.S. cities shown, a much larger response is attributed to U.S. emissions than Canadian.

The U.S. EPA modeling of influences from U.S. and Canadian emissions are shown in Figure 3-10 for the MDA8 ozone concentration averaged from May to September. The U.S. EPA modeling includes two projects with varying levels of detail. The Ozone PA simulations include a 2016 simulation and zero out attribution from coarse hemispheric (108-km) and fine regional simulations (12-km). The RCU project included a 2016 simulation and projections to the future for 2023 and 2028. Only the RCU figure simulations (2023 and 2028) included source apportionment-based attribution. For more details, see Appendix A. The U.S. zero out modeling results are shown as absolute MDA8 ozone concentrations from May to September and are not directly comparable to the ECCC modeling differences in modeled annual average daily maximum ozone concentrations. However, the broad patterns in the transboundary transport can be compared between the modeling runs.

Figure 3-10 shows the ozone concentrations attributed to emissions from the U.S. for the 2016 base case. The U.S. contribution is largest in the Ohio Valley and the northeastern U.S. In all years, the largest contribution of the U.S. ozone concentrations to Canada occurs in the Windsor-Quebec Corridor, with smaller contributions observed in southern Quebec, and southern British Columbia. Some smaller contributions can also be seen in the 2016 base case in New Brunswick and Nova Scotia, but these areas are outside the geographic scope of the RCU modeling. This is broadly consistent with areas of largest influence observed in the ECCC zero out modeling results. Areas designated nonattainment of the 2015 ozone NAAQS that are within 500 km of the border have relatively large U.S. local contributions (Figure 3-10a), and low contributions from Canada (Figure 3-10b).

RCU23 and RCU28 modeling was used to project the U.S. and Canada contributions to ozone concentration for future years (Figure 3-10, panels c-f). Smaller contributions are found in future years, which is consistent with projected emissions reductions (see Section 3.4). However, the PA16 used a different model and a different attribution technology, which prevents a quantitative comparison between the attributions in 2016 and 2023. The differences in attribution between 2023 and 2028, both for RCU modeling, show continued reductions in ozone. Furthermore, the patterns in influence of transboundary flow are consistent between 2016 and future modeled years, with transboundary

*Figure 3-10. Attribution of ozone for PA16 from (a) U.S. emissions and (b) Canadian emissions; for RCU23 from (c) U.S. emissions and (d) Canadian emissions; and for RCU28 from (e) U.S. emissions and (f) Canadian emissions. All results are MDA8 ozone concentrations averaged from May-Sept.*



influence from the U.S. into southwestern Ontario evident for all modeling years. This is broadly consistent with the findings of the ECCC modeling.

Differences in Canadian contributions to ozone concentrations are observed in Alberta between the PA16 (Figure 3-10b) and the RCU23 and RCU28 modeling runs (Figure 3-10d,f). As noted in Appendix A, the RCU23 modeling was performed for the 12 km domain with boundary conditions from a 36 km domain. The larger domain did not track attribution, so boundary conditions in the 12 km domain cannot distinguish the Canadian contribution. As a result, Canada's emissions outside the domain are categorized as part of the "boundary conditions" rather than as attributed to Canada. As a result, the RCU23 and RCU28 values near the domain edge may underestimate Canada's contribution. Again, the difference between PA16 and RCU23 reflect model and methodology differences described above in addition to the emission year. Even so, a decreasing trend from 2016 to 2023 and 2028 is observed. Note that the same color scale has been used for the U.S. and the Canadian contributions, which allows for comparison.

Figure 3-11 shows the ratio of U.S. and Canada contributions from the U.S. EPA modeling PA16, RCU23 and RC28 modeling runs. These results can be used to identify areas where the contributions from the U.S. and Canada are relatively similar (light red, white and light blue) or where one country's contributions dominate. Red indicates areas where U.S. contributions are larger than Canada's and blue indicates the reverse. Transport is clearly important when the U.S. contribution in Canada is comparable to the Canadian contribution, and vice versa.

In the U.S., the figures show that Canadian ozone contributions are most often less than a quarter of U.S. contributions. Contributions from Canada can be closer to half of U.S. contributions in a narrow band along the northern edge of Montana, North Dakota, New York, Vermont, New Hampshire, and Maine. The maximum ozone contribution of Canada in the U.S. is in the northwest corner of Washington. These influences can also be seen in Figure 3-10.

In Canada, the U.S. ozone contributions are frequently larger than Canadian contributions. This is most often the case in less populated areas. For example, U.S. contributions can be four times the Canadian contributions in the western portions of Ontario. In the more populated parts of Ontario, however, the U.S. contribution is two to two-thirds times the Canadian contribution. One notable area of exception is Windsor and the entirety of Essex County, where larger levels of transboundary influence can be seen due to the area's close proximity to the border and U.S. metro areas. Although emissions from the U.S. influence ozone concentrations in Canada, it is noted that local emissions and local formation of ozone in some areas of eastern Canada play a significant role in elevated ozone concentrations close to or above the CAAQS. Furthermore, the *Canadian Smog Science Assessment* (Environment Canada, 2012) indicated that the intercontinental transport of pollutants from Asia into North America can occur through the winter and early spring, contributing to background concentrations[19] at the surface and leading to increases in ambient air pollution across Canada and the U.S., from the mid-latitudes extending into the Arctic. Intercontinental transport of ozone and precursors (*e.g.*, from Asian emissions) is explicitly captured in the U.S. EPA modeling at the hemispheric scale that provides hourly boundary conditions to the finer scale modeling. The ECCC modeling implicitly includes long-range transport via climatological boundary conditions. The climatological boundary conditions represent

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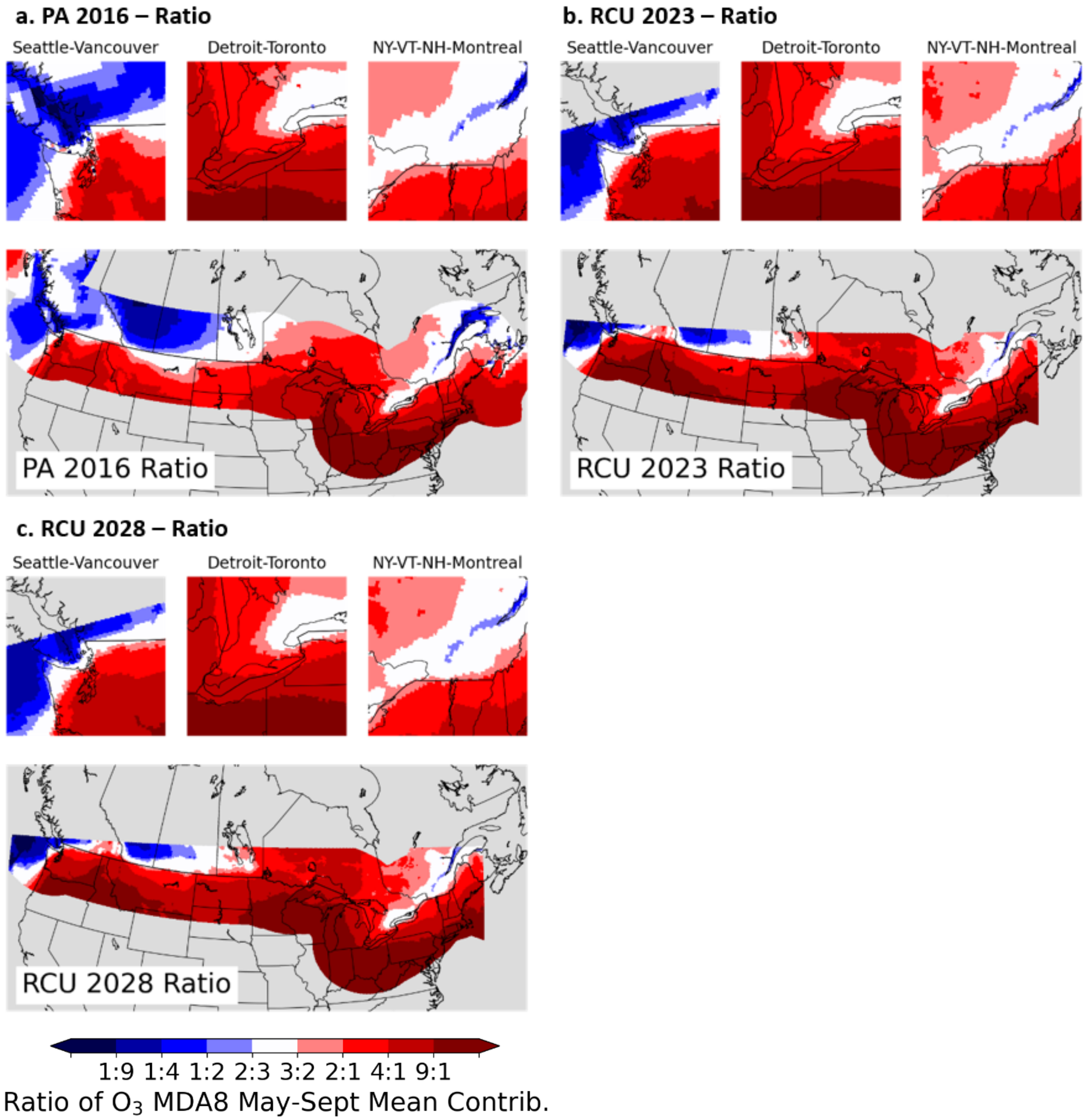
[19] Background concentrations in winter and spring also include natural sources of ozone including ozone transported into the troposphere from the stratosphere.

concentrations of ozone and precursors that flow in from outside the domain, and therefore includes trans-pacific transport (*e.g.*, from Asia) in an average sense, but would not include episodic behavior associated with long-range transport events.

*Figure 3-11. Ratio of ozone from U.S. to Canada from U.S. PA16 (a), RCU23, (b), and RCU28 (c) modeling scenarios.*



## 3.6 Health Impacts

Short-term exposure to ozone can cause a broad range of respiratory effects depending on the level of exposure. These health effects range from small, transient and/or reversible changes in lung function, airway inflammation and respiratory symptoms (*e.g.*, coughing, scratchy throat, and pain when taking a deep breath), to more serious health outcomes such as emergency department visits and hospital admissions (US EPA, 2020c). In addition to the above effects, Health Canada concluded that acute exposure to ozone likely causes an increased risk of total non-accidental and cardiopulmonary mortality (Health Canada, 2013). Studies of short-term exposures to high levels of ozone also report a likely association with metabolic effects (US EPA, 2020c). Long-term ozone exposure is suggested to be linked to aggravation of lung diseases such as asthma and chronic obstructive pulmonary disease (COPD) and may lead to asthma development (US EPA, 2020c). Studies in locations with elevated concentrations also report associations of long-term exposure to ozone with deaths from respiratory causes. Certain populations, such as people with pre-existing respiratory conditions (*e.g.*, asthma), children, older adults, outdoor workers, people carrying certain gene variants (*e.g.*, antioxidant enzyme and inflammatory mediator variants), and people with reduced intake of certain nutrients (*e.g*., vitamins C and E) are at higher risk for health effects (US EPA, 2020c). Health Canada has concluded that there is a lack of evidence of a threshold below which there is no risk to population health (Health Canada, 2013). In Canada, the health burden of above-background ozone in 2016 was estimated to be 4,100 premature deaths annually plus large numbers of adverse respiratory outcomes, with an economic cost of $30 billion per year (2016 Canadian dollars) (Health Canada, 2021).

Population-weighted averages are used to estimate the ambient concentrations that a population is, on average, exposed to. As such, population-weighted concentrations are a good indicator of the direction and magnitude of health impacts. ECCC and U.S. EPA zero-out modeling runs were used to estimate contributions of U.S. and Canadian sources to the population-weighted average May-Sept mean MDA8 ozone concentration. This analysis uses the Gridded World Population dataset version 4 for population estimates on a 2.5 arcminute grid. Ozone concentration data were interpolated to the grid with the population data to calculate the population weighted average. Contributions to population-weighted average concentrations are reported in Figure 3-12 and Figure 3-13. Each panel represents a different population: (a) all people within the 500 km buffer or a PEMA state, (b) the U.S. population within that area, and (c) the Canadian population within that area. Each source definition represents a subset of emission sources: “All” represents the total emissions sources from Canada and the U.S., “U.S.” represents only the anthropogenic sources within the U.S., and “Canada” represents only the anthropogenic sources within Canada. For the U.S. EPA modeling, the “Canada” label includes emissions from both Canada and Mexico. However, Mexican contributions are small enough to neglect near the Canadian border, and therefore this is labeled as Canada.

Ozone population-weighted averages from the ECCC modeling are shown in Figure 3-12. For all populations within the 500-km buffer on either side of the border or a PEMA state (Figure 3-12a), larger contributions to population-weighted average ozone are found from U.S. emissions, and smaller contributions from Canadian emissions[20]. This is expected because the population in the U.S. is larger and has a concomitant larger impact. Similar contributions are observed for U.S. populations within the 500-km buffer or a PEMA state (Figure 3-12b). For populations in Canada within the 500-km buffer (Figure 3-12c), emissions from Canada and the U.S. have similar influences on population-weighted average ozone concentrations. Total population-weighted ozone concentrations (“All” in Figure 3-12a) are lower in the Canadian portion of the 500-km buffer than in the U.S. portion. Population weighted ozone concentrations are projected to decrease in both Canada and the U.S. between 2015 and 2025,

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[20] Note that the sum of the anthropogenic U.S. emissions and the Canadian emissions is lower than “All” because there is also a significant component from natural and intercontinental transport.

primarily due to a drop in contributions from U.S. sources. Between 2025 and 2035, there is little change in projected population-weighted ozone concentrations.

The U.S. EPA modeling of population-weighted averages is shown in Figure 3-13 and is broadly consistent with the ECCC modeling. Within 500 km of the border on the U.S. side (Figure 3-13b), total 2016 average MDA8 ozone concentration is projected to decrease by approximately 10% by 2028. The Canadian contribution is generally small, as is the magnitude of decreases between years. Trends cannot

*Figure 3-12. ECCC modeled population-weighted May-Sept mean MDA8 ozone contributions (ppb) from sources (All, U.S., Canada) to populations within 500-km or the PEMA states, the U.S. portion, or the Canadian portion.*

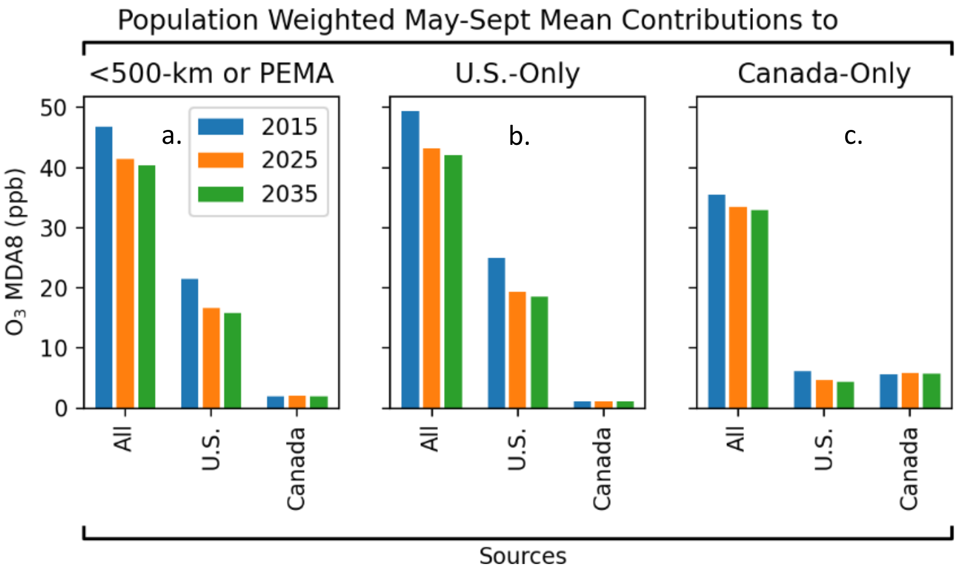
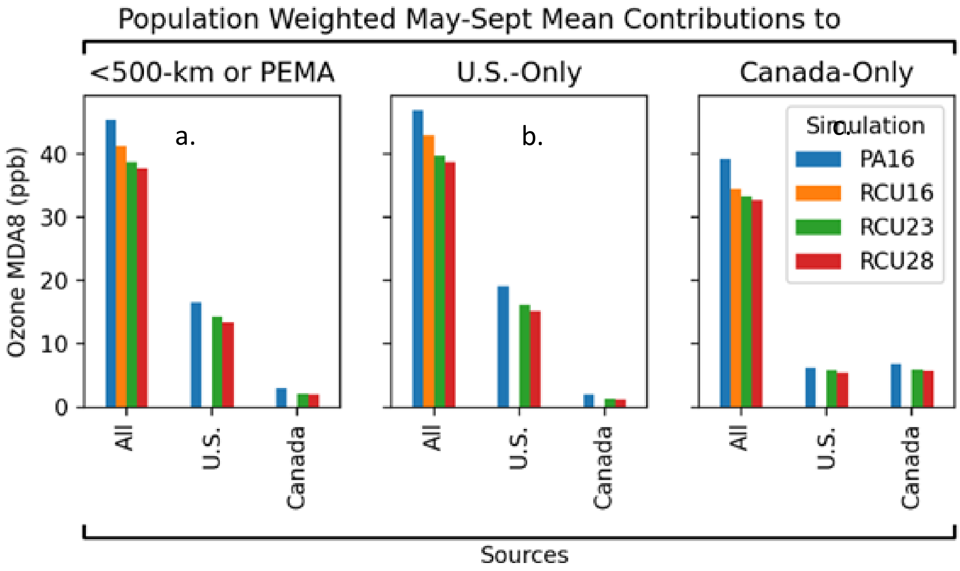


*Figure 3-13. U.S. EPA modeled population-weighted May-Sept mean MDA8 ozone contributions (ppb) from sources (All, U.S., Canada) to populations within 500-km or the PEMA states, the U.S. portion, or the Canadian portion.*

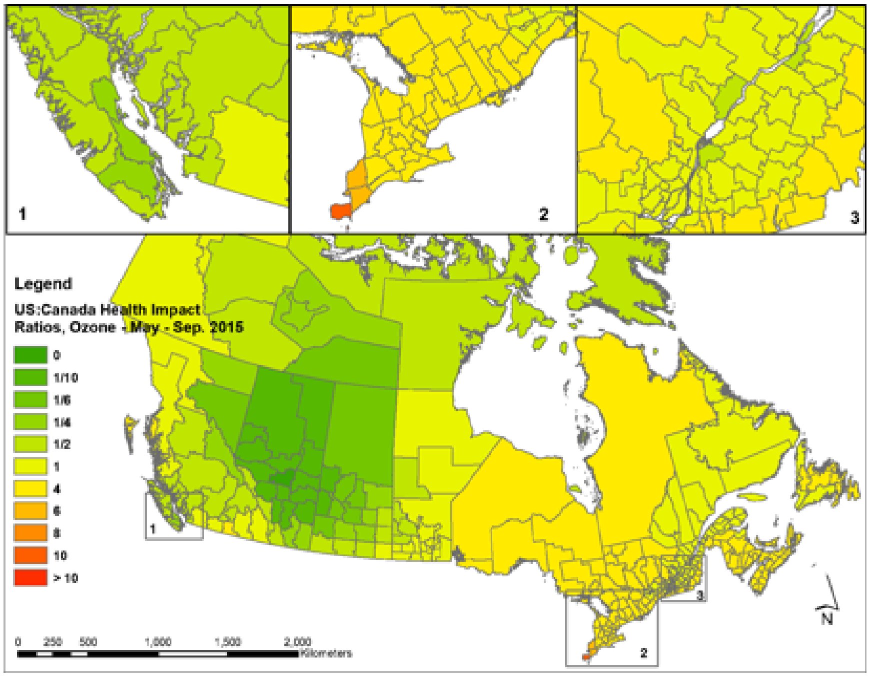


be quantitatively determined by direct comparison of PA16 and RCU23 modeling results for U.S. and Canada-specific contributions. However, decreased emissions in the U.S. shown in the “All” category between RCU16 and RCU23 can largely be seen as coming from U.S. sources, given the relatively marginal Canadian contributions in this area. Within 500 km of the border on the Canadian side (Figure 3-13c), total 2016 MDA8 ozone concentration decreases by 5% in 2028. For population weighted values, the U.S. and Canadian contributions are nearly equal in 2016. Based on the RCU modeling, the contribution magnitudes from the U.S. in Canada are decreasing more rapidly from 2023 to 2028 than the contributions from Canada. The U.S. contribution decreases are larger in magnitude on the U.S. side of the border than on the Canadian side, though similar in percentages.

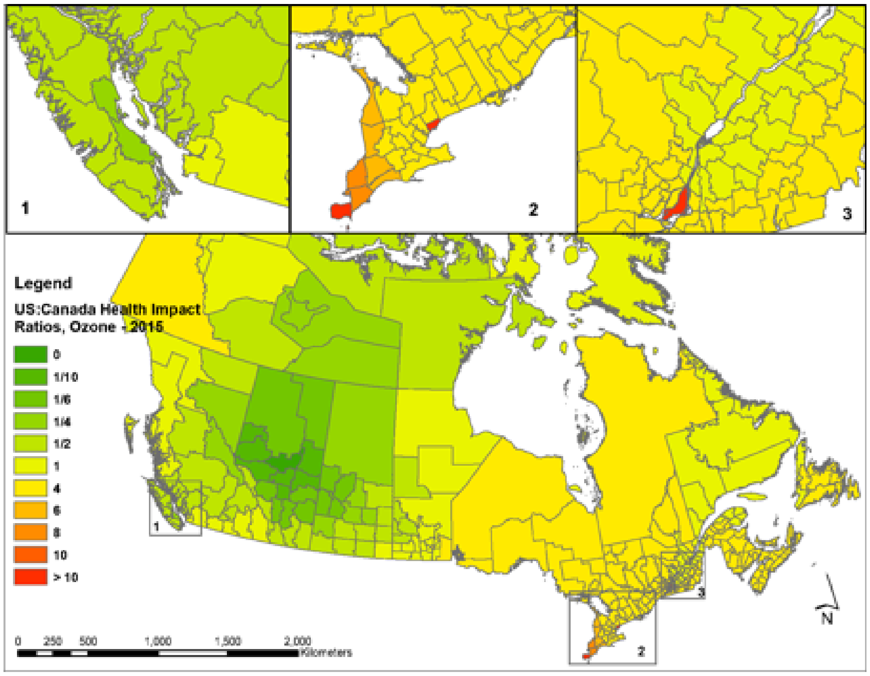
To attribute ozone health impacts in Canada to U.S. and Canadian sources, Health Canada applied the Air Quality Benefits Assessment Tool (AQBAT v3) (Judek et al., 2019) to data produced by the ECCC zero-out modeling runs for 2015, as in Pappin et al., (2024). Since the modeling runs should be interpreted qualitatively, as described in Section 3.5, the health impact analysis should also be interpreted accordingly. Figure 3-14 maps the estimated ratios of source contributions from the U.S. to those from Canada to total ozone health impacts by Canadian census division. Areas in orange and yellow suggest that U.S. sources contribute more to local health impacts than Canadian sources, while areas in green suggest larger contributions from Canadian sources. The transboundary flow of ozone and its precursors from the U.S. to Canada contributes to a significant portion of health impacts in Central and Atlantic Canada and British Columbia and is the dominant source of health impacts overall in Ontario, Quebec, Manitoba, Nova Scotia, New Brunswick, Newfoundland and Labrador, and Prince Edward Island (Pappin et al., 2024). Some large areas of Quebec and Newfoundland and Labrador see comparable contributions from Canadian and U.S. sources, though many (but not all) are relatively unpopulated. In addition, one-third of the total contribution of health impacts from transboundary ozone occurs in Canadian provinces where U.S. contributions are less than Canadian contributions (ratios < 1). Transboundary ozone health impacts mostly occur in Ontario and Quebec within 300 km of the Canada-U.S. border. Health impacts in Canada attributable to transboundary ozone are projected to decline from 2015 to 2025, and increase from 2025 to 2035 due in part to an increasing number of Canadians susceptible to adverse health effects as a result of ageing, and population growth due to higher immigration (Pappin et al., 2024). Note that these results do not consider the impact of climate change on meteorology and the resulting impacts on air quality and health.

*Figure 3-14. Ratio of contributions from U.S. sources to those from Canadian sources to total ozone-related health impacts in Canada (estimated as an economic value per year) by census division. (a) Ratios include health impacts estimated using May-September average daily maximum 1-hr ozone concentrations; (b) Ratios include health impacts estimated using annual average daily maximum 1-hr ozone concentrations. Ratios > 1 indicate that U.S. sources contribute more to local health impacts than Canadian sources, while ratios < 1 indicate that Canadian sources contribute more than U.S. sources. Ratios are based on zeroing-out in GEM-MACH for 2015. (Adapted from Pappin et al., 2024)*

a. Ratios for May-September average daily maximum 1-hr ozone concentrations



b. Ratios for annual average daily maximum 1-hr ozone concentrations



## 3.7 Environmental Impacts

There is a causal relationship between ozone exposure and impacts on vegetation (US EPA, 2020c). Ozone is taken up by plants through pores, or stomates, in their leaves and can cause direct physical damage resulting in premature plant aging, reduced photosynthesis, reduced plant reproduction, visible foliar symptoms (*e.g.*, discoloring, necrosis, *etc.*), reduced uptake of carbon dioxide, increased vulnerability to pest attacks, increased tree mortality, reduced primary productivity, and more (US EPA, 2020c). These changes at the individual plant level can lead to reductions in growth and yield of vegetation, reduction in carbon storage capacity and ecosystem changes as plants that are more resistant to ozone can become dominant. The extent of damage to the plant depends on the ambient ozone concentration, the vegetation's ozone detoxification capacity, the size of the stomata opening (which is impacted by time of day), the moisture content of the soil and meteorological conditions. Ozone can also reduce the capacity of forest ecosystems to provide ecological services such as air filtration and carbon sequestration. These deleterious impacts on vegetation may translate into reduced crop yields (US EPA, 2020c).

To estimate the impacts of transboundary ozone flows on Canadian crop production, ECCC used the agricultural module of the Air Quality Valuation Model (AQVM2) to estimate the change in sales revenue for the agricultural sector for 19 different types of crops in Canada associated with different levels of ground-level ozone. First, modeled ozone concentrations produced by GEM-MACH are fed into exposure-response functions for both the baseline and emission reduction scenarios to estimate yield reductions. Next, these yields are multiplied by the hectares of crops seeded in year 2016 and the crop prices. The process is repeated for each crop type in each of the 68 Census Agricultural Regions in Canada, and aggregated provincially to estimate the potential change in sales revenues for crop producers.

The estimated ozone-related crop yield reductions attributable to Canadian and transboundary emissions are of similar magnitude. Such an outcome can be expected because ozone and ozone precursors (such as $NO_X$) can travel over larger distances from transboundary sources into Canadian territory. Due to the extensiveness of farmlands in Alberta, Saskatchewan and Manitoba and the significant ozone precursor emissions originating from Alberta, the Prairies' share of the yield reductions from Canadian emissions is larger. However, estimates of crop yield reductions stemming from transboundary emissions are relatively higher along the Windsor-Quebec City Corridor. It is worth cautioning that the crop yield reductions are not directly proportional to the change in ambient air quality as they also depend on the cultivated area and its proximity to emission sources, so areas experiencing the highest ozone concentrations may not always experience the highest crop yield reductions.

## 3.8 Methane as an Ozone Precursor

Methane emissions in Canada and the U.S. also contribute to global tropospheric ozone production. Because methane has a long lifetime in the atmosphere, methane concentrations are well-mixed, and a decrease in tropospheric methane emissions anywhere eventually leads to a decrease in tropospheric ozone concentrations everywhere. However, ground-level ozone concentrations, particularly in urban areas, can depend on local and regional chemistry. Impacts of local methane enhancements on ozone production needs further exploration. Several studies have developed approaches for quantifying the effects of methane emission reductions; the most recent is the Global Methane Assessment (UNEP, 2021). The Global Methane Assessment evaluates benefits of a 10 Mt methane emission reduction, which include fewer ozone-related respiratory deaths (428[21] in the U.S. and 35[22] in Canada) and fewer asthma-related emergency department visits (1,500 in the U.S. and 224 in Canada)[23]. Other benefits include fewer ozone-related cardiovascular deaths, increased crop production, and less ecological damage to natural vegetation. Although methane emissions near the U.S.-Canada border would not disproportionately impact transboundary conditions due to its high rate of global mixing , it plays a role as a contributor to ozone production in the U.S., Canada.

## 3.9 Summary

In 2000, the Ozone Annex (Annex 3) to the AQA set out commitments by Canada and the U.S. to reduce emissions of $NO_X$ and VOCs that contribute to transboundary ozone pollution. These commitments aimed to help both countries attain their respective air quality goals, and to protect human health and the environment. At the time the Annex was introduced, the long-term goal was to ensure that ambient concentrations of ozone did not exceed the respective ozone air quality standards in each country.

Canada and the U.S. met their commitments in the Ozone Annex to reduce emissions of $NO_X$ and VOCs from stationary and mobile sources and from solvents, paints, and consumer products. Canada's national emissions of $NO_X$ and VOCs have decreased by 36% and 49%, respectively, between 1990 and 2020. The U.S. national air emissions of $NO_X$ and VOCs decreased by 70% and 48% respectively between 1990 and 2020.[24]

Ambient ozone concentrations have also declined within the Canada-U.S. border region since the establishment of the Ozone Annex. Annual 4th highest MDA8 ozone concentrations have decreased by more than 10 ppb at many monitoring stations across Ontario, Quebec, and the Maritimes, and by as much as 20 ppb at some stations in the Great Lakes states and Ohio Valley, where ozone concentrations are highest. Regulatory programs and non-regulatory programs designed to meet emissions reduction commitments in the Ozone Annex, as well as programs designed to meet other goals, regulatory or statutory requirements for Canada and the U.S. individually, have contributed to the reduction in ozone concentrations.

Despite the progress made, areas in southern Ontario and southern Quebec continue to have higher ozone concentrations which approach or exceed the current 2020 ozone CAAQS of 62 ppb. Modeling projections suggest that ozone concentrations are expected to decrease in future years, as new regulations targeted for emission reductions take effect. However, southern Ontario is projected to have 4th highest MDA8 ozone concentrations that will continue to exceed the 2025 ozone CAAQS (60 ppb) in 2035, based on ECCC air quality modeling.

Transport in the northeastern U.S. and Ohio Valley is favorable for predominantly U.S.-to-Canada transport, and transport from the U.S. continues to contribute a large fraction of anthropogenic ozone in Canada. U.S. EPA and ECCC modeling results of U.S. and Canadian emissions influences are broadly consistent – the largest influence continues to be in the Windsor-Quebec Corridor with emissions from the U.S. also affecting ozone concentrations in southwestern British Columbia, in the greater Vancouver

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[21] The 95% confidence intervals are (287-553) for total ozone-related respiratory deaths in the U.S.

[22] The 95% confidence intervals are (24, 46) for total ozone-related respiratory deaths in Canada.

[23] The uncertainty in asthma-related accident emergency department visits is ±37%.

[24] https://www.epa.gov/air-trends/air-quality-national-summary

and Victoria area, southern Alberta, the Greater Toronto-Hamilton area, and the Montreal area for both current emissions and future projected emissions. The impact of U.S. emissions on ozone concentrations in Canadian cities within 500 km of the border ranges from less than 5% in Calgary up to approximately 40% in Windsor based on ECCC modeling results. Canada-to-U.S. ozone transport can also be seen to a lesser extent within a narrow band along the border in northern Montana, North Dakota, New York, Vermont, New Hampshire, and Maine with the highest area of influence in northwest Washington.

Ozone continues to have significant impacts on public health and agricultural production in the U.S. and Canada. Transboundary transport is seen in both countries, but has a larger influence in Canada, leading to greater impacts on air quality and health in Canada from U.S. air pollutant emissions than in the U.S. as a result of Canadian air pollutant emissions. Modeling results predict that transboundary flow of ozone and its precursors from the U.S. to Canada contributes to a significant portion of health impacts in central and Atlantic Canada and is the dominant source of health impacts in Ontario, Quebec, Nova Scotia, New Brunswick, Newfoundland and Labrador, and Prince Edward Island. Analysis also suggests that transboundary flow from the U.S. into Canada also causes crop yield reductions, particularly along the Windsor-Quebec City Corridor.

# 4 Fine Particulate Matter

*What is $PM_{2.5}$:* PM is a general term used to describe a mixture of solid particles and liquid droplets suspended in the air. PM is characterized according to size, and includes coarse PM ($PM_{10}$), which consists of particles with diameters that are 10 microns (μm) in diameter and smaller, and fine PM ($PM_{2.5}$), which consists of particles with diameters 2.5 μm or smaller. This assessment focuses on $PM_{2.5}$, due to its substantial health burden and because this size fraction can remain suspended in the air for several days to weeks, can be transported by winds over large distances, and thus is subject to atmospheric transboundary transport in North America. $PM_{2.5}$ directly emitted to the atmosphere (also called primary $PM_{2.5}$) originates from sources such as dust, sea spray, or combustion sources. Secondary $PM_{2.5}$ is formed via chemical reactions in the atmosphere involving precursor gases originating from a wide variety of transportation, combustion and industrial sources. Key $PM_{2.5}$ precursor gases include $NH_3$, $SO_2$, $NO_X$, and VOCs.

*Previous bilateral science assessments on $PM_{2.5}$ under the AQA:* Canada and the U.S. have previously explored adding an annex to the AQA to address transboundary $PM_{2.5}$. The two countries agreed to work on the necessary scientific, technical, and regulatory foundations required to consider adding a PM annex to the AQA, including a suggestion to update the Canada-U.S. Transboundary Particulate Matter Assessment that was completed in 2004. The 2004 assessment (US EPA & Environment Canada, 2004) found that PM concentrations varied significantly over geographic regions, and that transboundary transport of PM and PM precursors can be significant enough to compromise attainment of national standards in both countries (US EPA & Environment Canada, 2004). In 2011, Canada and the U.S. created the Regulatory Cooperation Council to better align the regulatory approaches of the two countries, where possible. One of the environment-related initiatives required the two countries "to consider the expansion of the Canada-U.S. AQA to address transboundary PM, the air pollutant most commonly associated with premature mortality, based on comparable regulatory regimes in the two countries" (Regulatory Cooperation Council, 2011).

The 2013 Transboundary Particulate Matter Science Assessment was completed by Canada and the U.S. and concluded that "because of the important health and environmental effects associated with $PM_{2.5}$, it would be beneficial for both countries to track progress and exchange information relevant to achieving $PM_{2.5}$-related emissions reductions, air quality improvement, and program implementation" (ECCC & US EPA, 2016). It further stated that "there would be value in addressing $PM_{2.5}$ in some manner under the Agreement." As such, Canada and the U.S. agreed to include $PM_{2.5}$ as part of this review and assessment. This review and assessment reflects the evolution in the scientific understanding of $PM_{2.5}$-related impacts on human health, advances in modeling tools used to estimate impacts, and regulations implemented in both countries since 2013.

## 4.1 Air Quality Standards and Guidelines

As described in Section 3.1, both Canada and the U.S. have established air quality standards to protect human and ecosystem health. Table 4-1 provides a comparison of the most current CAAQS and the U.S. NAAQS for $PM_{2.5}$. The metrics (or statistical forms) of the Canadian and U.S. standards are identical, although the U.S. has primary (health-based) and secondary (welfare-based) standards for $PM_{2.5}$.

*Table 4-1. National ambient air quality standards for $PM_{2.5}$*

| **Averaging Time** | **CAAQS Numerical Value** | | | **US NAAQS Level** | **CAAQS and NAAQS Metric**[25,26,27] |
|---|---|---|---|---|---|
| | **2015** | **2020** | **2025** | | |
| 24-hour (calendar day) | 28 µg/m³ | 27 µg/m³ | (under review) | 35 µg/m³ (primary and secondary) | The 3-year average of the annual 98th percentile of the daily 24-hour average concentrations. |
| Annual (calendar year) | 10.0 µg/m³ | 8.8 µg/m³ | (under review) | 12.0 µg/m³ (primary)[28]<br>15.0 µg/m³ (secondary) | The 3-year average of the annual average of the daily 24-hour average concentrations. |

## 4.2 Emission Trends in Primary $PM_{2.5}$ and $PM_{2.5}$ Precursors

Ambient concentrations of $PM_{2.5}$ are affected by both direct emissions of primary $PM_{2.5}$ and emissions of precursors, which can lead to the formation of secondary $PM_{2.5}$. In many locations within the U.S. and Canada, secondary PM accounts for the majority of $PM_{2.5}$ mass, much of which is derived at least in part from anthropogenic precursors. In Canada, the long-range transport of ambient $NH_3$ contributes to increased $PM_{2.5}$ concentrations, particularly in regions where local $NH_3$ emissions are low, such as southern Ontario and southern Quebec (Environment Canada, 2009). The sections below show trends in direct emissions of primary $PM_{2.5}$ and $NH_3$ as a $PM_{2.5}$ precursor. Emissions of other $PM_{2.5}$ precursors —

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[25] See footnote 14

[26] See footnote 15

[27] See footnote 16

[28] The U.S EPA has announced a proposed revision to the primary (health-based) annual $PM_{2.5}$ standard. At the time of this report, this revision has not had a final decision. More information on the PM NAAQS rule can be found at: https://www.epa.gov/pm-pollution/national-ambient-air-quality-standards-naaqs-pm

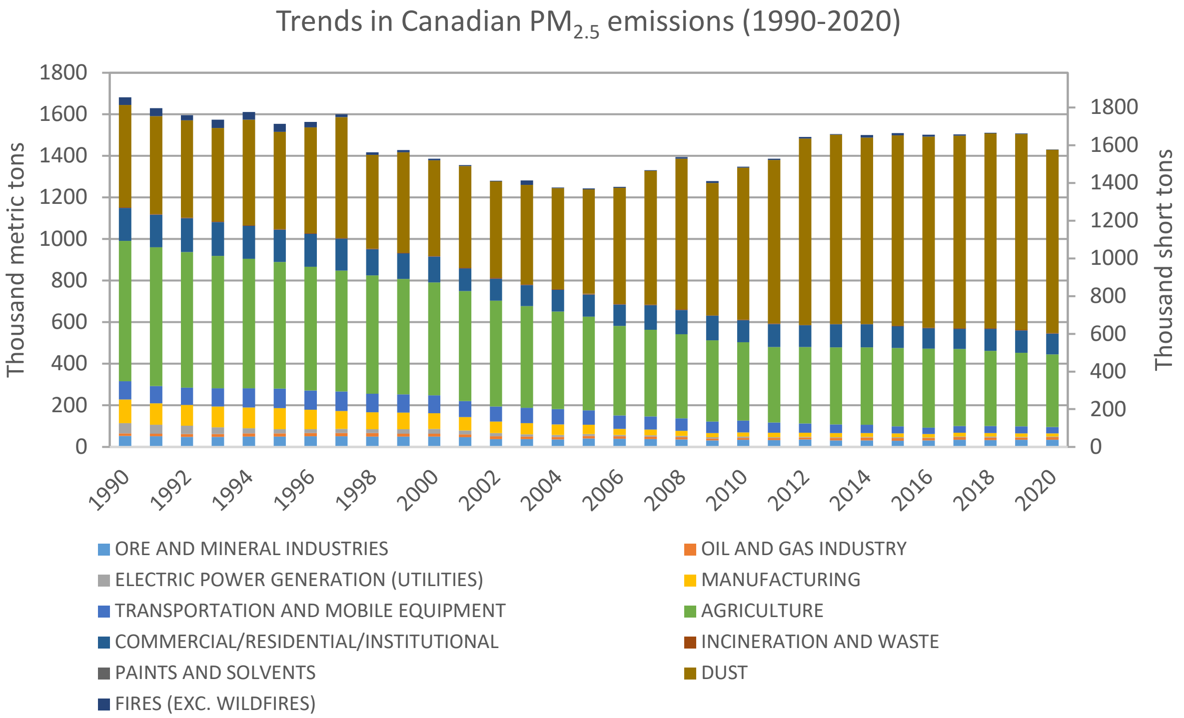
$NO_X$, $SO_2$, and VOCs – (see Section 2.1 and Section 3.1) have all decreased significantly in both Canada and the U.S. Though $PM_{2.5}$ is not covered under the AQA, the U.S. and Canada continue to take domestic actions to reduce $PM_{2.5}$ and its precursors.

### 4.2.1 Canada

Figure 4-1 shows emissions of primary $PM_{2.5}$ in Canada from anthropogenic sources (excluding wildfires). In 2020, approximately 1.4 million metric tons of primary $PM_{2.5}$ were emitted in Canada (ECCC, 2022). Dust sources accounted for 62% of total $PM_{2.5}$ emissions, with the most dominant dust sources being construction operations (35% of total $PM_{2.5}$ emissions) and unpaved roads (25% of total $PM_{2.5}$ emissions). Agriculture was the next largest contributor and accounted for 24% of total $PM_{2.5}$ emissions, most of which were attributed to crop production.

Commercial/residential/institutional sources accounted for an additional 7% of total $PM_{2.5}$ emissions in 2020, with home firewood burning (6%) being the most important contributor. From 1990 to 2020, Canada's national emissions of primary $PM_{2.5}$ decreased 15% (250 thousand metric tons) (ECCC, 2022).

*Figure 4-1. Canadian $PM_{2.5}$ Emission Trends. 1990-2020*

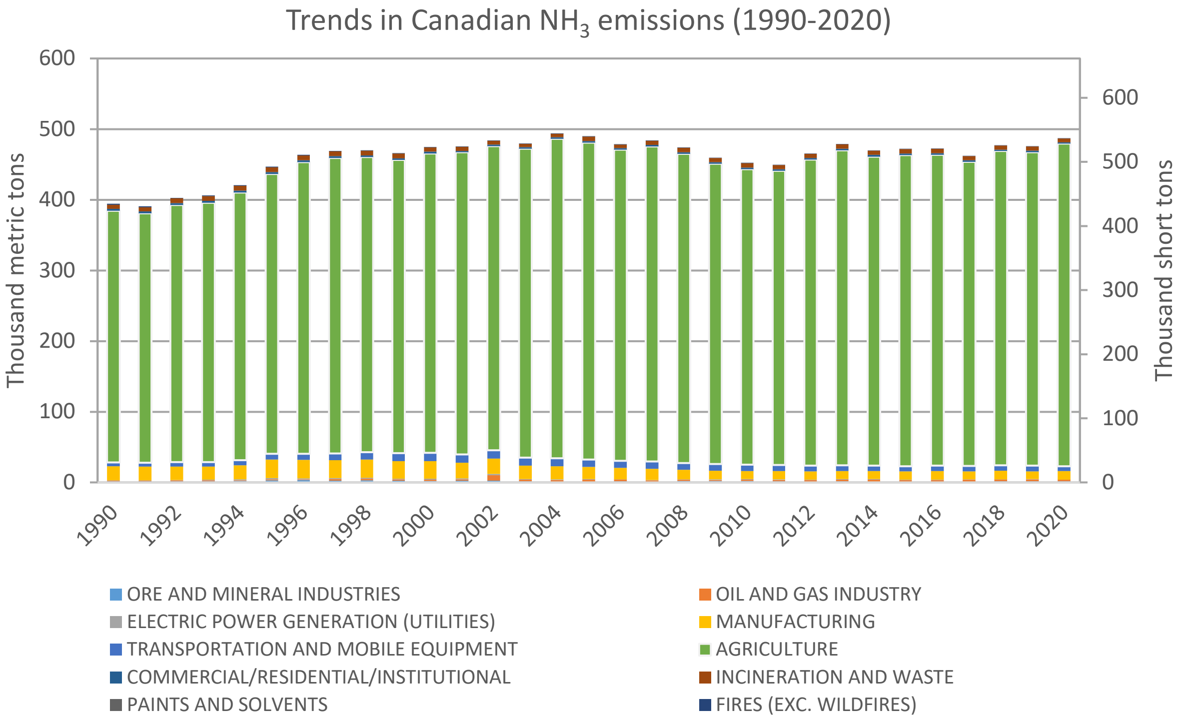


Data source: Canada's Air Pollutant Emissions Inventory 1990-2020 (ECCC, 2022)

As noted above, emissions from wildfires are not included in Canada's APEI, with the exception of prescribed burning which is included in the "Fire (Excluding Wildfires)" source category. Wildfires are, however, a significant source of primary $PM_{2.5}$ in Canada. Using wildfire seasons from 2013 to 2016 as a reference, the total $PM_{2.5}$ emitted from wildfires ranges from 790,000 to 1,700,000 metric tons (Munoz-Alpizar et al., 2017). Therefore, wildfire emissions are on the same order of magnitude as total primary $PM_{2.5}$ emissions from anthropogenic sources (~1,200,000 to 1,700,000 metric tons).

Emissions of $NH_3$ in Canada from 1990 to 2020 are shown in Figure 4-2. In 2020, approximately 487 thousand metric tons of $NH_3$ were released. $NH_3$ emissions originated primarily from agriculture, which accounted for 94% (457 thousand metric tons) of total emissions. From 1990 to 2020, $NH_3$ emissions increased by 24% (93 thousand metric tons), in contrast to the general downward trends of other air pollutant emissions (*e.g.*, $NO_X$, $SO_2$, and VOCs) in Canada. $NH_3$ emissions increased until 2004 and the largely plateaued with some year-to-year fluctuations. Thie increasing trend between 1990 and 2004 was driven by emissions from animal production and the increasing use of inorganic nitrogen fertilizers in crop production. Animal production, which accounts for the majority of $NH_3$ emissions, steadily increased from 1990 to 2005, followed by a decrease from 2006 to 2011, and has since declined slowly. Emissions from crop production, however, have been steadily increasing since 2006, and now account for 38% of $NH_3$ emissions.

*Figure 4-2. Canadian $NH_3$ Emission Trends. 1990-2020*



Data source: Canada's Air Pollutant Emissions Inventory 1990-2020 *(ECCC, 2022)*

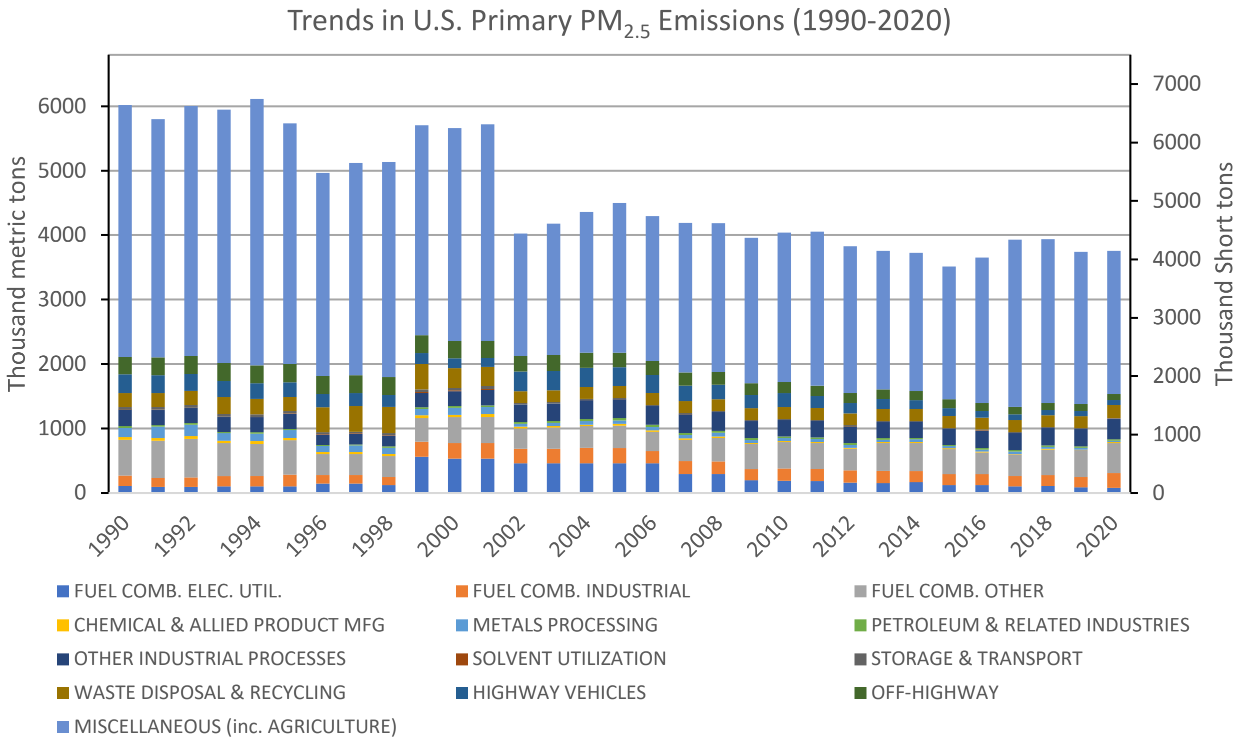
### 4.2.2 United States

Figure 4-3 shows the emissions trend for U.S. primary $PM_{2.5}$. Approximately 4.1 million short tons of primary $PM_{2.5}$ were emitted in 2020, mostly from miscellaneous sources and other non-industrial combustion sources, such as residential wood burning and residential cooking. Miscellaneous sources include agricultural crop and livestock dust (17% of total $PM_{2.5}$), road dust (18% of total $PM_{2.5}$) and agricultural and prescribed fires (20% of total $PM_{2.5}$).

Since 1990, national emissions of total primary $PM_{2.5}$ decreased by 38%. The same regional and multi-state programs that decreased ozone concentrations also reduced some $PM_{2.5}$ precursors such as $NO_X$. Acid Rain programs reduced $SO_2$, and other precursors of $PM_{2.5}$, thereby decreasing $PM_{2.5}$ concentrations. Similarly, transportation sector programs reduced precursors to $PM_{2.5}$.

On an urban scale, sources that emit $PM_{2.5}$ vary from city to city. Generally, urban primary $PM_{2.5}$ emissions include industrial activities, motor vehicles, fuel combustion, wood smoke, construction, and road dust. Both urban primary sources and regional secondary $PM_{2.5}$ generation contribute substantially to $PM_{2.5}$ mass in urban locations. Source contributions to primary $PM_{2.5}$ emissions have changed over time. For example, changes in both gasoline and diesel emissions controls have led to reductions in primary $PM_{2.5}$ emitted from newer vehicles, and primary emissions from stationary fuel combustion, industrial activities, and non-road vehicles have also decreased. Methods for quantifying $PM_{2.5}$ emissions were updated starting in 2002. These changes had the effect of doing two things from 2002 onward: 1) providing a consistent basis for methods across as many sectors as possible, and 2) elucidating trends at a higher sectoral level, so that categories like “miscellaneous” which previously could not be looked at in detail, can now be examined for the sources contributing to the overall changes in the “miscellaneous” categories. The updates in 2002 onward include providing trends at an Emissions Inventory System sector level (60 sectors) in addition to providing trends at a “Tier 1” level (about 13 sectors). Noting that due to the ECCC emissions inventory not including wildfire data, the U.S. EPA removed wildfire data using [EPA’s Air QUAlity TimE Series Project (EQUATES)]() from 2002 to present. To ascertain and remove pre-2002 wildfire emissions data, the U.S. EPA used a correlation that derived $PM_{2.5}$ and $NH_3$ emissions from available 1990-2001 carbon monoxide emissions, so was able to extend the series back to 1990 for $PM_{2.5}$ and $NH_3$.

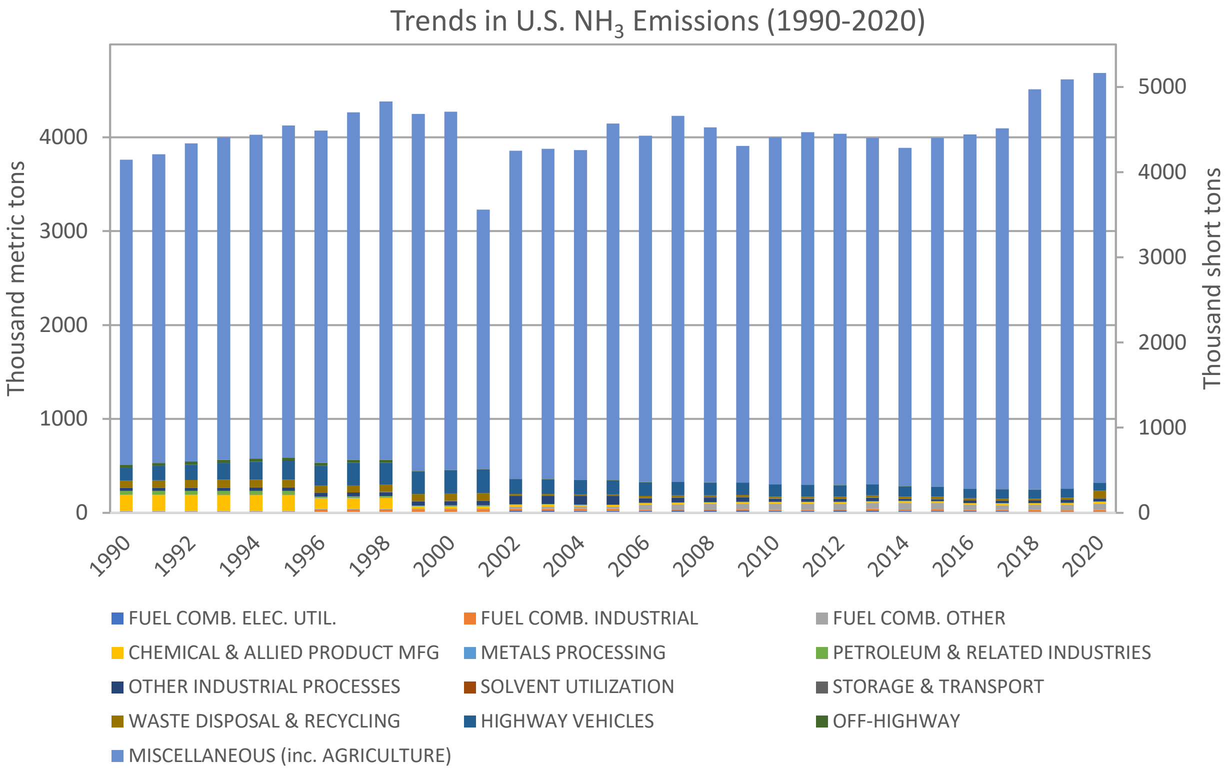
*Figure 4-3. U.S. $PM_{2.5}$ emission trends from 1990-2020*



Data source: 2020 U.S. National Emissions Inventory (US EPA, 2023a).

Figure 4-4 shows the trend in U.S. $NH_3$ emissions. Note that emissions from the agricultural sector are in the miscellaneous category. Similar to Canada, $NH_3$ emissions in the U.S. have increased, slowly but consistently, resulting in a 25% increase from 1990 to 2020. While many industrial and transport sectors showed a decrease in $NH_3$ emissions, these sectors combined account for less than 10% of overall $NH_3$ emissions.

*Figure 4-4. U.S. $NH_3$ emission trends from 1990-2020*



Data source: 2020 U.S. National Emissions Inventory (US EPA, 2023a).

Methods for quantifying $NH_3$ were also updated starting in 2002 in addition to a shift in wildfire classification. $NH_3$ trends have also likely increased over time due to improvements to the emissions trends methodology (migrating to EQUATES methods from the year 2002 on) (K. M. Foley et al., 2023) and from more robust accounting of the emissions from the agricultural livestock waste sector. Due to noted improvements in trends methods, these changes over time better represent emissions-based changes and minimize any impact from shifts in methods.

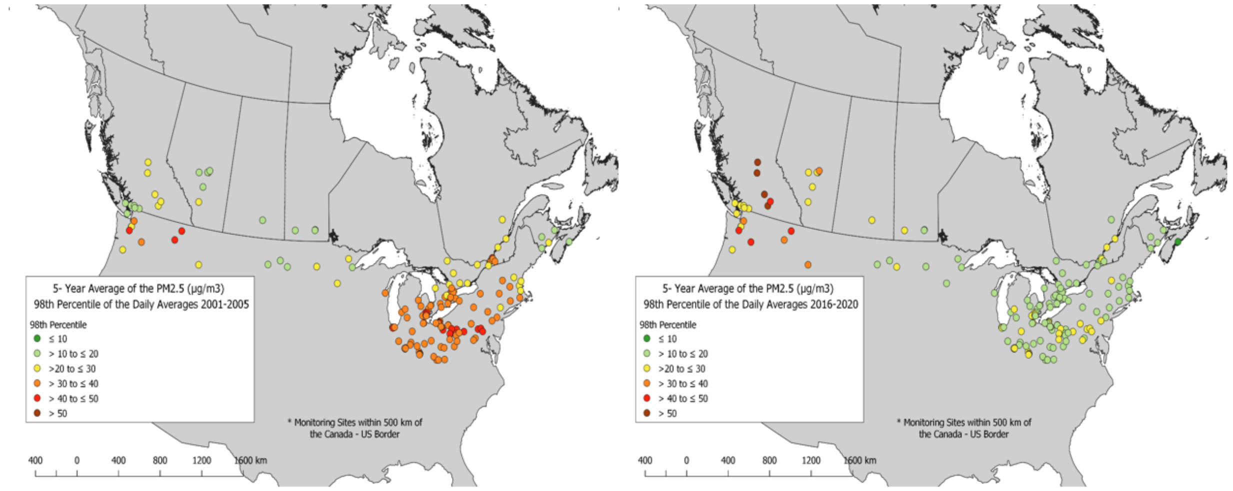
## 4.3 Ambient Concentrations of $PM_{2.5}$

Ambient $PM_{2.5}$ concentrations in Canada and the U.S. were analyzed to assess historical trends since continuous $PM_{2.5}$ monitoring was widely deployed in 2001, and to evaluate current concentrations. Monitoring data were included for individual stations located within 500 km of the Canada-U.S. border for two time periods, 2001-2005 and 2016-2020. Data are presented using statistical forms or metrics that are similar to both the CAAQS and U.S. NAAQS, but extend over a longer averaging period in order to average over any variability introduced due to factors such as wildfires. The values were calculated based on the "as measured" concentrations without adjusting for influences of transboundary flow and episodic high $PM_{2.5}$ events, such as wildfires, high wind dust, and other activities.

Figure 4-5 shows the five-year average of the annual 98th percentile of the daily 24-hour average $PM_{2.5}$, referred to as the "24-hour $PM_{2.5}$". Comparing the later time period to 2001-2005, nearly all stations east of the Manitoba-Ontario border recorded lower 24-hour $PM_{2.5}$, by more than 10 micrograms per cubic meter (µg/$m^3$) for many stations across Ontario, Quebec and the Maritimes and most U.S. stations from Lake Michigan eastward. Some U.S. stations in this area recorded decreases of nearly 20 µg/$m^3$. This reflects reductions of high concentrations that were the target of air quality regulations. For 2016-2020, several stations in Quebec demonstrated no improvements, with measured 24-hour $PM_{2.5}$ concentrations exceeding 20 µg/$m^3$.

West of the Manitoba-Ontario border, increasing wildfire activity across western North America had a large impact on $PM_{2.5}$ concentrations. Many stations recorded higher 24-hour $PM_{2.5}$ concentrations, with increases greater than 20 µg/$m^3$ from 2001-2005 to 2016-2020 at some locales. Higher $PM_{2.5}$ concentrations occurred in British Columbia and the U.S. Northwest, with 24-hour concentrations exceeding 50 µg/$m^3$ for several stations inland from the coast. Wildfires across Western North America between 2016 and 2020 likely had a significant influence on the increased $PM_{2.5}$ concentrations (Jaffe et al., 2020). Additionally, over the last few years many monitoring agencies, including in both Canada and the U.S., have migrated to continuous PM methods that are more sensitive to measuring smoke as PM.

*Figure 4-5. $PM_{2.5}$ concentrations (98th percentile of the daily averages) along the Canada-U.S. border for (a) 2001-2005 and (b) 2016-2020.*



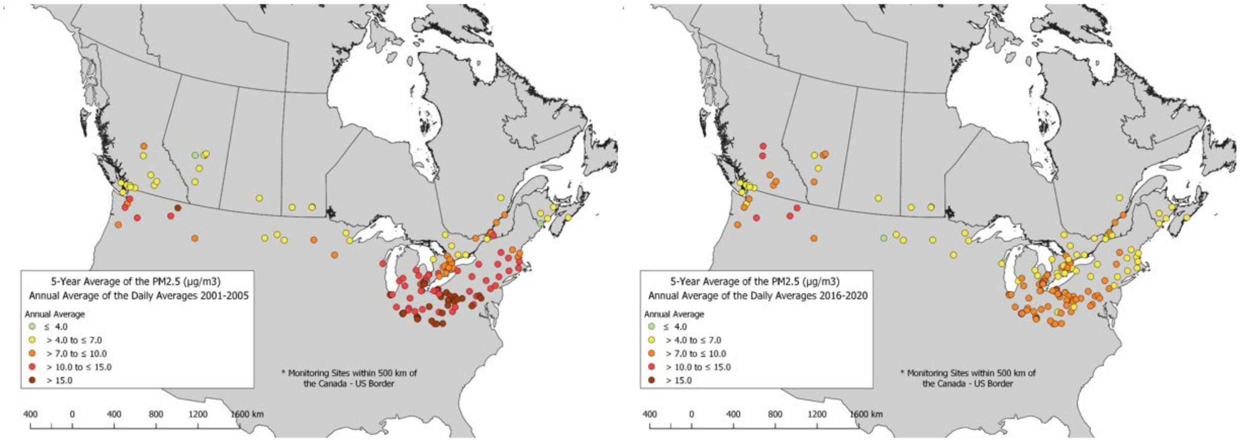
a. 2001-2005

b. 2016-2020

Note: Data are the averages of annual 98th-percentile of the daily 24-hour average $PM_{2.5}$, for (a) 2001-2005 and (b) 2016-2020. Only sites that met data completeness requirements were used to develop this map. For data completeness, required 75% or more of all daily average values available in the year, and at least 60% of daily average values available in each quarter. 98th percentile values that did not meet these criteria, but that exceeded the 2020 CAAQS were also included.

Figure 4-6 shows the annual average of daily 24-hour average $PM_{2.5}$ concentrations, averaged over 5 years. This is referred to as the “annual $PM_{2.5}$”. Spatial patterns and temporal trends are broadly comparable to the 24-hour metric. East of the Manitoba-Ontario border, annual $PM_{2.5}$ concentrations have decreased from 2001-2005 to 2016-2020 at many stations. Despite these decreases, concentrations were still elevated at several stations – with the highest annual $PM_{2.5}$ concentrations recorded at Duncan & Decarie-Montreal and Park Primevere, and Windsor, Ontario, where observed concentrations approached or exceeded the CAAQS. Numerous stations west of the Manitoba-Ontario border recorded higher average annual $PM_{2.5}$ during the 2016-2020 period. This is consistent with increased influence from wildfire smoke (Schichtel et al., 2017). In the U.S., annual $PM_{2.5}$ concentrations decreased for almost all stations east of Lake Michigan. In the West, however, high concentrations of annual $PM_{2.5}$ persist likely due to the influence of wildfire smoke.

*Figure 4-6. $PM_{2.5}$ concentrations (Annual average of the daily averages) along the Canada-U.S. border for (a) 2001-2005 and (b) 2016-2020.*



a. 2001-2005 b. 2016-2020

Note: Data are the averages of annual averages of the daily 24-hour average $PM_{2.5}$, for (a) 2001-2005 and (b) 2016-2020. Only sites that met data completeness requirements were used to develop this map. For data completeness, required 75 % or more of all daily average values available in the year, and at least 60% of daily average values available in each quarter.

## 4.4 Projected Changes in $PM_{2.5}$

Modeling of current and projected emissions scenarios conducted by ECCC and U.S. EPA were used to estimate how $PM_{2.5}$ concentrations might change in the future in both countries. The modeling is described in Appendix A. The ECCC modeling was performed using the GEM-MACH model and is presented for a base year 2015, as well as for two projected BAU 2025 and 2035 scenarios. The U.S. EPA modeling was performed for a 2016 base year using two methods considered in the 2020 Ozone PA and in support of implementing the Regional Haze Rule (RHR) (US EPA, 2019b)[29]. The U.S. EPA modeling uses the CMAQ in the PA modeling and CAMx for the RHR modeling. The RHR modeling was also performed for a future 2028 projection. The projected years for both the Canada and U.S. modeling include regulations that will be in force in the future. As the ECCC emissions inventory does not include wildfires, the ECCC and U.S. EPA modeling inventory summaries in this section do not include wildfire emissions (U.S. EPA modeling does however include wildfires).

The ECCC and U.S. EPA modeling results estimated annual average $PM_{2.5}$ concentrations. Similar to the ozone modeling results presented above (Section 3.4), the projections use a single meteorology, but emissions are varied. So, the projected changes in the future are due entirely to changes in emissions of precursors, and the subsequent transportation, transformation, and fate of chemical constituents.

Figure 4-7 presents annual average $PM_{2.5}$ concentrations from the ECCC modeling efforts for the 2015 base year, and for future years in 2025 and 2035. Peaks in $PM_{2.5}$ concentrations are localized, with higher concentrations simulated near urban centers, as well as some regionally elevated concentrations.

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[29] Simulations with newer inventory versions and years are available for $PM_{2.5}$, which include data fusion with observations (*e.g.*, (US EPA, 2019a, 2022a)). These newer simulations were not used here because they did not include particle source apportionment technology necessary to assess transboundary issues.

These results are expected as there are more primary $PM_{2.5}$ emissions in cities and industrial areas. Because many of these emissions are at ground-level, they are unlikely to be transported as far as sources emitted aloft. Secondary PM can be formed downwind of primary source regions by precursors with longer lifetimes, leading to $PM_{2.5}$ that spreads beyond the source regions.

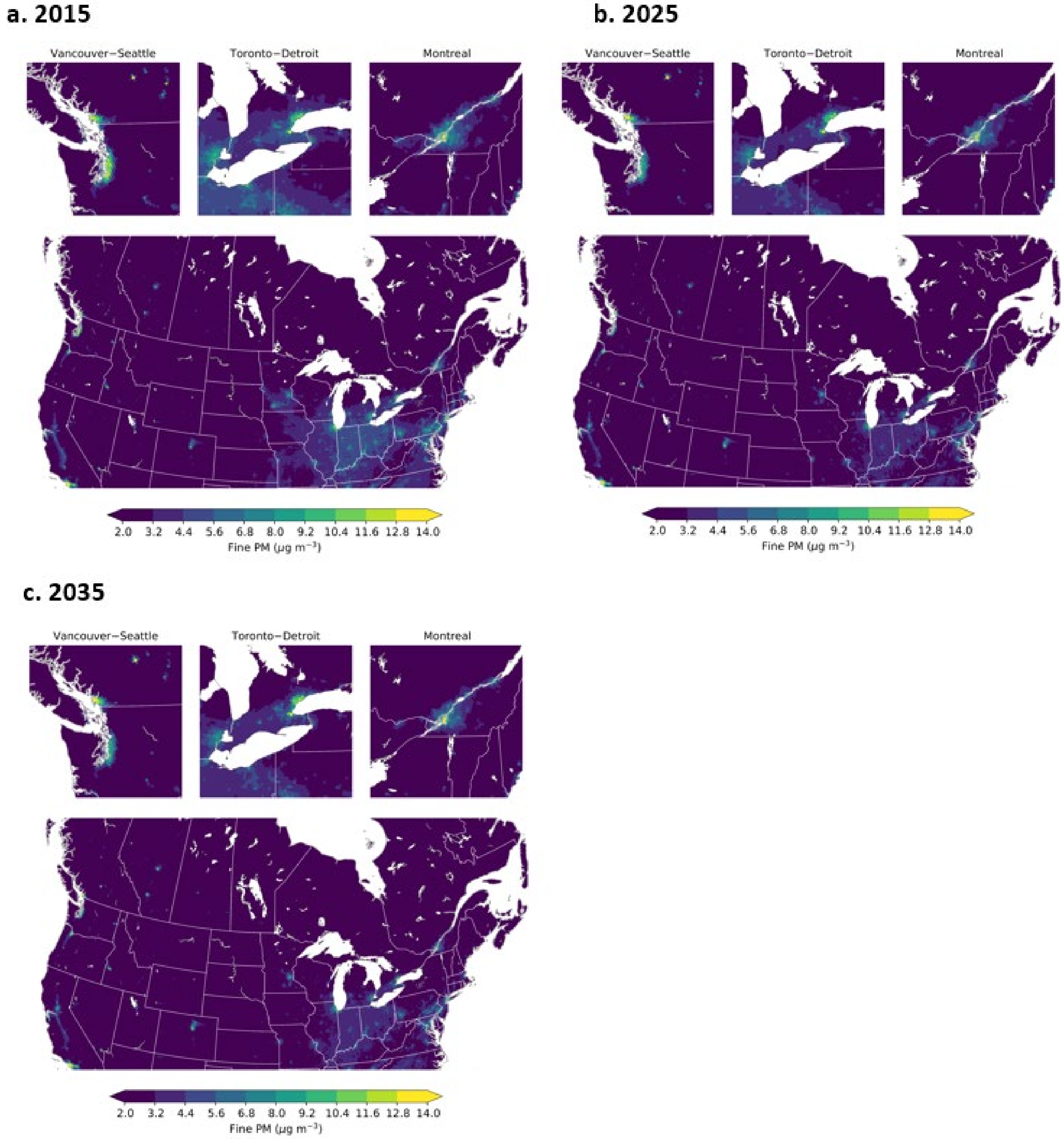
For the 2015 base year (Figure 4-7a) in Canada, the highest concentrations (greater than 10 µg/m$^3$) estimated by the ECCC modeling effort occurred in some of Canada's largest cities: the greater Toronto-Hamilton area, greater Montreal, Quebec City, Calgary, Edmonton, and Vancouver. Elevated $PM_{2.5}$ concentrations were also estimated near Kamloops – British Columbia, Grande Prairie – Alberta, and Trois-Rivières – Quebec. In the U.S., ECCC modeling estimated higher $PM_{2.5}$ concentrations near urban centers in the Ohio Valley area particularly around Cleveland and Detroit, with high $PM_{2.5}$ concentrations also found in the Midwest continental U.S. Elevated $PM_{2.5}$ concentrations in the northeastern U.S. are also estimated throughout the broader Ohio Valley area and some areas of the eastern seaboard. The modeled $PM_{2.5}$ concentrations in urban areas are broadly consistent with spatial patterns in monitoring data (Figure 3-5b). However, the modeling did not include consideration of wildfire emissions, so modeled $PM_{2.5}$ concentrations in western Canada and the U.S. are expected to be lower than the measured data.

The ECCC modeling projections predict decreases in $PM_{2.5}$ in future years (Figure 4-7b and Figure 4-7c), with elevated concentrations persisting in urban areas in Canada and the U.S. The decreases in $PM_{2.5}$ are consistent with projected decreases in emissions of precursors as new regulations come into effect. In 2035, the annual average $PM_{2.5}$ CAAQS is predicted to continue to be exceeded in cities within British Columbia, Alberta, Ontario, and Quebec.

Figure 4-8 shows the U.S. EPA modeling results for the 2016 base year and for projections to 2028. For the base year, the magnitude and spatial distribution of $PM_{2.5}$ concentrations from the PA16 (Figure 4-8a) and RHR16 (Figure 4-8b) modeling efforts are broadly consistent over most regions in Canada and the U.S. Similar to ozone (Section 3.4), the PA16 results show higher concentrations in southern Alberta because the emissions in Alberta were revised downward before the RHR. Other minor differences are likely associated with the selection of models, which use different algorithms for $PM_{2.5}$ formation and loss processes.

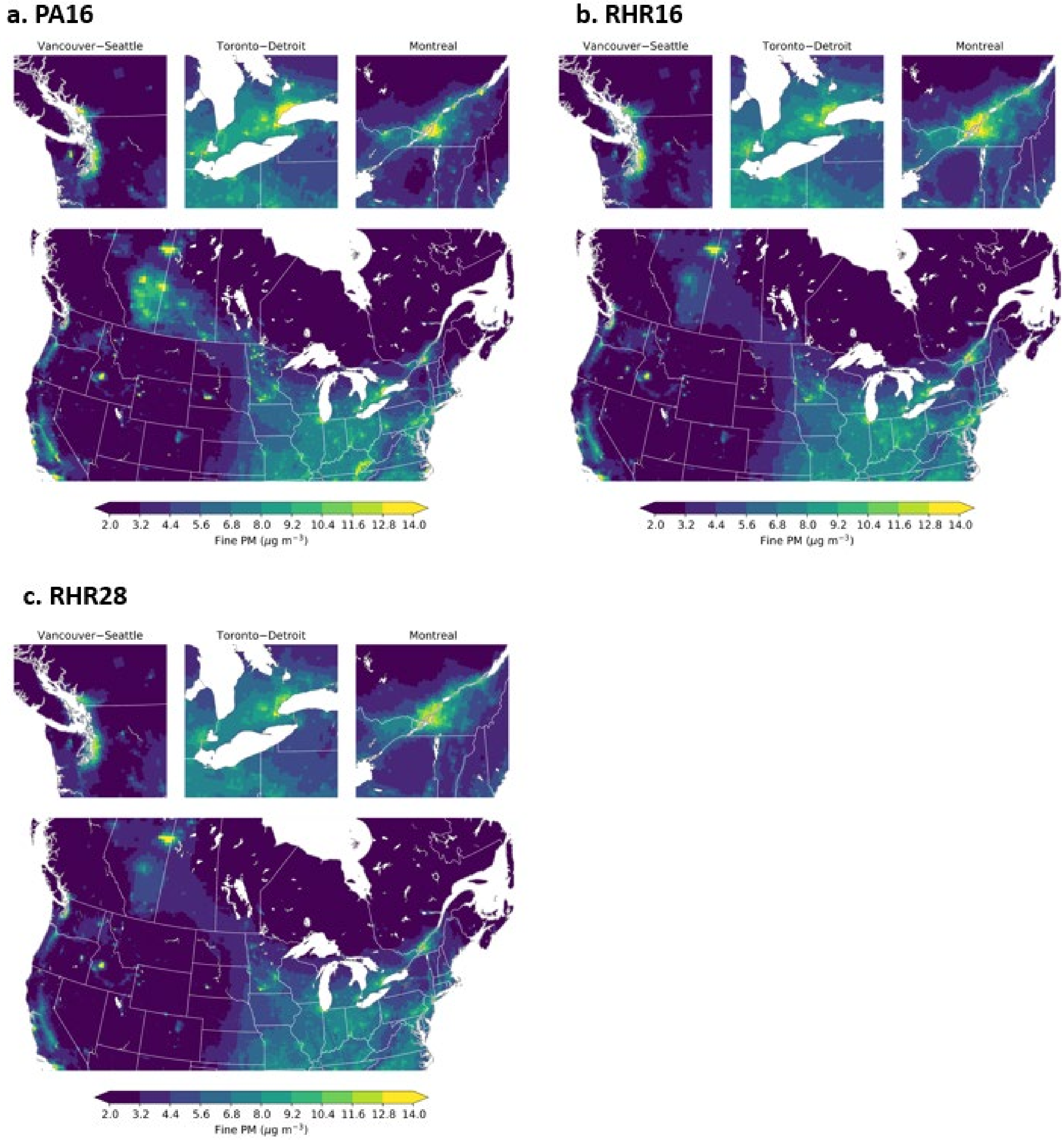
The U.S. EPA modeling (Figure 4-8) and the ECCC modeling (Figure 4-7) show broadly consistent spatial patterns and projected changes in $PM_{2.5}$ concentrations. However, a systematic difference in $PM_{2.5}$ concentrations between the U.S. EPA and ECCC annual average $PM_{2.5}$ concentrations can be seen between Figure 4-7 and Figure 4-8. These differences can likely be attributed to differences in models (*i.e.*, CAMx vs GEM-MACH), emission years, meteorological base years, and inventory development methods. Despite prediction differences, qualitative information on emissions sources and trends relevant to policy decisions can be obtained from these results and further verification and alignment could be pursued in the future.

*Figure 4-7. Annual average $PM_{2.5}$ concentrations from ECCC modeling scenarios for the base year (a) 2015, and for future projections (b) 2025 and (c) 2035.*



The 2028 RHR modeling projections (Figure 4-8c) suggest that future $PM_{2.5}$ concentrations will decrease especially near U.S. urban areas. Regional concentrations are also predicted to decrease, with the most obvious reductions in the Ohio Valley and along the Atlantic coast. The Windsor-Quebec Corridor also shows some decreases, but not generally as large as those in the U.S.

*Figure 4-8. Annual average $PM_{2.5}$ concentrations from U.S. EPA modeling scenarios for the base year (a) PA16 and (b) RHR16, and for future projections (c) RHR28.*



## 4.5 Influence of Transboundary Flow

$PM_{2.5}$ is known to have both local and regional patterns associated with primary emissions and secondary formation. The overall lifetime of $PM_{2.5}$ is often longer than other pollutants, except for ozone. For example, secondary organic PM has an estimated lifetime of 4 to 10 days (Tsigaridis et al., 2014). Fine particles can be transported across a wide range of distances dependent on various conditions, including long-range transport events of dust, $SO_4$, and wildfire smoke (e.g., *Global Sources of Local Pollution*, 2010; Mathur, 2008; Mathur et al., 2017; Uno et al., 2009). However, high $PM_{2.5}$

associated with local emissions such as during winter-time inversions tends to be more localized than regional impacts with ozone, as often happens in summer months. Qualitative evidence for general transport can be seen in typical near surface meteorological patterns such as sustained high pressure in either winter (PM inversions) or summer (regional ozone episodes).

Although the highest $PM_{2.5}$ concentrations are typically associated with local and regional emission sources, intercontinental transport associated with forest fires or dust storms produce concentrations that may exceed short-term health-based ambient air quality standards. Intercontinental transport of $PM_{2.5}$ may interfere with visibility for natural surroundings in Western North America. Intercontinental transport of $PM_{2.5}$ from sources other than wind-blown dust or wildfires are not usually sufficient to exceed health-based ambient air standards (HTAP, 2010).

Appendix B shows a meteorological analysis of winds affecting transboundary flow across the Canada-U.S. border. Winds generally follow a west-to-east flow, with more variability near the surface. Near the surface in summer, there are generally north-to-south winds between Alberta and Montana, while there is a general southwest-to-northeast flow between Detroit and Windsor. In the winter, the south-to-north flow between Alberta and Montana is typically more evident but the Detroit-Windsor flows are less well defined. Thus, the transport of $PM_{2.5}$ between countries depends on the location and season.

Modeling was used to estimate the influence of Canada and U.S. emissions on $PM_{2.5}$ concentrations using a similar approach as presented for ozone (see Section 3.5 for a description of the approach). The ECCC modeling and the U.S. EPA PA modeling approaches used a zero-out method, and the U.S. EPA RHR modeling approach used Particulate matter Source Attribution Technology (PSAT). Modeling data are presented as difference maps, where positive values indicate larger influence of emissions on a given location. Results from modeling are interpreted qualitatively here because, like ozone, $PM_{2.5}$ exhibits nonlinear behavior (Ansari & Pandis, 1998; West et al., 1999)[30].

Figure 4-9a shows the ECCC modeled annual average $PM_{2.5}$ concentration differences between the 2015 base case and the corresponding scenario without U.S. emissions, which can be used to estimate the influence of U.S. emissions. The largest influences are seen in southern Canada, with the highest influence in southern Ontario, followed by the Vancouver region and the Montreal area. Smaller influences are noted in Manitoba and part of Ontario near the Manitoba border. The influence of U.S. emissions on ambient $PM_{2.5}$ concentrations over the U.S. is significant, with the largest influence seen in the northeastern U.S. extending toward the south of Lake Superior, as well as around the Seattle area.

Figure 4-9b shows the modeled annual average of $PM_{2.5}$ concentration differences between the 2015 base year and the scenario without Canadian emissions, which allows for estimation of the influence of Canadian emissions on both Canada and U.S. Canadian emissions had a strong influence on ambient $PM_{2.5}$ concentrations in Canada, mainly over cities in southwestern British Columbia, in the southern part of the Prairies, and around the Windsor-Quebec City Corridor.

Canadian emissions also had some impact in the U.S. near the Canada-U.S. border, particularly, the Seattle area, northern Montana, North Dakota and the northeastern U.S. Comparing Figure 4-9a and

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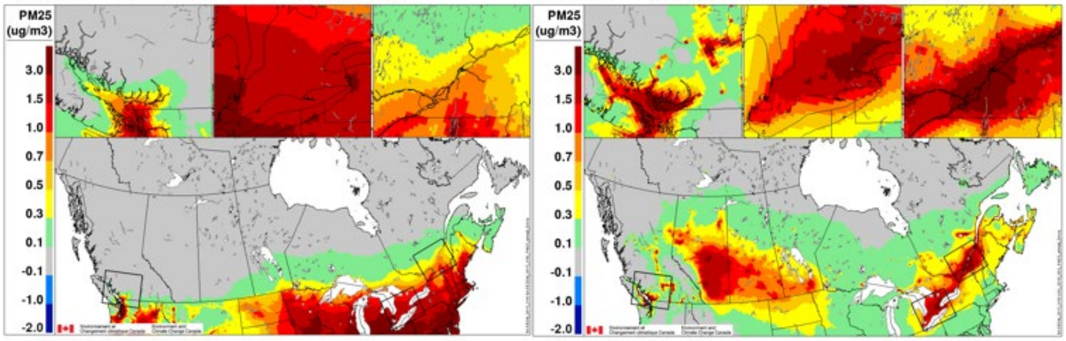
[30] For example, reduced emissions of $SO_2$ limit the formation of $SO_4^{2-}$ aerosol, and, therefore, a decrease in $PM_{2.5}$ would also be expected. However, the reduced formation of $SO_4^{2-}$ may be offset by increased formation of other PM species leading to no decrease in $PM_{2.5}$ concentrations, or even a slight increase. In particular, conditions of reduced $SO_2$ and $SO_4^{2-}$ may be favorable for the formation of more $NO_3^-$ , especially under cold wintertime conditions.

Figure 4-9b shows the relative impact of the Canadian vs the U.S. emissions on $PM_{2.5}$ for a region. For example, the Windsor area showed more impact of U.S. emissions than Canadian, most of southern Ontario shows approximately an equal impact of Canadian and U.S. emissions, and Toronto shows a greater impact of Canadian emissions than U.S. emissions.

ECCC modeling projections to 2035 are also used to estimate influences from emissions in the U.S. (Figure 4-9c) and Canada (Figure 4-9d) on $PM_{2.5}$ concentrations. Although $PM_{2.5}$ concentrations are expected to decrease between 2015 and 2035 (see Section 4.4), the spatial patterns of influence of emissions remains consistent, since the same meteorology was used for both years. Emissions from the U.S. are expected to continue to influence Canadian $PM_{2.5}$ concentrations with the largest impact in southern Ontario and the Vancouver region, and smaller impacts in the Montreal area and southern Manitoba. However, the region of greatest impact of U.S. emissions on Canada has decreased in southern Ontario and southern Quebec compared to the 2015 simulations. Similarly, the influence of Canadian emissions from Alberta on Montana and North Dakota has decreased from the 2015 runs.

*Figure 4-9. Influence of U.S. and Canadian emissions on annual average $PM_{2.5}$ concentrations from ECCC zero out modeling. Results are shown for the 2015 base case for (a) influence of U.S. emissions and (b)influence of Canadian emissions, and for the 2035 projections for (c) influence of U.S. emissions and (d) influence of Canadian emissions.*

**a. 2015 – Influence of U.S. Emissions** **b. 2015 – Influence of Canadian Emissions**



**c. 2035 – Influence of U.S. Emissions** **d. 2035 – Influence of Canadian Emissions**

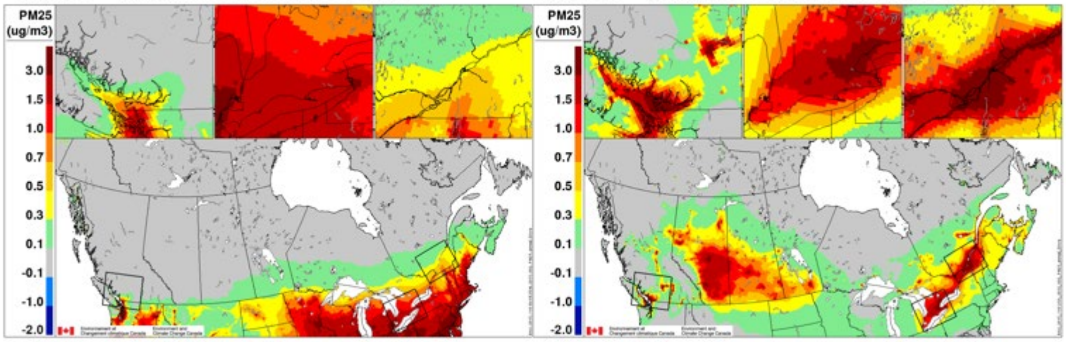
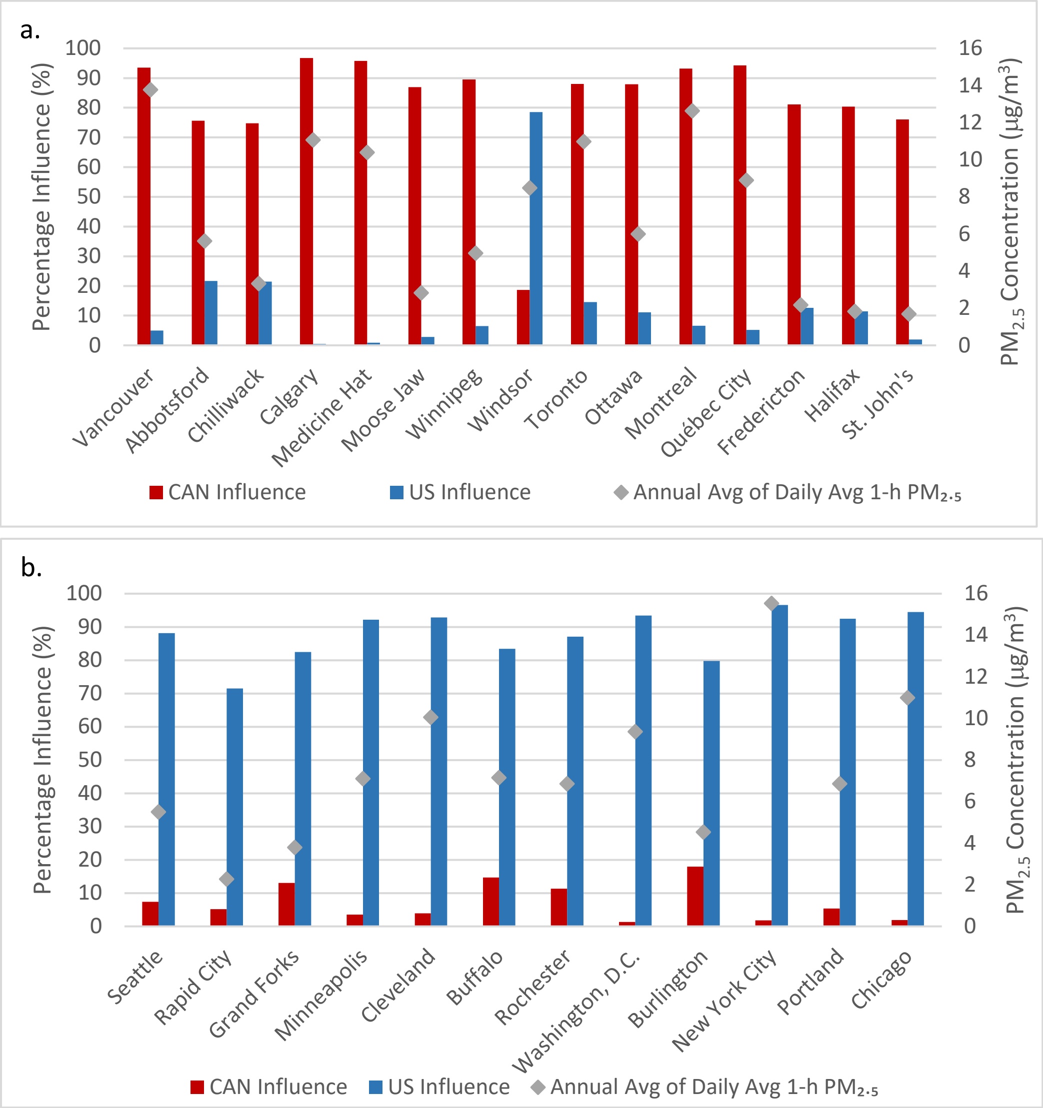


Figure 4-10 shows the influence of Canadian and U.S. emissions on the annual average $PM_{2.5}$ concentrations for major Canadian cities using the ECCC modeling estimates for the 2015 base year. Canadian cities are more affected by Canadian emissions than by U.S. emissions, except for Windsor, which is located 4 km from the large urban area of Detroit in the U.S. U.S. emissions are responsible for approximately 10-20% of ambient $PM_{2.5}$ concentrations in several Canadian cities, including Abbotsford and Chilliwack in southwestern British Columbia, and Fredericton, Halifax, Ottawa, and Toronto in Eastern Canada. Ambient annual average $PM_{2.5}$ concentrations in these cities range from approximately 2 µg/m$^3$ in Fredericton and St. John's to greater than 10 µg/m$^3$ in Toronto.

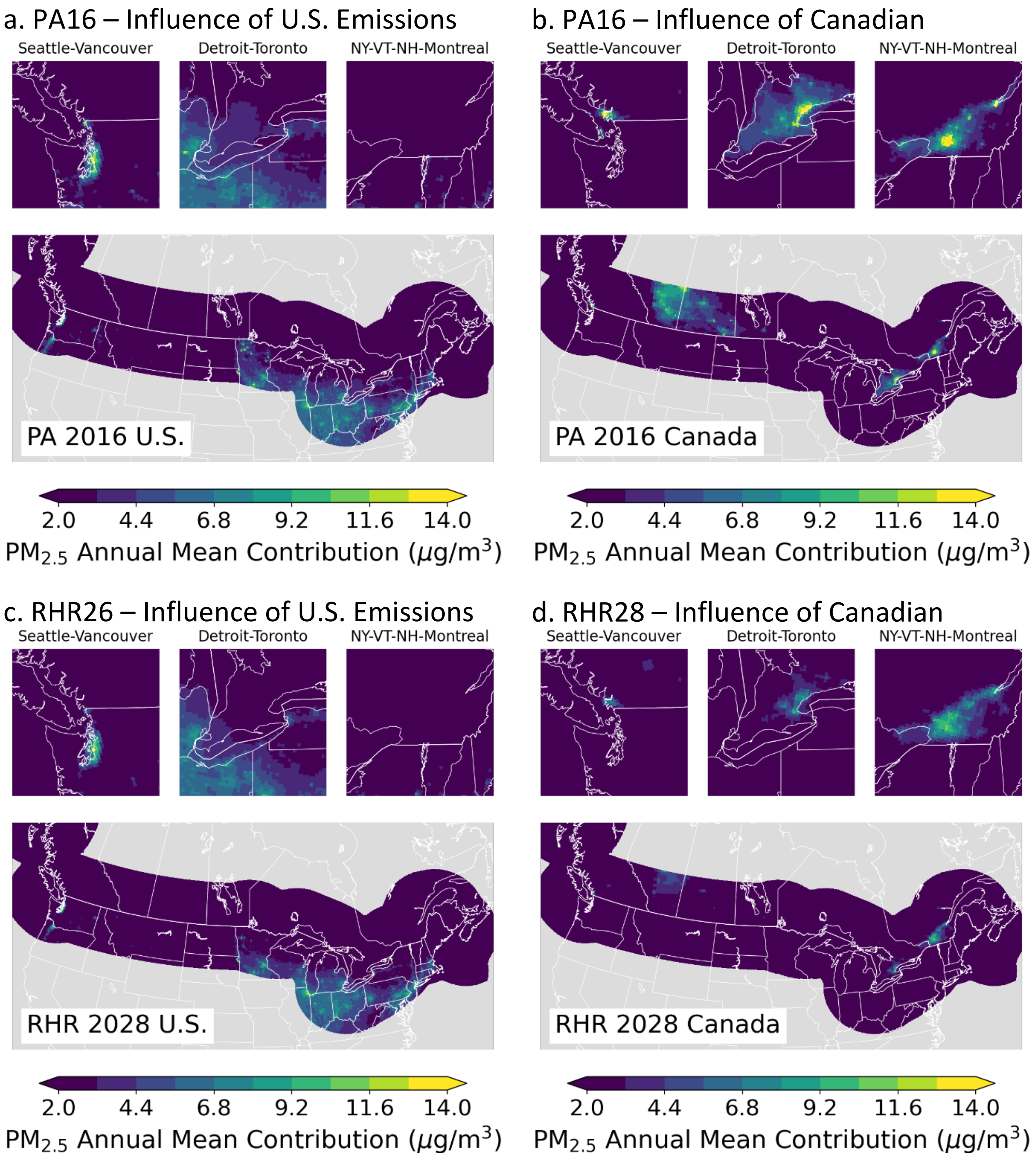
*Figure 4-10. Influence of Canadian and U.S. emissions (left y-axis) and PM$_{2.5}$ concentrations (right y-axis) for the 2015 PM$_{2.5}$ annual average concentrations in (a) Canadian and (b) U.S. cities from ECCC modeling.*



U.S. cities are more affected by U.S. PM$_{2.5}$ emissions than Canadian emissions. The influence of Canadian emissions on U.S. cities ranges from less than 2% of total ambient PM$_{2.5}$ concentrations in Washington D.C., Chicago, and New York City, up to approximately 15% of total concentrations in Buffalo and Burlington. Total overall ambient PM$_{2.5}$ concentrations in these cities range from approximately 5 µg/m$^3$ in Burlington, Vermont, to approximately 15 µg/m$^3$ in New York City.

The U.S. EPA modeling results for the U.S. and Canada contributions are shown in Figure 4-11. The contributions are qualitatively consistent with ECCC zero-out modeling (Figure 4-9). Emissions from the U.S. have a larger influence on PM$_{2.5}$ concentrations in the U.S. (Figure 4-11a), with some influence extending across the border into Canada, for example into southern Ontario. Emissions from Canada primarily influence PM$_{2.5}$ concentrations within Canada (Figure 4-11b). The future contributions from

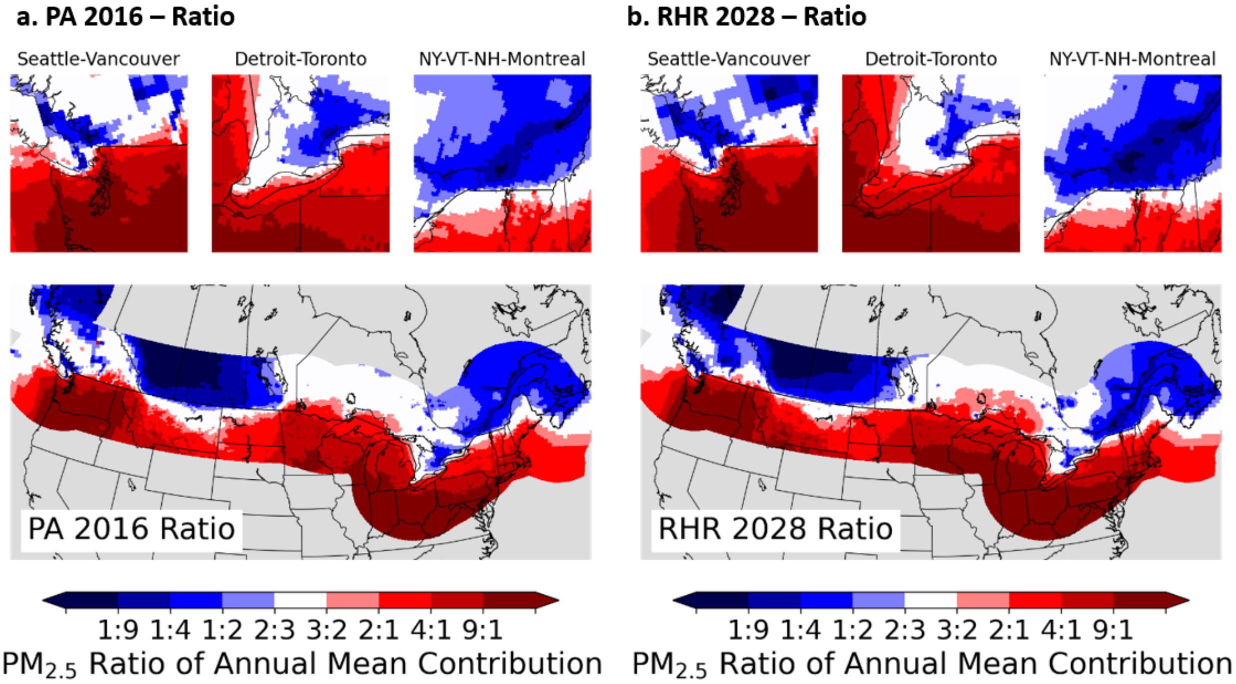
*Figure 4-11. Influence of U.S. and Canadian emissions on annual average $PM_{2.5}$ concentrations from U.S. EPA zero out modeling. (a) U.S. and (b) Canadian attribution for PA16 modeling scenario; (c) U.S. and (d) Canadian attribution for RHR28 modeling scenario.*



RHR modeling show decreases in $PM_{2.5}$ concentrations compared to the PA 2016 modeling. However, the comparison is only qualitative because the two simulations used different models and different apportionment approaches as outlined previously.

Figure 4-12 shows the relative magnitude of emissions contributions for the U.S. EPA modeling estimates. In the U.S., the contributions are substantially larger than those from Canada in most locations. This is similar to the findings for ozone (Section 3.5). In Canada, $PM_{2.5}$ concentrations are more strongly influenced by Canadian emissions compared to ozone. This is particularly true in the populated areas. Overall, the findings that concentrations are most heavily influenced by domestic emissions is consistent with the patterns of long-range transport of this pollutant, especially in comparison with ozone.

*Figure 4-12. Ratio of $PM_{2.5}$ from the U.S. and Canada from U.S. PA16 (left) and RHR28 (right) modeling scenarios.*



## 4.6 Health Impacts

Short-term (hours to days) exposure to $PM_{2.5}$ can cause serious heart and lung events like heart attacks, heart failure, stroke, and asthma attacks and premature death (Health Canada, 2022a; US EPA, 2019c). Adverse outcomes also include increased emergency room visits and hospitalizations for cardiovascular and respiratory disease. Long-term (months to years) exposure to $PM_{2.5}$ can cause premature death, and can likely cause lung cancer, and heart and lung diseases. Exposure to $PM_{2.5}$ may also lead to adverse neurological and developmental outcomes. Studies evaluated in Canadian and U.S. science assessments do not find evidence of a threshold for $PM_{2.5}$ associated health effects (Health Canada, 2022a; US EPA, 2019c). Children, older adults, smokers, people carrying certain gene variants (*e.g.*, antioxidant enzyme) and those with pre-existing cardiovascular and respiratory conditions (*e.g.*, asthma) are at greater risk (Health Canada, 2022a). In Canada, the health burden of above-background $PM_{2.5}$ in 2016 was estimated to be 10,000 premature deaths annually, with an economic cost of $80 billion per year (2016 Canadian dollars) (Health Canada, 2021).

ECCC and U.S. EPA zero-out modeling was used to estimate contributions of U.S. and Canadian sources to population-weighted annual average $PM_{2.5}$ concentrations. For population, this analysis uses the Gridded World Population dataset version 4 to define population. $PM_{2.5}$ concentration data were interpolated to the same grid as the population data to calculate the population weighted average. Contributions to population-weighted average concentrations are reported in Figure 4-13 and Figure 4-14. Each panel represents a different population: (a) all people within the 500-km buffer or a PEMA state, (b) the U.S. population within that area, and (c) the Canadian population within that area. Each source definition represents a subset of emission sources: “All” represents the total emissions sources from Canada and the U.S., “U.S.” represents only the anthropogenic sources within the U.S., and “Canada” represents only the anthropogenic sources within Canada. For the U.S. EPA modeling, the “Canada” label includes emissions from both Canada and Mexico. However, Mexican contributions are small near the Canadian border.

Figure 4-13 shows population-weighted average $PM_{2.5}$ concentrations from the ECCC modeling runs. Total population-weighted $PM_{2.5}$ concentrations from “All” sources for 2015 are similar for U.S. (Figure 4-13b) and Canadian (Figure 4-13c) portions of the buffer region. Within the U.S. (Figure 4-13b), most contributions are from U.S. emissions, and these contributions are projected to decrease between 2015 and 2025. Emissions from Canada have a small influence on total concentrations in the U.S. This is consistent with the relatively localized nature of $PM_{2.5}$. Within Canada (Figure 4-13c), most contributions are from Canadian emissions, which are projected to remain relatively stable over the years modeled, with a slight projected increase for the 2035 timeframe, largely due to increases within Canada. Within Canada, of the total modeled population-weighted concentration of approximately 6 μg/m$^3$, there is a non-negligible contribution from U.S. emissions of approximately 15%.

*Figure 4-13. ECCC modeled population weighted annual mean $PM_{2.5}$ contributions (μg/m$^3$) from sources (All, Canada, U.S.) to populations within 500-km or PEMA states, the U.S. portion, or the Canadian portion.*

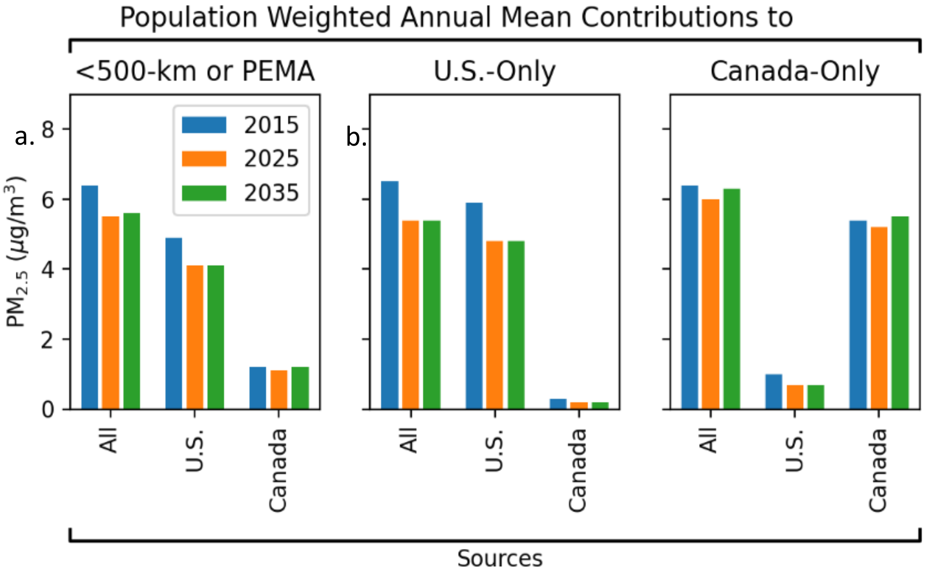
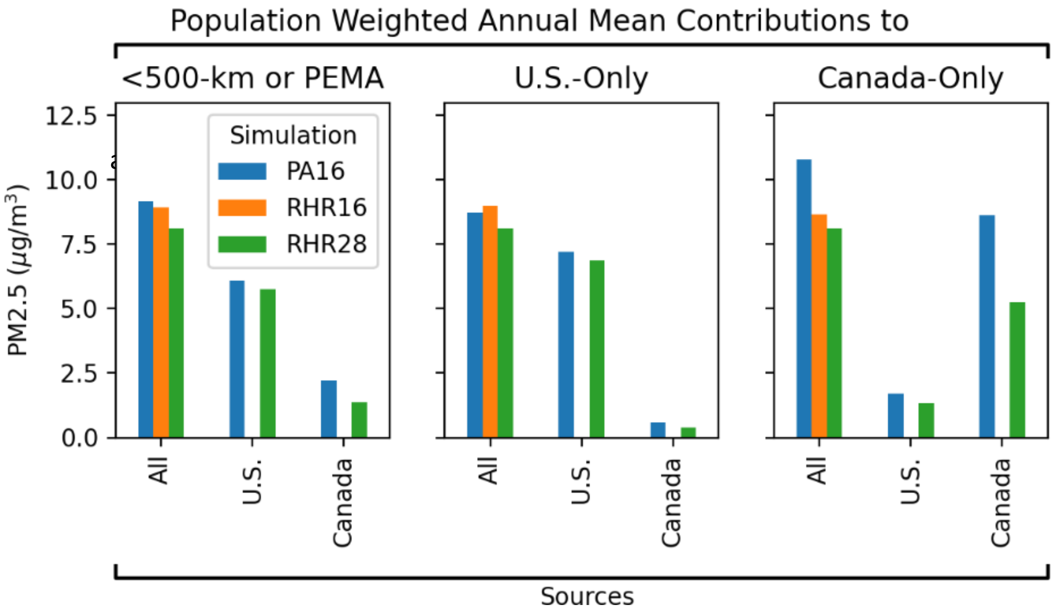


Figure 4-14 shows the population-weighted averages for the U.S. EPA modeling runs. The U.S. EPA modeling is qualitatively consistent with ECCC modeling, but $PM_{2.5}$ concentrations are larger for U.S. EPA modeling than the ECCC modeling, as discussed in Section 4.3. Within the U.S. (Figure 4-14b), total 2016 $PM_{2.5}$ decreases by ~10% by 2028. The Canadian contribution is very small. The U.S. contribution is larger (~75%) and accounts for the simulated decreases. Within Canada (Figure 4-14c), total present-day

$PM_{2.5}$ is similar to concentrations in the U.S. In this case, the RHR model is favored because this modeling reflects the revised emissions in Alberta. On both sides of the border, the domestic sources contribute over 80% of the exposure metric.

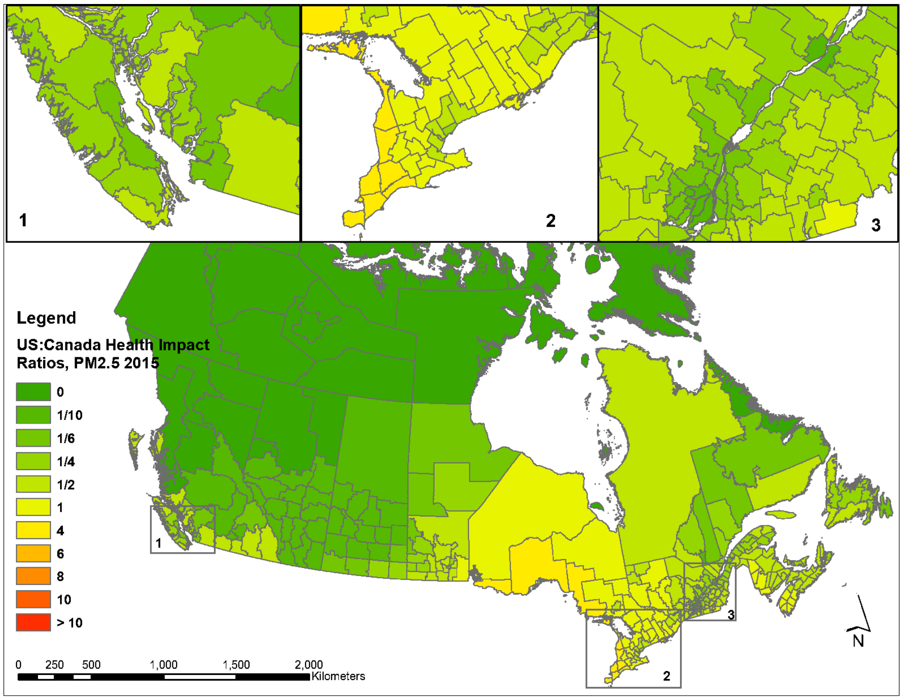
*Figure 4-14. U.S. EPA modeled population weighted annual mean $PM_{2.5}$ contributions (µg/m$^3$) from sources (All, Canada, U.S.) to populations within 500-km or PEMA states, the U.S. portion, or the Canadian portion.*



To attribute $PM_{2.5}$ health impacts in Canada to U.S. and Canadian sources, Health Canada applied the Air Quality Benefits Assessment Tool (AQBAT v3) (Judek et al., 2019) to the ECCC zero-out modeling runs for 2015, as in Pappin et al. (2024). Since the modeling runs should be interpreted qualitatively as described in Section 4.5, the health impact analysis should also be interpreted accordingly. Figure 4-15 maps the estimated ratios of source contributions from the U.S. to those from Canada to total $PM_{2.5}$ health impacts by census division. Areas in yellow suggest that the contribution of U.S. sources to local health impacts exceeds that of Canadian sources, while areas in green suggest a greater contribution from Canadian sources. Health impacts resulting from the transboundary flow of $PM_{2.5}$ and its precursors from the U.S. to Canada are largest near the Canada-U.S. border and in the Central and Atlantic Canadian regions. A key finding is that despite smaller ratios of U.S. to Canadian source contributions for $PM_{2.5}$ than for ozone, the aggregate health impacts (*e.g.*, total premature deaths) of transboundary $PM_{2.5}$ in Canada exceed those of transboundary ozone due to the considerably higher risks associated with each unit of exposure to $PM_{2.5}$ (Pappin et al., 2024). U.S. contributions to $PM_{2.5}$ health impacts in the populous census divisions of Toronto, Montreal, and Vancouver are smaller than the contributions from Canadian sources, yet they account for roughly one-quarter of the transboundary $PM_{2.5}$ health burden in Canada due to the size of these populations. Transboundary $PM_{2.5}$ health impacts mostly occur in Ontario and Quebec and within 200 km of the Canada-U.S. border. Health impacts in Canada attributable to transboundary $PM_{2.5}$ are projected to decline from 2015 to 2025, and increase from 2025 to 2035 due in part to projected changes in $PM_{2.5}$ concentrations, an increasing number of Canadians susceptible to adverse health effects as a result of ageing, and population growth due to higher immigration (Pappin et al., 2024).

These analyses demonstrate that while transport from the U.S. to Canada is less pronounced for $PM_{2.5}$ than for ozone, $PM_{2.5}$ from U.S. emissions sources has an impact on health outcomes in Canada. The areas where transported pollutants are the most significant contributors (fractionally) tend to be less populated. Areas of highest population density have the largest fraction of local (*i.e.*, not transported) contributions. However, contributions from transboundary flow have a significant impact on health along the border region, including within Canada's largest urban centers (Toronto, Montreal, and Vancouver).

*Figure 4-15. Ratio of contributions from U.S. sources to those from Canadian sources to total $PM_{2.5}$-related health impacts in Canada (estimated as an economic value per year), by census division. Ratios include health impacts based on annual average $PM_{2.5}$ concentrations. Ratios > 1 indicate that U.S. sources contribute more to local health impacts than Canadian sources. Ratios < 1 indicate that Canadian sources contribute more than U.S. sources. Ratios are based on zeroing-out in GEM-MACH for 2015. (Adapted from Pappin et al., 2024)*



## 4.7 Environmental Impacts

Ecosystems are affected by the deposition and sedimentation of $PM_{2.5}$ and mobilization of $PM_{2.5}$ components, including $NH_4^+$, $NO_3^-$, and $SO_4^{2-}$, into the environment. $PM_{2.5}$ can deposit on vegetation, such as plants, by wind, precipitation, and through direct contact with $PM_{2.5}$-containing water droplets in clouds, fog, and mist. $PM_{2.5}$ deposition can directly affect photosynthesis, respiration, transpiration, and other normal plant functions. Indirect effects of $PM_{2.5}$ deposition include alteration of soil composition and uptake of harmful compounds by plants resulting from greater exposure time to $PM_{2.5}$ and its chemical components (Grantz et al., 2003; US EPA, 2004).

Impacts to vegetation from $PM_{2.5}$ can lead to nutrient imbalances, detrimentally impacting some species. Particulate nitrogen and sulfur deposition from both dry deposition and particle scavenging can also indirectly contribute to ecosystem acidification. Particle scavenging by precipitation was the source of approximately 70% of $NH_4^+$, 60% of $SO_4^{-2}$, and 30% of $NO_3^-$ in wet deposition at several long-term monitoring stations in Canada (Cheng & Zhang, 2017). These effects contribute to economic losses through impacts like lowered crop yields (Chameides et al., 1999) and lower radial growth rates in important forest species like pine and eastern hemlock (Farahat et al., 2016; Mandre & Korsjukov, 2007). Additional impacts may include material degradation (e.g., accelerated stone decay, corrosion of transmission lines, etc.) and aesthetic amenity losses.

One of the readily noticeable environmental impacts of $PM_{2.5}$ is the impairment of “visibility”. Suspended $PM_{2.5}$ can block and scatter sunlight, thereby impairing visibility. Visibility impairment is related to $PM_{2.5}$ concentration, composition, and relative humidity and is typically variable in location and time. While the most prominent effects of visibility reduction include the loss of scenic vistas, severe visibility impairment can pose a hazard for aviation, as well as marine, highway, and rail transportation.

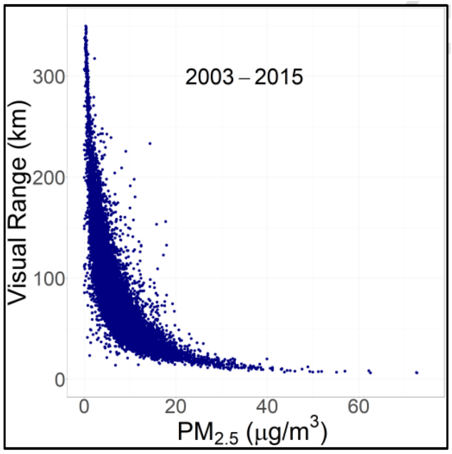
### 4.7.1 Canadian Actions to Protect Visibility Under the AQA

Canada continues to address the AQA commitment to prevent air quality deterioration and ensure visibility protection by implementing the *Canadian Environmental Protection Act, 1999* and the *Impact Assessment Act, 2019* and by following the principles of “continuous improvement” and “keeping clean areas clean”. These principles underpin Canada’s AQMS and the associated CAAQS. ECCC has also developed a statistical model to estimate light extinction from routine air quality measurements and has analyzed the visibility impact of emission reduction scenarios. This modeling work has guided policy decisions to improve visibility.

In the absence of broader scale dedicated visibility monitoring, routine PM speciation monitoring data can be used to reconstruct visibility conditions. Visibility calculations have been completed using data from all speciation stations operating for greater than two years in Canada from the years 2003-2015 (AMC, 2021b; Dabek-Zlotorzynska et al., 2011). As shown in Figure 4-16 relatively modest levels of $PM_{2.5}$ can result in drastic reduction in visual range.

In general, rural stations have slightly better visibility than urban stations. Kananaskis, Alberta has nearly pristine visibility with the highest average visual range (197 km). Rural stations in southern Ontario and Quebec have comparable average visual range to urban stations in western Canada. Urban stations in Atlantic Canada have a better visual range than those in western Canada. Windsor and Toronto, Ontario have the lowest visibility with average visual ranges of 64 km and 69 km, respectively. The species contributing to visibility impairment (extinction) varies regionally across the country, with $SO_4^{2-}$ species being more predominant in eastern Canada and ammonium nitrate and organic matter more prevalent in the west. Rural stations tend to have less contribution from elemental carbon and $NO_2$, and more contribution from $SO_4^{2-}$ species compared to urban stations. On average, across all stations during the span between 2003 and 2015, $PM_{2.5}$ contributed to 68% of total extinction, while $NO_2$, coarse mass and Rayleigh scattering make up the remainder.

*Figure 4-16. The relationship between daily $PM_{2.5}$ concentration and visual range reconstructed from particle speciation data stations across Canada for the years 2003-2015.*



Data source: Visibility was calculated using a modified version of the second revised IMPROVE (Interagency Monitoring and Protective Visual Environment) (Pitchford et al., 2007) using aerosol speciation data publicly available from the National Air Pollution Surveillance Program https://open.canada.ca/data/en/dataset/1b36a356-defd-4813-acea-47bc3abd859b

### 4.7.2 U.S. Actions to Protect Visibility Under the AQA

The Regional Haze Program has a goal of reaching natural conditions at Class I areas, which include National Parks and Wildlife refuges that can be far from emission sources. This is particularly true in the West and Northwest U.S. As result of the remote nature of Class I areas in the West and Northwest, the $PM_{2.5}$ concentrations are generally low, as are the visibility extinctions. Because Class I areas are often not near U.S. sources, the contributions from both the U.S. and Canadian anthropogenic sources may be considered "transported." In the ozone and $PM_{2.5}$ sections of this report, the meteorological transport shown in Appendix B highlight the transport patterns from Canada to the U.S. in the Montana Class I areas that are affected by transboundary flows.

Montana has several Class I areas that may have large transport from Canada, including UL Bend (Montana)[31]. Figure 4-17 shows that the unadjusted goal of natural conditions (labeled "Unadj 2064" in the figure) is 18 inverse megameters ($Mm^{-1}$), which is 13 $Mm^{-1}$ less than the current conditions (31 $Mm^{-1}$). Thus, emissions reductions may be required to reach the unadjusted goal. Figure 4-17 also indicates that the U.S. anthropogenic contributions are smaller than the international anthropogenic contribution in the 2028 modeling. The data indicate that eliminating the U.S. contribution would be likely insufficient to reach even the 2028 glidepath value (26 $Mm^{-1}$). Given that the concentrations changed very little from 2016 to 2018, we can infer that the present-day situation is comparable. Given the location and known transport issues, it is not surprising that 9.81 $Mm^{-1}$ (out of 14.52 $Mm^{-1}$) of the international category are from Canada. The large impact of international sources and the $NO_3^-$ model performance suggest a need to collaboratively improve the understanding and control of these

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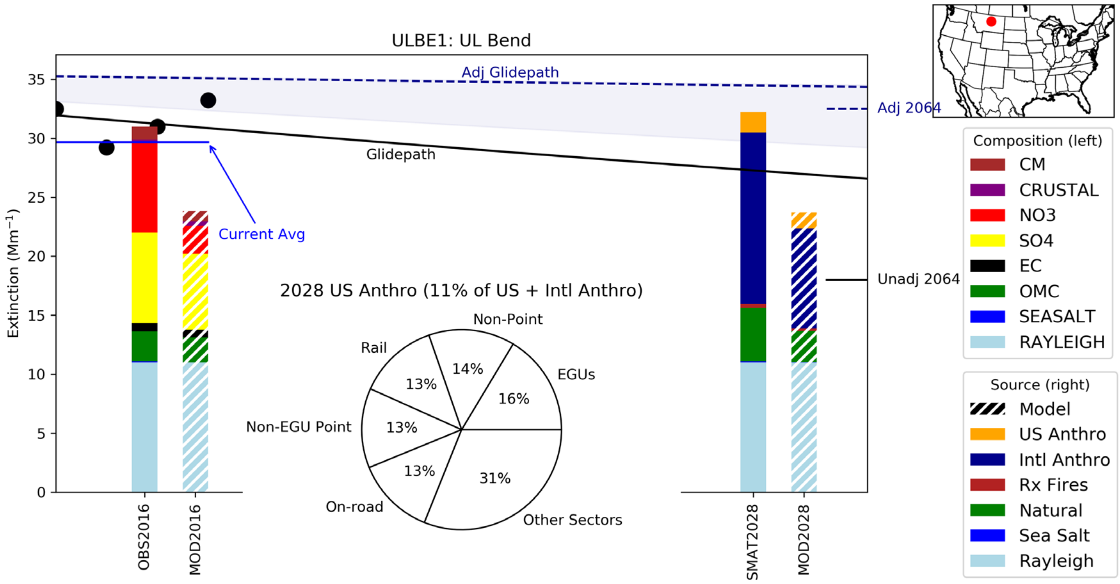
[31] Further information on the Montana 2nd planning period SIP: https://deq.mt.gov/files/Air/AirQuality/Documents/RegionalHaze/State%20of%20Montana%20Regional%20Haze%20SIP.pdf

emissions to address regional haze. The other Class I areas that appear to have a high Canadian contribution are Lostwood (North Dakota) and Medicine Lake (Montana).

*Figure 4-17. Chemical and source composition of regional haze in the UL Bend Park in Montana. Observations (OBS2016) and raw model predictions for 2016 (MOD2016) are shown as bar plots where the stacked components are chemical. Raw model (MOD2028) and projected observations (SMAT2028) are shown as stacked bar plots where the components represent Natural ((Rayleigh (natural light scattered from air molecules is referred to as Rayleigh scattering and causes the blue appearance of the sky) Sea Salt, and Natural), International, and U.S. contributions. The pie chart shows the U.S. sector-specific contributions. The labeled lines represent the "glidepath" and "adjusted glidepath" that are part of the Regional Haze Rule analysis.*



## 4.8 Summary

Ambient concentrations of $PM_{2.5}$ are affected by both direct emissions of primary $PM_{2.5}$ and emissions of precursors (*e.g.*, $NO_X$, $SO_2$, VOCs, $NH_3$), which can lead to the formation of secondary $PM_{2.5}$. From 1990 to 2020, Canada's emissions of primary $PM_{2.5}$ decreased by 15% (ECCC, 2022), having plateaued at approximately 1.5 million metric tons per year. U.S. national emissions of primary $PM_{2.5}$ decreased by 38% between 1990 and 2020, having gradually decreased until 2015, and then plateaued in recent years. Estimated primary $PM_{2.5}$ emissions from wildfires were not included in this analysis for either country. The regional and multi-state programs that led to decreased ozone concentrations also reduced emissions of several chemical precursors to secondary $PM_{2.5}$ ($NO_X$, $SO_2$, and VOCs). This includes acid rain programs and transportation sector programs undertaken since the signing of the AQA. While emissions of $NO_X$, $SO_2$, and VOCs have all decreased significantly in both Canada and the U.S., emissions of $NH_3$ (another $PM_{2.5}$ precursor) have increased by 24% in Canada and by 25% in the U.S. from 1990 to 2020.

Consistent with the decreases in emissions of primary $PM_{2.5}$ and its precursors, $PM_{2.5}$ concentrations have decreased from 2001-2005 to 2016-2020 at many stations east of Lake Michigan in both Canada and the U.S. $PM_{2.5}$ concentrations are largest near urban areas and particularly in the Ohio Valley, Atlantic coast, and the Windsor-Quebec Corridor, with observed concentrations for several stations in

Canada approaching or exceeding the CAAQS. West of Lake Michigan, higher concentrations of annual $PM_{2.5}$ persist in measured concentrations, likely due to the influence of wildfire smoke. Modeling, which does not include projected wildfire emissions, indicates that $PM_{2.5}$ concentrations are projected to decrease from 2015 to 2025 (ECCC) and 2016 to 2028 (U.S. EPA). However, in 2035 some of the largest cities in Canada are projected to continue experiencing $PM_{2.5}$ concentrations that exceed the 2020 CAAQS.

In Canada and the U.S., local contributions are strongest near population centers. The meteorological analysis presented in Appendix B suggests that the impact of $PM_{2.5}$ is more localized than gas-phase pollutants such as ozone. However, due to prevailing winds and large emission sources, the U.S. is a source of $PM_{2.5}$ for some regions of Canada including southern Ontario and Quebec. For $PM_{2.5}$, transport is a smaller contributor of anthropogenic concentrations relative to local emission sources on either side of the border. However, there may be significant contributions from transport within specific areas that straddle the U.S.-Canadian border, such as the Canadian Windsor / U.S. Detroit area.

Despite management actions to reduce $PM_{2.5}$ and precursor emissions, Health Canada has concluded that ambient $PM_{2.5}$ contributes to significant health impacts in Canada (Health Canada, 2021). Although transboundary $PM_{2.5}$ is a relatively minor component of $PM_{2.5}$ concentrations in nearly all areas along both sides of the U.S.-Canada border, the modeled aggregate health impacts of transboundary $PM_{2.5}$ in Canada exceed those of transboundary ozone due to higher risks associated with each unit of exposure to $PM_{2.5}$. Emission reductions in the U.S. have reduced and continue to reduce exposures. The largest effect of these reductions is on the U.S. side of the border. However, both countries continue to deal with significant health impacts from $PM_{2.5}$. Transboundary impacts, particularly in the Detroit-Windsor area persist and result in significant Canadian air quality and health impacts. Transported emissions – including transboundary emissions from Canada to the U.S. – also contribute to visibility impairment in Class I areas in the northwestern U.S.

Sources of $PM_{2.5}$ are diverse and relative contributions from different sectors can vary greatly by location. Optimal mitigation strategies may vary accordingly. $PM_{2.5}$ is a pollutant that continues to be of significant concern for its impacts on human health and the environment in both countries.

# 5 Scientific and Technical Cooperation

Annex 2 of the AQA outlines additional commitments by the Parties to collaborate on scientific and technical activities and economic research. These commitments guide efforts to improve our understanding of transboundary air pollution and its impacts as well as improve capabilities to control such pollutants in accordance with commitments in Annexes 1 and 3 and in support of other shared goals. Table 5.1 highlights the key provisions of Annex 2. Progress to meet the requirements of Annex 2 and examples of related collaborative projects are summarized below. The collaborative projects discussed in this section are not an exhaustive list of all U.S. – Canada work but rather are intended to illustrate specific examples of where strengthening scientific connections has improved each country’s abilities to understand and address challenging transboundary air quality issues.

*Table 5-1. Summary of Annex 2 – Scientific and Technical Activities and Economic Research*

| Joint Commitments | Description |
|---|---|
| **1. Determine and report on air pollutant concentrations and deposition:** | The Parties agree to coordinate air pollution monitoring through:<br>• Coordination of existing networks;<br>• Additions to monitoring tasks of existing networks of those air pollutants Parties agree should be monitored;<br>• Addition of stations or networks where no existing monitoring facility can perform a necessary function for the purposes of the Agreement;<br>• The use of compatible data management procedures, formats, and methods, and;<br>• The exchange of monitoring data, modeling, and comparison of methods |
| **2. Determine and report air emissions levels, historical trends, and projections** | The Parties agree to coordinate activities through:<br>• Identification of air emissions information that should be exchanged for purposes of the Agreement;<br>• Use of measurement and estimation procedures of comparable effectiveness and data management formats and methods, and;<br>• Exchange of air emissions information. |
| **3. Cooperate and exchange information** | The Parties agree to share Information related to:<br>• Monitoring the effects of changes in air pollutant concentrations and deposition with respect to changes in various effects categories;<br>• Determination of any effects of atmospheric pollution on human health and ecosystems;<br>• Development and refinement of atmospheric models for purposes of determining source receptor relationships and transboundary transport and deposition of air pollutants;<br>• Development and demonstration of technologies and measures for controlling emissions of air pollutants, in particular acidic deposition precursors, subject to their respective laws, regulations and policies;<br>• Analysis of market-based mechanisms, including emissions trading;<br>• Any other scientific activities or economic research the Parties may agree upon. |
| **4. Consult on approaches to, and share information and results of research** | The Parties agree to consult on:<br>• Methods to mitigate the impacts of acidic deposition, including environmental effects<br>• Economic aspects of methods to mitigate the impacts of acidic deposition |

## 5.1 Data Sharing

In the late 1970s, parallel efforts to monitor wet deposition in the U.S. and Canada set the stage for the 1986 Joint Report of the Special Envoy on Acid Rain and eventual signing of the AQA in 1991. Following the AQA's signing, in 1994, the Parties agreed to notify each other of specific sources of pollution within 100 miles of the U.S.-Canada border. This agreement persists to this day as the Parties continue to notify one another about any emissions sources expected to emit greater than 90 metric tons per year of $SO_2$,

$NO_X$, carbon monoxide, total suspended particulates (TSP), or VOCs as well as of any modifications to existing power plants expected to increase emissions of any of these pollutants by 40 metric tons per year. Notification lists can be found here for U.S. sources (US EPA, 2023f) and here for Canadian sources (ECCC, 2023).

As outlined in Section 1.4 of this report, the Parties continue to coordinate air quality monitoring data from a variety of data sources for joint and domestic purposes. Various departments of the U.S. and Canadian governments share data and are established partners in support of AirNow, a centralized hub for real-time air quality information (AirNow, 2023). Also, through joint participation in the NADP and its associated networks, the parties carry out co-located measurements at Canadian and U.S. measurement sites. Outside these formal efforts, the Parties maintain ongoing informal dialogue across a range of topics related to monitoring networks and measurement methods. For example, ECCC often attends EPA's National Ambient Air Monitoring Conference (NAAMC), most recently held in August 2022. EPA and ECCC air monitoring experts share information on an ad-hoc basis on $PM_{2.5}$, ozone, and other areas air pollutants, and ECCC has often invited EPA representatives to join Canadian ambient air monitoring workshops.

Additionally, the Parties continue to engage in pursuit of enhanced monitoring methods such as satellite-based measurements. The parties collaborate as well with the National Aeronautics and Space Administration (NASA) on the Pandora project to develop new trace gas retrieval algorithms, data products, and new techniques for satellite validation, as well as the Tropospheric Emissions: Monitoring of Pollution (TEMPO) Mission[32] (Naeger et al., 2021), to be deployed in 2023 – "the first space-based instrument to monitor major air pollutants across the North American continent every daylight hour at high spatial resolution" (NASA, 2023).

The parties continue to update and improve their emission inventories and projections for several important air pollutants, including $PM_{10}$, $PM_{2.5,}$ VOCs, $NO_X$, and $SO_2$, to reflect the latest information available. As referenced throughout this report, Canada's emissions inventory data are for the year 2020, as published in Canada's 2022 APEI (ECCC, 2022). The U.S. emissions data are based on national and state-level trend information from the 2020 National Emission Inventory (US EPA, 2023c), available in Spring 2023. Making the emissions inventories publicly available contributes to the success of both nations' emission reduction goals and air quality management programs. Emission inventories identify air pollutant sources, track progress on strategies to control emissions, and provide important data for use in air quality models. In addition to the biennial Progress Report under the AQA, Canada and the U.S. report emission concentrations through several agreements and councils such as: the Arctic Council and the Convention on Long Range Transboundary Air Pollution (LRTAP).

Measurement and emissions data support air quality modeling tools used to evaluate the impact of changing emissions and meteorology on air quality to inform scientifically grounded policy decisions. U.S. and Canadian emissions data inform modeling efforts like CMAQ (https://www.epa.gov/cmaq), utilized in support of the AQA[33] as well as other outside efforts ranging from studies of global scale to local impact. One recent prominent effort is EQUATES (US EPA, 2023d) that includes a set of modeled meteorology, emissions, air quality, and pollutant deposition for the years 2002-2019 for the

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[32] https://tempo.si.edu/overview.html

[33] For example, the U.S. EPA PA modeling runs, using the CMAQ model, are presented throughout this Review and Assessment to evaluate acid deposition, ozone, and $PM_{2.5}$.

conterminous U.S. and the Northern Hemisphere using the Weather Research and Forecasting (WRF) model and CMAQ. This time series streamlines and increases the accessibility and applicability of modeling data for use by a wide variety of government as well as public stakeholders. EQUATES uses ECCC's 2002-2017 APEI and EQUATES emission are being used in ECCC's Regional Air Quality Deterministic Reforecast System (RAQDRS). These efforts benefit both countries collectively and individually. For example, deposition data from Canada's Borden Forest has played a pivotal role in evaluating and improving U.S. and Canadian regional air quality models (Wu et al., 2018), and shared deposition data and modeling through the Air Quality Model Evaluation International Initiative Phase 4 (AQMEII4) are used to improve both countries' chemical transport models (Clifton et al., 2023). The Parties will continue to seek out similar opportunities to bolster our capacity through collaboration.

## 5.2 Collaborative Projects

Collaborative air quality monitoring efforts and shared emissions data set the foundation for all joint efforts under the AQA. Utilizing air quality, emissions, and/or modeling data, the Parties have undertaken various joint projects since the signing of the AQA to address specific shared environmental concerns under the agreement. These projects have covered a range of areas, including co-location of deposition monitoring methods (Feng et al., 2021; Sirois et al., 2000; Wetherbee et al., 2010), modeling comparisons (Kaplan et al., 1995; Li et al., 1994; McNaughton & Vet, 1996; Schwede et al., 2011), a North American Emissions Control Area (US EPA, 2010), cooperation on ozone and PM under the North American Research Strategy for Tropospheric Ozone (Hales, 2003; McMurry et al., 2004; Russell & Dennis, 2000; Schere, 1996; Schere & Hidy, 2000) and collaboration on satellite and ground-based remote sensing measurements (Shephard et al., 2011; Szykman et al., 2019; White et al., 2023). These collaborations provide foundational knowledge on air quality issues, leverage many of the shared resources outlined in Annex 2, and strengthen partnerships in pursuit of domestic and shared international goals.

Beginning in 1997, both countries undertook a review of transboundary ozone transport, and published the report, Ground Level Ozone: Occurrence and Transport in Eastern North America (AQC, 1999). The joint workplan, a project of collaborative data analysis and emissions modeling, put forward in this report culminated in the addition of the Ozone Annex to the AQA in 2000.

Following this addition, the Parties pursued various other ventures to bolster information sharing and leverage resources. The Parties published scientific assessments concerning the transboundary transport of PM between the two countries for 2004 (US EPA & Environment Canada, 2004) and again for 2013 (ECCC & US EPA, 2016). In 2005, the Parties completed multiple pilot projects under the Canada-U.S. Border Air Quality Strategy including:

- The Great Lakes Basin Airshed Management Framework (US EPA & Environment Canada, 2005a),
- Maintaining Air Quality in a Transboundary Air Basin: Georgia Basin – Puget Sound (US EPA & Environment Canada, 2005b), and
- A Study on the Feasibility of Emissions Cap and Trading for $NO_X$ (Nitrogen Oxides) and $SO_2$ (Sulfur Dioxide) (US EPA & Environment Canada, 2005c).

The conclusions of these reports, particularly those identifying region-specific challenges and solutions, provide a foundation of knowledge as the Parties assess the future direction of the AQA.

Building on these successful collaborative ventures, in 2007 the Canada-U.S. AQC approved a work plan identifying general areas of collaboration between ECCC and the EPA on vehicle and engine emission control (AQA Transportation Workplan). This Workplan outlined several key areas of focus, including: the development of national air pollutant and greenhouse gas emission standards for vehicles and engines; the development of national fuel standards; the administration of vehicle, engine and fuel compliance programs; and, the development of international standards for a variety of international fora such as the United National Economic Commission for Europe (UNECE) World Forum for Harmonization of Vehicle Regulations (WP.29), the International Maritime Organization (IMO), and the International Civil Aviation Organization (ICAO), among others.

Strategic priority projects initiated by the Parties in 2021 and 2022 in support of the AQA Transportation Workplan include, but are not limited to:

- Supporting the development of more stringent light and medium-duty greenhouse gas standards
- Research and analysis in support of President Biden's and Prime Minister Trudeau's Roadmap for a Renewed U.S.-Canada Partnership (The White House, 2021) and the Canada-U.S. High Level Dialogue on Climate Ambition (State Department, 2021)
- The development and implementation of enhanced testing approaches and procedures for monitoring compliance with vehicle and engine emission standards, such as vehicle monitoring (OBD Data Logging), real world emission equipment, remote sensing
- Continuing to lead the Electric Vehicles and the Environment (EVE) with the UNECE Working Party on Pollution and Energy (GRPE) a framework with EPA acting as chair and ECCC as Secretary

Also in 2021, SC2 (Scientific Cooperation) of the AQC initiated a pilot series of science information exchange workshops. The goals of these workshops are to share knowledge about new developments and key advances in science topics of common interest, enhance scientific collaborations, and strengthen connections with SC1 (Program Monitoring and Reporting/Policy). The workshops received positive feedback from participants and continued in 2022. Topics discussed to date include: impacts of the COVID-19 pandemic on air quality; wildland fires; emerging pollutants/sources of increased interest; and $NH_3$.

Today, the Parties continue to collaborate in adherence to the commitments of the AQA and are working to develop new approaches to further mitigate persistent areas of concern for air pollutants under the AQA. One example is the Michigan-Ontario Ozone Source Experiment (MOOSE) Project where partners include the Michigan Department of Environment, Great Lakes, and Energy; the Lake Michigan Air Directors Consortium; the Ontario Ministry of Environment, Conservation and Parks; ECCC; EPA; the U.S. Forest Service; NASA; and the National Oceanic and Atmospheric Administration (NOAA). This project was initiated in 2021 and focuses on the southern Ontario-southeast Michigan area, areas where poor air quality is of great concern. The Detroit area in southeast Michigan was only recently redesignated as in attainment of the U.S. 2015 ozone NAAQS, and the southern Ontario area frequently exceeds the Canadian ambient air quality standards for ozone. MOOSE's primary purpose is to better understand what contributes to elevated ozone concentrations in this border region. Findings from the first year of MOOSE are being published in a special issue in the journal *Atmosphere* (Olaguer & Su, 2023). NOAA and ECCC scientists are also collaborating on the incorporation of a forest canopy parameterization developed at ECCC (Makar et al., 2017) into the CMAQ model to improve ozone predictions in the U.S.

Scientists at the EPA and ECCC also actively participate in the NADP Total Deposition (TDEP) Science Committee. The mission of TDEP is to improve estimates of atmospheric deposition by advancing the science of measuring and modeling atmospheric wet, dry, and total deposition of species such as sulfur, nitrogen, and mercury. Combining results from methods and models developed under TDEP with those developed under ECCC's Atmospheric Deposition Analysis Generated from optimal Interpolation from Observations (ADAGIO) project was identified as a core goal under the AQA (Schwede et al., 2019) Scientists from the TDEP and ADAGIO projects participate in the World Meteorological Organization (WMO) initiative on the Measurement-Model Fusion for Global Total Atmospheric Deposition project. The measurement-model fusion approaches used by the U.S. and Canada are leading the way on a global scale.

Finally, the Parties recently published the 2020-2022 AQA Progress Report (ECCC & US EPA, 2023) in line with AQA commitments. This is the fifteenth of these reports.

### 5.3 Summary

The Parties have strengthened their relationship through collaboration and science exchanges under the AQA. Since 1994, the Parties have notified each other of specific sources of pollution within 100 miles of the U.S.-Canada border. The U.S. and Canadian governments share data through a range of programs and tools such as AirNow and the NADP, and maintain ongoing informal dialogue across a range of topics related to monitoring networks and measurement methods. Canada and the U.S. collaborate and share emission inventories, summaries, and analyses under several agreements and reports such as the AQA bi-annual Progress Report, Arctic Council, and LRTAP. In addition to meeting their respective pollution commitments, the Parties completed multiple pilot projects and joint modeling and analysis under the Ozone Annex. These efforts have expanded information sharing and knowledge of transboundary transport, to the benefit of each country.

Looking to the future, SC2 co-chairs held a series of science exchange workshops to share information, continue to build connections, and inform policy-related dialogue on current and emerging transboundary issues of concern. The Parties have accomplished a great deal under the AQA, continue to collaborate through a variety of projects and look for ways to continue working together in pursuit of shared goals to improve air quality conditions.

# 6 Conclusions

### 6.1 Reflecting on Over 30 Years of the AQA

The Canada-U.S. AQA is a model of successful bilateral cooperation resulting in significant improvement in the environment over its three-decade history. In 2007, the U.S. and Canada achieved the emissions reduction targets laid out in both the acid rain and ozone annexes, and these emissions have continued to decrease in the subsequent years. Regulatory and non-regulatory programs designed to meet emissions commitments in the Ozone and Acid Rain Annexes, and other program goals for Canada and the U.S. individually, have contributed to these reductions.

Accomplishment under the Agreement to date include:

- Between 1990 and 2020, significant reductions in national emissions of pollutants that cause acid rain:
    - $SO_2$ emissions decreased by 78% in Canada and 93% in the U.S.
    - $NO_X$ emissions decreased by 36% in Canada and 70% in the U.S.
- Additionally, between 1990 and 2020, national VOC emissions decreased by 49% in Canada and 48% in the U.S.;
- Total sulfur deposition in the eastern U.S. reduced by 81% from 2000-2002 to 2018-2020;
- Total sulfur deposition in eastern Canada decreased by 70% from 2000-2002 to 2016-2018;
- Publication of 15 joint biennial progress reports under the Agreement;
- Publication of several joint science assessments on transboundary ozone and $PM_{2.5}$;
- Establishment of work plans for cooperation on vehicle and engine emissions and for addressing emissions from the oil and gas sector.

The Agreement has also served as an effective mechanism for scientific cooperation and bilateral exchange of credible information to inform policy recommendations. Similarly, the Agreement has provided a collegial avenue to address issues of concern regarding pollution sources, which has proven beneficial and effective.

## 6.2 Key Findings from this Assessment

As noted in the Joint US-Canada 2020-2022 Progress Report, the pollutants covered by the Agreement ($SO_2$, $NO_X$, VOCs, and ozone) continue to impact human health and the environment in both countries and remain a concern.

Continued and remarkable success in both countries in reducing pollutants contributing to acid deposition ($SO_2$ and $NO_X$) has led to recent signs of recovery. There are areas in both countries, most notably in eastern Canada, that are still recovering from the historic pollutant loadings and receiving acid deposition that may be in exceedance of current critical loads. Modeling suggests transboundary influence on total deposition, particularly in the less populated parts of northern Montana and the northern parts of the province of Ontario, where deposition is lower than in the northeastern U.S. Furthermore, deposition of reduced nitrogen (including $NH_3$ and $NH_4^+$) has not decreased in recent decades, and increased deposition of reduced nitrogen has been observed in some areas.

Ozone also continues to have significant impacts on public health and agricultural production in the U.S. and Canada, despite progress under Annex 3 of the Agreement. Transport from the U.S. continues to contribute a large fraction of anthropogenic ozone in Canada, with the largest influence in the Windsor-Quebec Corridor, as well as southwestern British Columbia, in the greater Vancouver and Victoria area, southern Alberta, the Greater Toronto-Hamilton area, and the Montreal area. Air monitoring stations in southern Ontario and southern Quebec continue to measure ozone concentrations which approach or exceed the CAAQS, and modeling projections suggest continued CAAQS exceedances in 2035. Modeling suggests that transboundary flow of ozone and its precursors from the U.S. to Canada contributes to a significant portion of health impacts in central and Atlantic Canada and is the dominant source of health impacts in Ontario, Quebec, Nova Scotia, New Brunswick, Newfoundland and Labrador, and Prince Edward Island. Transboundary flow from the U.S. into Canada is also estimated to contribute to reduced crop yield, particularly along the Windsor-Quebec City Corridor.

Although the Agreement does not include $PM_{2.5}$, emissions of some of the precursors of $PM_{2.5}$ are addressed via actions to reduce $NO_X$, $SO_2$, and VOCs. However, direct emissions of primary $PM_{2.5}$ and $NH_3$ (a $PM_{2.5}$ precursor) are not addressed under the Agreement. Direct emissions of primary $PM_{2.5}$ have plateaued in recent years and emissions of $NH_3$ have increased in both Canada and the U.S. Adverse health impacts of $PM_{2.5}$ exposure are well documented and both countries continue to take action to address their respective emissions. In recent years, $PM_{2.5}$ concentrations are largest near urban areas and particularly in the Ohio Valley, Atlantic coast, and the Windsor-Quebec Corridor, with observed concentrations for several stations in Canada approaching or exceeding the CAAQS. Although modeling projects that $PM_{2.5}$ concentrations will decrease by 2035, they are also projected to continue to exceed the CAAQS in some of Canada's largest cities. The analysis presented in this review and assessment finds greater transport of $PM_{2.5}$ from the U.S. to Canada. Recent modeling and analysis also indicate that transboundary $PM_{2.5}$ increases morbidity and mortality in Canada and has a larger health impact than transboundary ozone. Modeling results support the conclusion that the majority of transboundary $PM_{2.5}$ impacts are within several hundred kilometers of the border and felt predominantly in the Michigan-Ontario and Quebec regions – with the largest impacts in the Detroit-Windsor area.

## 6.3 Looking Ahead

Improving air quality continues to be a priority for both Canada and the U.S. The AQA is an important example of progress made through diplomacy and an effective catalyst for cooperation. However, air quality remains a concern, including in transboundary areas, on either side of the border. Furthermore, air quality priorities of today are not necessarily those of 30 years ago.

Important results have been achieved under the current AQA and both countries have fully met their obligations. However, transboundary air pollution continues to impact both countries from a health and environmental perspective. Based on the scientific and technical conclusions, in order to continue to meet the objective of the AQA "to control transboundary air pollution between the two countries", it is recommended the Parties update the Agreement.

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# Appendix A: Modeling

## ECCC Air Quality Modeling

Recent air quality modeling was conducted by ECCC using the GEM-MACH (Global Environmental Multiscale model - Modeling Air quality and Chemistry), an on-line air quality prediction model that simulates concentrations of different air pollutants (Gong et al., 2015; Moran et al., 2021; Pendlebury et al., 2018). Air quality modeling was carried out to evaluate transboundary transport of ozone and $PM_{2.5}$ between the U.S. and Canada. Annual simulations were carried out on a continental scale, latitude-longitude grid with 10-km grid spacing and 2019 meteorology, excluding wildfire emissions. Modeling scenarios were performed for emission base year 2015, and BAU scenarios for 2025, 2030 and 2035, with both Canadian and U.S. emissions, as well as scenarios without Canadian or without U.S. emissions.

The Canadian anthropogenic emissions used for the base year 2015 are based on Canada's APEI that was generated by ECCC in late 2017. The business-as-usual scenarios for 2025, 2030 and 2035 are also projected based on APEI that was released by ECCC in late 2017 (Sassi et al., 2021). The projected emissions estimates account for projected changes in population, economic activity, and energy use, as well as the implementation over this period of air pollution control measures and expected facility openings or closures. The Canadian emission projections include regulations that will be enforced in the years to come, such as: *Regulations Amending the Reduction of Carbon Dioxide Emissions from Coal-fired Generation of Electricity Regulations*, *Regulations Respecting Reduction in the Release of Methane and Certain Volatile Organic Compounds (Upstream Oil and Gas Sector)*, *Multi-Sector Air Pollutants Regulations (MSAPR)*, in addition to others, including provincial and territorial regulations.

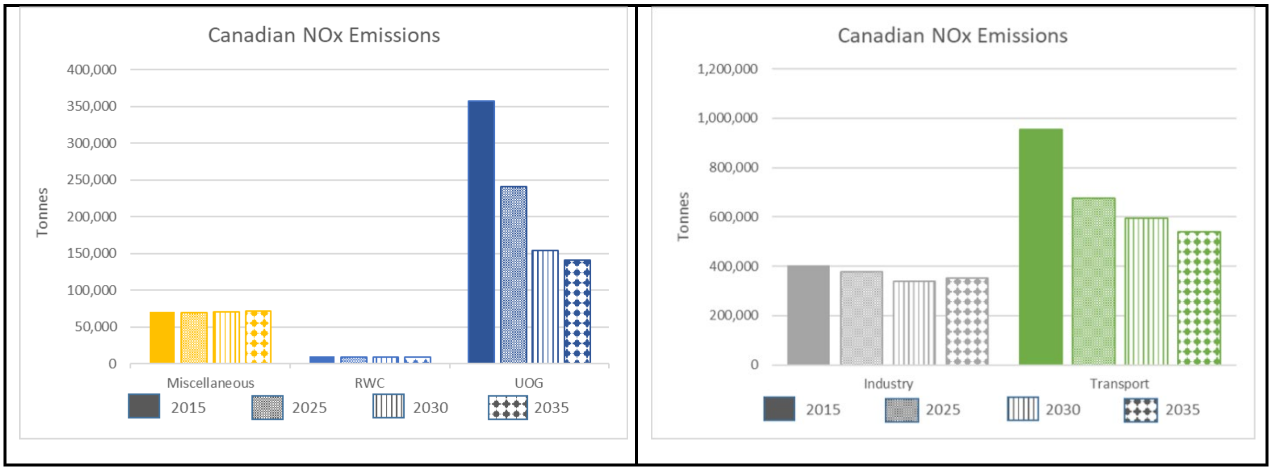
The anthropogenic U.S. emissions used for base year 2015 came from 2016 U.S. emissions, which principally include emissions from the 2016 EPA's Air QUAlity TimE Series Project (EQUATES) inventory. Additional details on this inventory can be found elsewhere (K. Foley, 2021; K. M. Foley et al., 2023). For the future BAU scenarios, U.S. projections for 2026 and 2032 were obtained from the 2016v2 EPA Modeling Platform (Eyth, 2021). The U.S. projections include several regulations, as described in the *Technical Support Document (TSD) Preparation of Emissions Inventories for the 2016v2 North American Emissions Modeling Platform* report (US EPA, 2021d).

Mexican emissions for all scenarios were based on the 2008 Mexican National Emissions Inventory, which was also obtained from the EPA 2016 Air Emissions Modeling Platform (Eyth, 2021).

### Emissions Input

The projected emissions of $NO_X$ and VOCs that were used in the GEM-MACH simulations for the BAU 2025, 2030 and 2035 scenarios are compared to the 2015 base-year emissions in Figure A-1 to Figure A-3. The projected years include regulations that will be in force in the future. Figure A-1 shows that between 2015 and 2035, miscellaneous[1] $NO_X$ emissions appear to increase slightly (2% overall from 2015 to 2035), residential wood combustion (RWC) emissions appear stable, and upstream oil and gas (UOG) emissions decrease substantially (a 60% decrease overall from 2015 to 2035, but the largest decrease of 32% occurs between 2015 and 2025). Industry emissions decrease in 2020 and 2025 and then increase slightly in 2035 (an overall decrease of 12% from 2015 to 2035) and transport emissions show an overall decrease of 44% from 2015 to 2035, but the largest reduction of 29% occurs between 2015 and 2025.

*Figure A-1. Canadian $NO_X$ emissions for 2015, 2025, 2030 and 2035.*



[1] Miscellaneous emissions include emissions from the following emission source categories: commercial/residential/institutional, paints and solvents, Incineration and waste, and fires (includes prescribed burning and structural fires).

Figure A-2 shows the Canadian VOC emissions for base year 2015, and BAU 2025, 2030 and 2035. Emissions from agriculture remain stable across all years, while emissions from industry show an increase of 91% from 2015 to 2035, with the biggest increase of 69% from 2015 to 2025. Miscellaneous sources show an overall increase of 39% between 2015 and 2035. RWC emissions increase in 2025 but decrease in 2035 for an overall decrease of 3% between 2015 and 2035. UOG emissions decreases between 2015 and 2025 by 19% and then increases from 2025 to 2035, for an overall decrease of 2% between 2015 and 2035. Transport emissions show an overall decrease of 45% between 2015 and 2035.

*Figure A-2. Canadian VOC emissions for 2015, 2025, 2030 and 2035.*

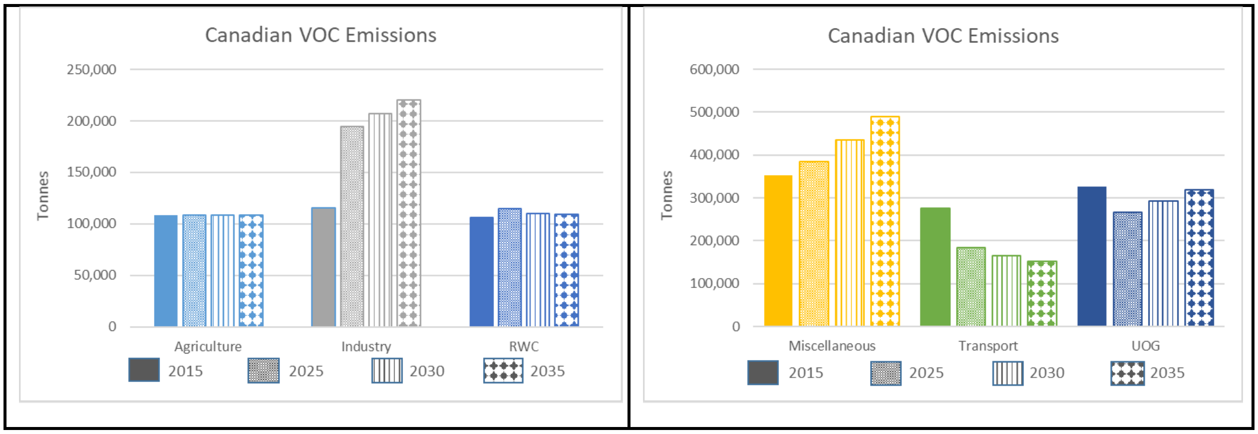
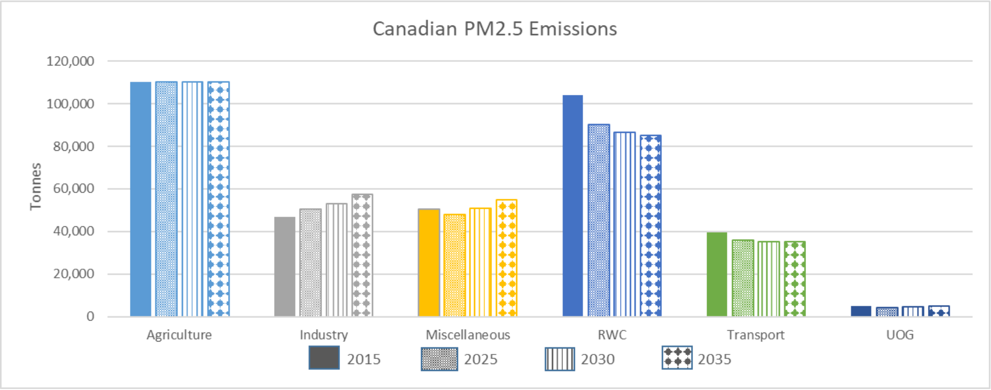
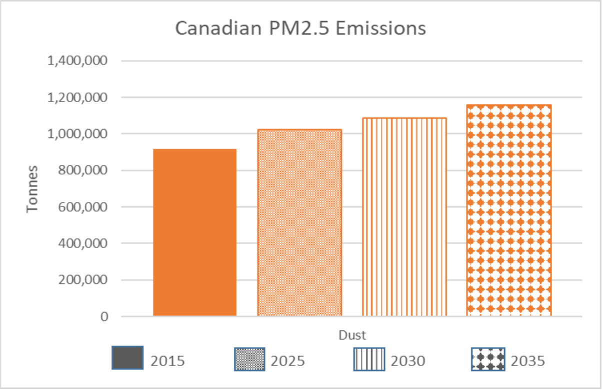


Figure A-3 shows the Canadian primary $PM_{2.5}$ emissions for base year 2015, and BAU 2025, 2030 and 2035. Agriculture emissions are stable between base year 2015 and 2035. Industry emissions increase by 23% from 2015 to 2035, while miscellaneous emissions decrease slightly and then increase, with an overall increase of 9% from 2015 to 2035. RWC shows an overall decrease of 18% from 2015 and 2035, with the biggest reduction between 2015 and 2025 (13%). Transport emissions decrease by 12% from

2015 to 2035. Dust emissions increase by 26% from 2015 to 2035. Upstream oil and gas emissions are relatively small and remain stable over the period.

*Figure A-3. Canadian primary PM$_{2.5}$ emissions for 2015, 2025, 2030 and 2035.*





## Model Uncertainty

All modeling runs are performed with the same 2019 meteorological conditions, and for one year only. This highlights the impact of the changes in emissions and resulting atmospheric chemistry. However, these runs are not meant to represent possible future scenarios since they do not take into account the expected changes in climate. Warming temperatures are expected to worsen air quality in Canada and the U.S. through warmer summer exacerbating poor summertime air quality events, and lengthening the season in which poor air quality events might occur. In addition, where changes are small, one year may not be enough time to produce statistically significant results.

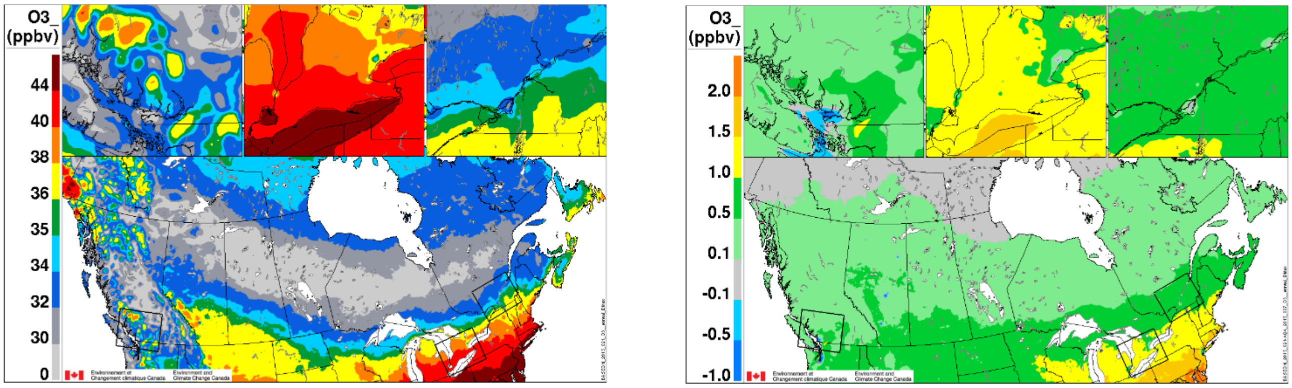
## 20% Modeling Runs

Zero-out emissions scenarios are extreme cases that can be used to look at the impact of one region on another in a qualitative sense. However, due to nonlinearity in the chemistry they cannot be used for an accurate quantitative analysis. For example, pollutant concentrations in each zero-out scenario cannot be added together to get the BAU scenario amounts, even above background levels. Runs with 20% reduction scenarios were also performed with GEM-MACH that show less of an impact on pollutants and show regions of nonlinearity, but broadly back up the message of the zero-out emissions scenarios. Specific examples are given in each section.

Figure A-4 shows the gridded yearly average daily maximum ozone concentrations in ppbv for 2015. Figure A-4a shows the 2015 BASE CASE concentrations. Figure A-4b shows differences between the BASE

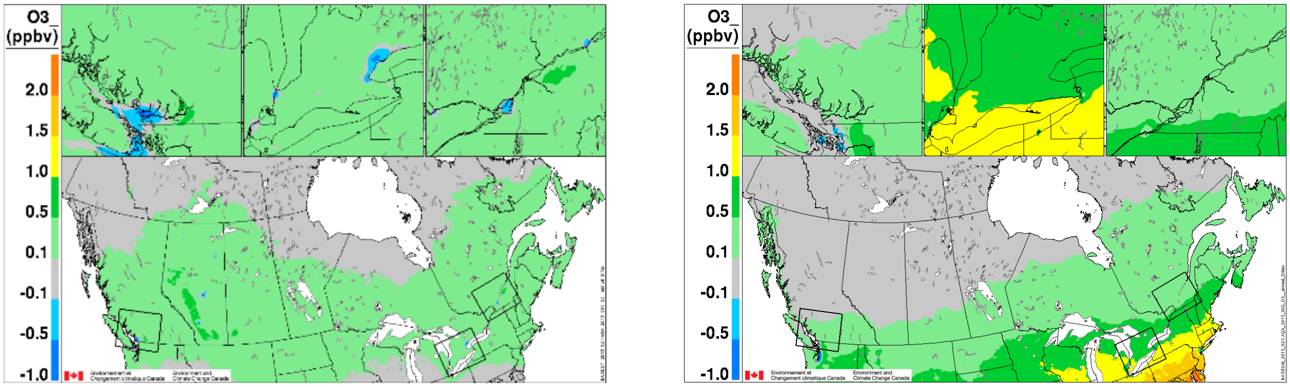
CASE and the Canada and U.S. 20% reduction scenario concentrations. Figure A-4c shows differences between the BASE CASE and the Canada 20% reduction scenario concentrations (cf. Figure 3-8d), and Figure A-4d shows differences between the BASE CASE and the U.S. 20% reductions scenario concentrations (cf. Figure 3-8c).

*Figure A-4. Yearly average daily maximum ozone concentrations from ECCC modeling for (a) BASE CASE 2015, (b) BASE minus Canada and U.S. 20% reduction 2015, (c) BASE minus Canada 20% reduction 2015, and (d) BASE minus U.S. 20% reduction 2015.*



a. BASE case 2015

b. BASE – Canada and U.S. 20% reduc. 2015



c. BASE – Canada 20% reduc. 2015

d. BASE – U.S. 20% reduc. 2015

## U.S. EPA Air Quality Modeling

### Ozone

U.S. EPA has completed two recent modeling projects that quantify the transboundary transport of ozone between the U.S. and Canada. Table A-1 summarizes the two projects that include eight simulations: the 2020 Ozone Policy Assessment (PA) (US EPA, 2020d) and the Final Revised Cross State Air Pollution Rule Update (RCU) (US EPA, 2020a). The air quality modeling used to support each of these efforts relied on different photochemical models, simulation years, and technology to attribute pollution to the U.S. or Canada. The details of the simulations are summarized in Table A-1 and in the discussion below.

For the 2020 PA EPA performed CMAQ v5.2.1 simulations for 2016. That project included a base case with all emissions and several simulations with specific regions of all anthropogenic emissions removed.

By comparing the base case to a zero-out of U.S. or Canada, we can attribute a portion of the total pollution to that country. For that model application, a pure Canada zero-out was not performed. Instead, both Canada and Mexico were simultaneously removed. The two countries are far enough apart and transport patterns sufficiently different that the Canada (or Mexico) contribution can generally be distinguished by location. Although the focus of that 2020 PA was ozone, the model results also include PM predictions that can be useful for this work.

The RCU used CAMx (v7., beta 6) for 2016, 2023 and 2028 years. In 2023 and 2028, the Ozone Source Apportionment Technology with Anthropogenic Precursor Culpability Assessment (APCA) was enabled on a state-specific basis within the U.S. and for Canada. For more detail on the APCA technology and application, please see the support documentation (US EPA, 2020a).

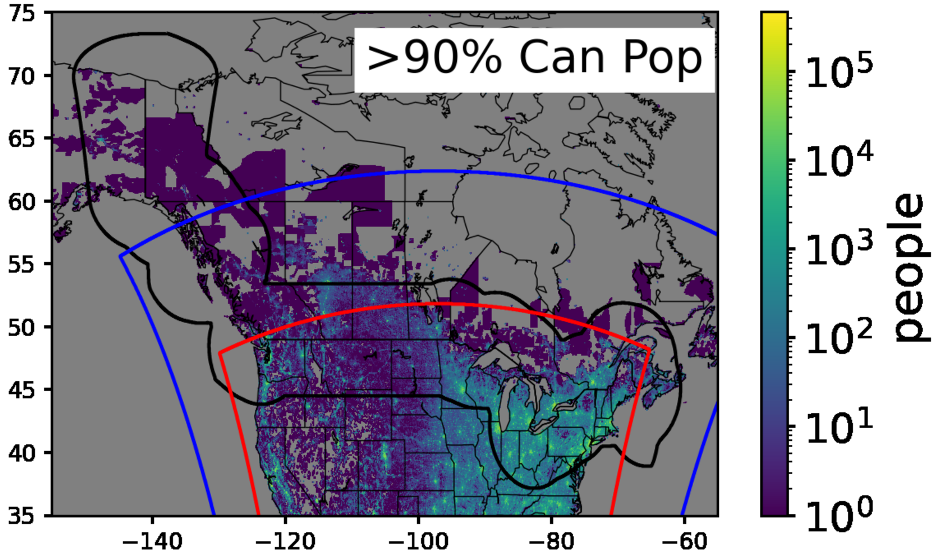
From these simulations, we can see how ozone changes from a nominal present-day (here, 2016) to various futures (2023, 2028). For ozone, there are multiple present-day estimates (PA16, RCU16) and ozone source attribution in the present-day and futures (PA16, RCU23, RCU28). Trends are best characterized by staying within a consistent set of simulations (*e.g.*, RCU16-RCU28).

*Table A-1. Description of simulations used in the U.S. analysis for ozone.*

| **Short Name** | **Previous Application** | **Model: Years**<br>**Attribution** | **108 km** | **36 km** | **12 km** |
|---|---|---|---|---|---|
| **PA16** | 2020 Ozone Policy Assessment | CMAQ: 2016<br>Attribution: Zero-out of all anthropogenic emissions (direct and precursors) separately for the U.S. and Canada, which allows for attribution of ozone, $PM_{2.5}$, nitrogen, sulfur | X | X | X |
| **RCU16**<br>**RCU23**<br>**RCU28** | Final Revised CSAPR Update | CAMx: 2016, 2023, 2028<br>Attribution: OSAT/APCA (2023 and 2028) | | | X |

As noted in Table A-1, the different studies used a combination of three domains. The largest domain covers the Northern Hemisphere using 108 km resolution in a polar stereographic project. This domain covers all the areas of interest, but does not resolve spatial gradients relevant to urban air quality. Our regulatory projects use a 12-km domain that covers the continental U.S. relatively tightly, but covers a smaller area and notably excludes large parts of Canada. To capture the full influence of the surrounding area, particularly international sources, we include a 36-km domain with larger extent than the 12 km and finer resolution than the 108 km simulations. These domains can be seen in Figure A-5.

*Figure A-5. U.S. and Canada population map with the black outline of a proposed analysis zone for the agreement analysis.  The EPA 36 km (blue) and 12 km (red domain extents are overlaid for reference. The 108 km grid domain covers the entire figure.*



The 36 km domain covers most of the analysis region except northern Territories and Alaska – this covers 99.9% of the population within the analysis region. The 12 km domain covers most of the analysis region along the 48°N latitude, but excludes portions of the west and the entire Canada/Alaska border. The 12 km domain includes 84.9% of the Canadian population within the analysis region. The 36km domain completely covers the 48°N latitude border and portions of the Alaska/Canada border, but only the 108 km domain completely covers the analysis region. Along the border, as shown in Figure A-5 *,* 92.4% of Canada's population of 38.4 million[34] lives within 500 km of the U.S.-Canada border while a much smaller fraction of the U.S. population of 332.2 million[35] lives within 500 km of the border. This 500 km area includes most of the PEMA and also includes part of Virginia and all or parts of the northern U.S. states in the west (Washington, northern Idaho, Montana, northern Wyoming, North Dakota, northern South Dakota). To compare these results to each other, we use a common domain defined by the Center for International Earth Science Information Network (CIESIN) Global World Population (v4.2) dataset. This dataset uses a latitude/longitude grid with 2.5-minute resolution (~4.5 km at northern mid latitudes). Each grid cell is assigned a concentration from the photochemical model based on its cell centroid. By using a common grid, we can also apply population weighting to our results.

### $PM_{2.5}$

The EPA has completed two recent modeling projects that quantify the transport of $PM_{2.5}$ from U.S. and Canada. The two projects are the 2020 Ozone Policy Assessment (PA) and the Regional Haze Rule (RHR) modeling (US EPA, 2019b). The PA was discussed already in the Ozone sections above and Table A-1.

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[34] Statistics Canada, Quarterly population estimate from October 2021, https://www150.statcan.gc.ca/n1/daily-quotidien/211216/dq211216c-eng.htm?HPA=1&indid=4098-1&indgeo=0

[35] U.S. Census Bureau Population Clock as of October 1, 2021, https://www.census.gov/popclock/

Although $PM_{2.5}$ was not the target of the PA study, the results are shown here for comparison to the RHR. The RHR modeling is characterized by Table A-2 and the following narrative:

- The RHR performed Comprehensive Air quality Modeling with extensions (CAMx v7beta) for a 2016 and 2028 future year. In the future year, the simulations used the Particulate matter Source Attribution Technology (PSAT) to attribute U.S.-sector level contributions and Canada total contributions. For the purposes here, the U.S.-sectors are summed to create a total U.S. contribution.

*Table A-2. Description of simulations used in the U.S. analysis for ozone.*

| **Short Name** | **Previous Application** | **Model: Years**<br>**Attribution** | **108km** | **36km** | **12km** |
|---|---|---|---|---|---|
| **RHR16**<br>**RHR28** | Regional Haze Rule | CAMx: 2016, 2028<br>Attribution: PSAT (2028) | | X | X |

Using the PA and RHR modeling, PM has multiple estimates (PA16 and RHR16) and attribution estimates (PA16 and RRH28). The change in $PM_{2.5}$ concentrations between a nominal present day (2016) and a future (2028) provides an estimate of how emission changes are expected to influence $PM_{2.5}$ in the future. The attribution modeling (PA16 zero-out and RHR28 PSAT) provides an estimate of the magnitude of transboundary impacts.

As described in the ozone section above, the results use a combination of three domains and resolutions. The results are combined into a unified grid consistent with the Global World Population (v4.2) dataset. This dataset uses a latitude/longitude grid with 2.5-minute resolution (~4.5 km at northern mid latitudes).

For $PM_{2.5}$, the models provide chemically speciated results with different representations of particle diameter modes (nucleation, accumulation, and coarse). Measurements use an inlet to selectively measure particles below 2.5 μm diameter. This “cut size” includes all of nucleation mode, most of accumulation, and a small part of the coarse mode due to imperfect size selectivity. For the PA analysis, we are looking at the total Aitken and accumulation modes from the model to estimate total $PM_{2.5}$ that is measured. For the RHR analysis, the sum of fine mode aerosols are used.

# Appendix B: Meteorological Analysis of Transboundary Flow

Wind data from the Modern-Era Retrospective analysis for Research and Applications, Version 2 (MERRA-2) were used to provide a qualitative characterization of transport that does not rely on complex chemistry. The low-altitude winds (50-meter) are most relevant for transport along the border, whereas long-distance transport depends heavily on upper winds (500 hPa). The mid-level winds (850 hPa) show the transition between levels.

Figure B-1 shows the typical wind patterns as streamlines for three representative summer-time months and three representative altitudes. The summer months are relevant for the transport of ozone, $PM_{2.5}$, and their precursors. The upper-level winds (500 hPa) are only slightly favorable for Canada-to-U.S. long-range transport. The upper winds are fairly consistent from west-to-east with a slight north-to-south component. This is consistent with long-range transport from pacific Canada to the U.S. northeast. As we will see in the modeling results, this is not a major transport vector. The low-level winds (10-meter) show varying degrees of favorable transport for U.S.-to-Canada transport. The low-level winds vary by season and location. In the U.S. northwest, the transport patterns generally parallel the border at all seasons – indicating little transport. In the U.S. northeast in May, the streamlines generally parallel the U.S./Canada border except near New York. This emphasizes transport from Michigan to Toronto/Windsor and from Toronto/Windsor to New York. In July, the streamlines are more consistently south-to-north until they near the border where a west-to-east component is added. This will emphasize transport from Michigan and the Ohio Valley to Canada. By September, the streamlines are south-to-north showing transport from the U.S. to Canada. These results are generally consistent with photochemical modeling results shown later.

*Figure B-1. MERRA-2 monthly average surface pressures and wind fields are shown for May, July, and September of 2016. The winds are show at 50-meters (50M), 850 hPa and 500 hPa that represent increasing altitude.*

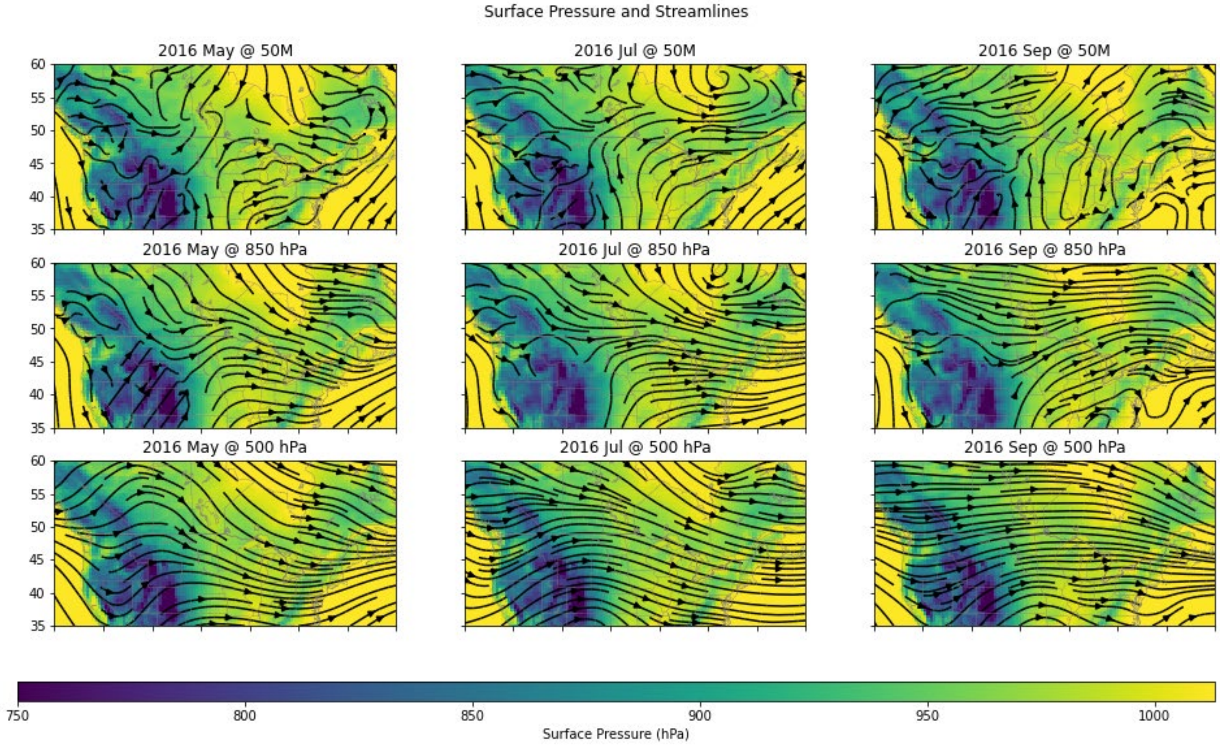

Figure B-2 shows the typical winds for months outside of the summer season (January, March and November), which are relevant to the transport of $PM_{2.5}$ and precursors, which can have peaks in both the summer and winter (Zhao et al., 2018). Similar to the summer months, the high-altitude wind flows are consistently west-to-east. The north-to-south component near Montana and Alberta is stronger in January. This is seen in both the 500 hPa and 850 hPa wind flows. This suggests some favorability for transport from Canada to the U.S. that is more relevant in this region and season and will affect annual average $PM_{2.5}$ more than it would summer ozone. Unlike the summer months, the winter does not have a clear south-to-north flow at the surface in the northeast. Instead, there is either convergence or flow parallel to the border. So, the near-surface winds in these months are generally less favorable for transborder transport in the northeast.

*Figure B-2. MERRA-2 monthly average surface pressures and wind fields are shown for January, March, and November of 2016. The winds are shown at 50-meters (50M), 850 hPa and 500 hPa that represent increasing altitude.*