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HEALTH IMPACT SCALE FOR AIR QUALITY IMPROVEMENTS IN THE CANADIAN LOWER FRASER VALLEY AIRSHED

HEALTH IMPACT SCALE FOR AIR QUALITY IMPROVEMENTS IN THE CANADIAN LOWER FRASER VALLEY AIRSHED

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EXECUTIVE SUMMARY

Metro Vancouver's Integrated Air Quality and Greenhouse Gas Management Plan (IAQGGMP), adopted by the Metro Vancouver Board of Directors in 2011, establishes three goals related to Metro Vancouver's authority to manage air quality:

1. protect public health and the environment,
2. improve visual air quality, and
3. minimize the region's contribution to global climate change.

The IAQGGMP includes 81 actions to support these goals. Although all of the actions have the potential to contribute toward achieving one or more of the three goals listed above, a formal analytical framework was not developed to rank or prioritize actions.

It is increasingly important for air quality management agencies to consider the cumulative health impacts of different air contaminants, their sources and ambient concentrations. A broader evaluation of the health impacts associated with key criteria air contaminants and hazardous air pollutants could inform planning decisions so that Metro Vancouver focuses policy and program development on actions expected to yield the most public health benefits in the region.

The goal of this study was to develop a relative health impact scale that indicates how changes in emissions of different air contaminants impact public health in the Canadian portion of the Lower Fraser Valley (LFV). Metro Vancouver shares the LFV airshed with the Fraser Valley Regional District to the east and Whatcom County in the State of Washington to the south.

The health impact scale was designed for the following uses:

1. To support prioritization of actions in Metro Vancouver's next air quality and greenhouse gas management plan;
2. To support other future air quality regulatory development initiatives such as evaluation of fees and regulatory impact assessments; and
3. To evaluate air quality improvements at a regional scale.

The study does not consider other impacts from air contaminants, including impacts on vegetation, agriculture, visual air quality, material degradation and climate change.

The ambient air contaminants considered in the study included criteria air contaminants (CACs) and hazardous air pollutants (HAPs, or air toxics) shown in Table ES-1.

Table ES-1 Ambient Air Contaminants Considered for the Health Impact Scale

Criteria Air Contaminants (CACs)	Hazardous Air Pollutants (HAPs)
Fine particulate matter ($PM_{2.5}$)	1,3-butadiene
Nitrogen dioxide (NO_2)	Benzene
Carbon monoxide (CO)	Carbon tetrachloride
Sulphur dioxide (SO_2)	Chromium VI
Secondarily formed ozone (O_3) and precursors	Diesel particulate matter (DPM)
	Formaldehyde

Exposure to ambient air concentrations of these air contaminants can potentially contribute to adverse human health effects, and have been documented to contribute to or be associated with one or more of the following adverse health endpoints shown in Table ES-2:

Table ES-2 Health Endpoint

Health Endpoints (1)	
Acute Exposure Mortality	Chronic Exposure Lung Cancer Mortality
Acute Respiratory Symptom Days	Chronic Exposure Respiratory Mortality
Adult Chronic Bronchitis Cases	Elderly Cardiac Hospital Admissions
Asthma Symptom Days	Minor Restricted Activity Days
Cardiac Emergency Room Visits	Respiratory Emergency Room Visits
Cardiac Hospital Admissions	Respiratory Hospital Admissions
Child Acute Bronchitis Episodes	Restricted Activity Days
Chronic Exposure Cerebrovascular Mortality	Non-fatal Cancer
Chronic Exposure Ischemic Heart Disease Mortality	

Notes: (1) Air contaminants have been associated with other health endpoints that are not listed in this table, including developmental, neurological, and cancers other than lung cancer.

The health impact of an action that results in a reduction in emissions of an air contaminant is dependent on:

- The change in ambient air quality that results from the reduction in emissions of that air contaminant; and
- The health impact associated with a change in population exposure due to that change in ambient air quality.

The health outcomes affected by air pollution vary by air contaminant, which makes it difficult to compare or sum the health benefits associated with emission reductions in multiple air contaminants. However, where the economic valuation (in dollars) of the health benefits can be determined, the impact of changes to multiple air contaminants can be assessed as the simple sum of the valuations associated with each air contaminant.

In general, for each individual air contaminant with an emission reduction:

$$\begin{array}{c}\textbf{Health}\\ \textbf{Impact}\\ \textbf{Scale}\\ \\ \mathbf{\$/tonne}\end{array} = \begin{array}{c}\textbf{Reduction in ambient}\\ \textbf{concentrations per tonne}\\ \textbf{of emission reduction}\\ \\ \mathbf{(\mu g/m^3)\ /\ (tonne/year)}\\ \mathbf{or\ ppm/(tonne/year)}\end{array} \times \begin{array}{c}\textbf{Valuation of Health}\\ \textbf{Benefits associated}\\ \textbf{with reduction in}\\ \textbf{ambient concentrations}\\ \\ \mathbf{(\$/year)/(\mu g/m^3)}\\ \mathbf{or\ (\$/year)/ppm}\end{array}$$

For actions that involve reductions in emissions of multiple air contaminants, the overall impact of the action is the summation of the impacts of individual air contaminants:

$$\begin{array}{c}\textbf{Impact}\\ \\ \mathbf{\$/year}\end{array} = \sum_{i=1}^{n} \begin{array}{c}\textbf{Health Impact}\\ \textbf{Scale for}\\ \textbf{Contaminant i}\\ \\ \mathbf{\$/tonne}\end{array} \times \begin{array}{c}\textbf{Reduction in}\\ \textbf{Emissions of}\\ \textbf{Contaminant i}\\ \\ \mathbf{tonnes/year}\end{array}$$

The scope of this project did not support a rigorous assessment of ambient concentration changes and health benefits of multiple air contaminants through detailed dispersion and photochemical modelling and analysis of changes in population exposure. Instead, the relative health impact scale was developed based on existing information and studies related to air quality in the Canadian LFV and other jurisdictions. As a result, the estimates of health impacts used to develop the scale relied on simplifying assumptions that are described below. A sensitivity analysis was conducted to provide some information on the effect of these assumptions.

For CACs, the relationships between emissions and ambient air concentrations were based on correlations between annual emission inventories and monitored ambient concentration in the Canadian LFV for the period 1995 to 2015. The valuation of health benefits was based on the Air Quality Benefits Assessment Tool (AQBAT), Version 2a, developed by, and available from, Health Canada. Valuations were converted to 2017 dollars.

For HAPs, the relationships between emissions and ambient air concentrations were based on the emission inventory and monitored (or estimated) ambient air concentrations for 2010, as documented in a previous study for Metro Vancouver (Sonoma, 2015). The valuation of health benefits was based on unit risk factors for developing cancer (from Health Canada or California Office of Environmental Health Hazard Assessment), and the associated valuation developed for non-fatal cancers used in the Bay Area Air Quality Management District multi-pollutant evaluation method (BAAQMD, 2016) converted to 2017 Canadian dollars.

The health impact scale was developed as a dollar valuation in order to sum impacts from multiple health outcomes. Recognizing that it may often be useful to scale or rank impacts without specifying dollar values, a relative scale was specified that is proportional to the valuation. The health impact scale is given below in Table ES-3.

Table ES-3 Health Impact Scale by Air Contaminant Emitted

Air Contaminant	Air Contaminant Type	Health Impact Scale	
		Valuation ($/tonne)	Relative Scale
Chromium VI	HAP	$14,153,000	32,714.5
$PM_{2.5}$	CAC	$356,822	824.8
VOC	CAC	$51,967	120.1
NOx	CAC	$30,324	70.1
1,3-Butadiene	HAP	$27,016	62.4
SOx	CAC	$19,844	45.9
Diesel PM	HAP	$16,370	37.8
Carbon tetrachloride	HAP	$6,514	15.1
Formaldehyde	HAP	$4,025	9.3
Benzene	HAP	$532	1.2
CO	CAC	$433	1.0

It is important to understand that the impact of any action is the product of the health impact scale (in $/tonne) and the reduction in emissions (in tonnes) associated with the action. While it may appear that the scale is dominated by chromium VI, total emissions of chromium VI were less than 42 kg in 2010, and potential reductions of emissions are expected to be in the order of 1 kg or less. In contrast, the potential reductions of other air contaminants are 3 to 6 orders of magnitude higher. As a result, the health and economic impacts of potential chromium VI reductions are relatively small compared to those of other air contaminants.

The health impact scale was applied to ten example actions and associated emission reductions provided by Metro Vancouver. They were developed for demonstration purposes and do not necessarily represent expected Metro Vancouver policy directions. The valuation, relative impact, and resulting rank of each action are given in Table ES-4.

Table ES-4 Valuation, Relative Impact and Rank by Actions

Actions		Valuation	Relative Impact	Rank
Action No.	Description of Action			
8	Restrictions on wood burning appliances	438,000,000	309,212	1
10	Implementation of ECA	193,000,000	136,231	2
9	Impact of Nonroad Diesel Engine Emissions Regulation	36,100,000	25,480	3
6	Zero emission vehicle mandate	22,300,000	15,759	4
4	Heavy duty vehicle inspection maintenance program	16,900,000	11,927	5
5	Residential wood stove incentives	3,330,000	2,356	6
1	Apply California-style VOC limits for automotive refinishing products	2,600,000	1,836	7
3	Voluntary program to replace gasoline lawn eqmt	832,000	588	8
7	Heavy duty vehicle licensing program	772,000	545	9
2	Phase out CrVI emissions from miscellaneous industrial sources	22,400	16	10

An analysis was conducted to investigate the sensitivity of the overall impacts to changes in the impact scales of specific air contaminants. Metro Vancouver identified nine (9) alternate scenarios that each represent a change in the impact scale compared to the “Base Case” health impact scale documented in Section 2.

The sensitivity analysis found that, for the sample actions evaluated, valuations are dominated by, and significantly affected by, the value of a statistical life (currently $7,612,582 CAD 2017 dollars). However, the analysis also found that the resulting ranking of actions was relatively insensitive to the range of changes to the scale that were investigated. That is, the range of changes would have little effect on prioritization of actions.

The scope of work did not allow for secondary formation of particulate matter to be incorporated into the impact scale.

It is recommended that:

- Air contaminants formation and algorithms relating changes in ambient concentrations to changes in emissions used in this study should be further investigated.
- Formation of secondary $PM_{2.5}$ should be further investigated Findings of the sensitivity analysis suggest that this could have an impact on valuations produced by the scale.
- Updating the health impact scale with outputs from the new version of AQBAT should be considered for the future. Changes in the AQBAT model may improve accuracy of CAC valuation, and provide more thorough treatment of a few HAPs. This project relied heavily on the current version of AQBAT (version 2a) for valuation of health impacts of CACs. Health Canada has indicated they are very close to release of a new version of AQBAT that includes modified concentration response functions and valuations. In addition, the new version includes some HAPs.

1. INTRODUCTION

1.1 Background

The Metro Vancouver Regional District (Metro Vancouver) is a regional district of 2.5 million residents, and has delegated authority for air quality management within its regional district boundaries. Metro Vancouver is situated within the Lower Fraser Valley (LFV) airshed – an international airshed shared with the Fraser Valley Regional District (FVRD) to the east and Whatcom County in the State of Washington to the south. As such, air quality management planning efforts address the international airshed to the extent possible, including collaboration with airshed partners in monitoring air quality, planning, and implementing improvement actions.

Under its delegated authority, Metro Vancouver has adopted three regional air quality management plans since 1994. The current Integrated Air Quality and Greenhouse Gas Management Plan (IAQGGMP), adopted by Metro Vancouver's Board of Directors in 2011, establishes three goals related to Metro Vancouver's authority to manage air quality:

1. protect public health and the environment,
2. improve visual air quality, and
3. minimize the region's contribution to global climate change.

The IAQGGMP includes 81 actions to support these goals. Although all of the actions have the potential to contribute toward achieving one of the three goals listed above, a formal analytical framework was not developed to rank or prioritize actions.

It is increasingly important for air quality management agencies to consider the cumulative health impacts of different air contaminants, their sources and ambient concentrations. A broader evaluation of the health impacts associated with key criteria air contaminants and hazardous air pollutants could inform planning decisions so that Metro Vancouver focuses policy and program development on actions expected to yield the most public health benefits in the region.

1.2 Objectives

The goal of this study was to develop a relative health impact scale that indicates how changes in emissions of different air contaminants impact public health in the Canadian LFV. The health impact scale was designed for the following uses:

1. To support prioritization of actions in MV's next air quality and greenhouse gas management plan;
2. To support other future air quality regulatory development initiatives such as evaluation of fees and regulatory impact assessments; and
3. To evaluate impacts at a regional scale.

The study did not consider other impacts from air contaminants, including impacts on vegetation, agriculture, visual air quality, material degradation and climate change.

Ideally, the impact scale would be a relative indication of health benefit per tonne of emissions, and would be developed for a priority list of air contaminants. A simple scale, presented in tabular format was desired to allow relatively straightforward application for a range of purposes.

The air contaminants considered included the criteria air contaminants (CACs) and hazardous air pollutants (HAPs, or air toxics) identified and prioritized by MV and subject to availability of information. The final list of air contaminants developed in consultation with Metro Vancouver is given in Table 1. The study focused predominantly on CACs and ground-level ozone that are input to AQBAT Version 2a. The selected HAPs have been associated with some of the highest cancer risks in past risk assessments for the Metro Vancouver area (e.g., Sonoma, 2015; Levelton, 2007).

Table 1 Ambient Air Contaminants Considered for the Health Impact Scale

Criteria Air Contaminants (CACs)	Hazardous Air Pollutants (HAPs)
Fine particulate matter ($PM_{2.5}$)	1,3-butadiene
Nitrogen dioxide (NO_2)	Benzene
Carbon monoxide (CO)	Carbon tetrachloride
Sulphur dioxide (SO_2)	Chromium VI
Secondarily formed ozone (O_3) and precursors	Diesel particulate matter (DPM)
	Formaldehyde

1.3 Limitations

The scope of this project did not support a rigorous assessment of ambient concentration changes and health benefits of multiple air contaminants through detailed modelling and analysis. Instead, the relative health impact scale was developed based on existing information and studies related to air quality in the LFV and other jurisdictions.

As a result, the estimates of health impacts used to develop the scale relied on numerous simplifying assumptions. These assumptions are described herein, and users of the health impact scale are advised to consider how these assumptions may affect the scale for any specific use. A sensitivity analysis (see Section 3) provides some information on the effect of these assumptions.

This project relied heavily on the current version of AQBAT (version 2a) for valuation of health impacts of CACs. Health Canada has indicated they are very close to release of a new version of AQBAT that includes modified concentration response functions and valuations. In addition, the new version includes some HAPs. Updating the health impact scale with outputs from the new version of AQBAT should be considered for the future. Changes in the AQBAT model may improve accuracy of the CAC valuation, and provide more thorough treatment of a few HAPs.

2. DEVELOPMENT OF THE HEALTH IMPACT SCALE

The health impact of a reduction in emissions of an air contaminant is dependent on:

- The change in ambient air quality that results from the reduction in emissions of that air contaminant; and
- The health benefits associated with that change in ambient air quality.

The health outcomes affected by air pollution vary by air contaminant, which makes it difficult to compare or sum the health benefits associated with emissions reductions from multiple air contaminants. However, where the economic valuation (in dollars) of the health benefits can be determined, the impact of changes to multiple air contaminants can be assessed as the sum of the valuations associated with each air contaminant.

In general, for each individual air contaminant with an emission reduction:

$$\boxed{\begin{array}{c}\textbf{Health}\\ \textbf{Impact}\\ \textbf{Scale}\\ \\ \textbf{\$/tonne}\end{array}} = \boxed{\begin{array}{c}\textbf{Reduction in ambient}\\ \textbf{concentrations per tonne}\\ \textbf{of emission reduction}\\ \\ (\mu g/m^3)\ /\ (\text{tonne/year})\\ \text{or ppm/(tonne/year)}\end{array}} \times \boxed{\begin{array}{c}\textbf{Valuation of Health}\\ \textbf{Benefits associated}\\ \textbf{with reduction in}\\ \textbf{ambient concentrations}\\ \\ (\$/\text{year})/(\mu g/m^3)\\ \text{or } (\$/\text{year})/\text{ppm}\end{array}}$$

For actions that involve reductions in emission of multiple air contaminants, the overall impact of the action is the summation of the impacts of individual air contaminants:

$$\boxed{\begin{array}{c}\textbf{Impact}\\ \\ \textbf{\$/year}\end{array}} = \sum_{i=1}^{n} \boxed{\begin{array}{c}\textbf{Health Impact}\\ \textbf{Scale for}\\ \textbf{Contaminant i}\\ \\ \textbf{\$/tonne}\end{array}} \times \boxed{\begin{array}{c}\textbf{Reduction in}\\ \textbf{Emissions of}\\ \textbf{Contaminant i}\\ \\ \textbf{tonnes/year}\end{array}}$$

The following sections outline the development of the two basic components of the Health Impact Scale – the relationships between air quality and emissions, and the relationships between valuation of health benefits and air quality. The criteria air contaminants (CACs) and hazardous air pollutants (HAPs) are discussed separately since the data sources and methodologies are unique to each.

2.1 Criteria Air Contaminants (CACs)

2.1.1 Health Benefits Valuation of Air Quality Reductions

The health benefits valuation associated with reduced air contaminant concentrations for CACs was derived using the Air Quality Benefits Assessment Tool (AQBAT) version 2a, developed by and available from Health Canada. AQBAT contains databases to complete health benefits valuations across Canada down to the Census Division (CD) level. AQBAT estimates the human health impacts from changes in ambient air contaminant concentrations. It does not consider any microenvironment of human exposure or human activity in the analysis.

The Canadian Lower Fraser Valley encompasses two CDs (Table 2) that were considered in the development of the impact scale.

Table 2 Census Divisions in the Canadian Lower Fraser Valley

Census Division	AQBAT Census Division ID
Metro Vancouver (MV)	418
Fraser Valley Regional District (FVRD)	417

AQBAT has a total of 18 health endpoints (or outcomes) associated with ambient concentrations of CACs. While AQBAT is capable of Monte Carlo simulation to determine uncertainty in overall health benefits valuation, this was not completed for derivation of the impact scale. Instead, the central value was extracted to develop the impact scale. The central value represents the health benefits resulting if mean input parameters are used in the AQBAT equations deterministically, and using the annual average concentration metric which, for AQBAT, is either the daily 1h maximum concentration (1-h daily max) or daily average concentration.

Therefore, either the annual average of the daily 1h maximum or annual average concentration is required. Table 3 summarizes AQBAT health endpoints and corresponding air contaminant and concentration metrics.

Table 3 AQBAT Input Concentration Metrics by Air Contaminant and Endpoint

Health Endpoint	Input Concentration Metric by Air Contaminant	
	Annual Average	Annual Average of Daily 1h Max
Acute Exposure Mortality	CO (ppm) NO_2 (ppb) SO_2 (ppb)	O_3 (ppb)
Acute Respiratory Symptom Days	$PM_{2.5}$ (µg/m³)	O_3 (May-Sep) (ppb)
Adult Chronic Bronchitis Cases	$PM_{2.5}$ (µg/m³)	—
Asthma Symptom Days	$PM_{2.5}$ (µg/m³)	—
Cardiac Emergency Room Visits	$PM_{2.5}$ (µg/m³)	—
Cardiac Hospital Admissions	$PM_{2.5}$ (µg/m³)	—
Child Acute Bronchitis Episodes	$PM_{2.5}$ (µg/m³)	—
Chronic Exposure Cerebrovascular Mortality	$PM_{2.5}$ (µg/m³)	—
Chronic Exposure Ischemic Heart Disease Mortality	$PM_{2.5}$ (µg/m³)	—
Chronic Exposure Lung Cancer Mortality	$PM_{2.5}$ (µg/m³)	—
Chronic Exposure Respiratory Mortality	$PM_{2.5}$ (µg/m³)	O_3 (May-Sep) (ppb)
Elderly Cardiac Hospital Admissions	—	CO (ppm)
Minor Restricted Activity Days	—	O_3 (May-Sep) (ppb)
Respiratory Emergency Room Visits	$PM_{2.5}$ (µg/m³)	O_3 (May-Sep) (ppb)
Respiratory Hospital Admissions	$PM_{2.5}$ (µg/m³)	O_3 (May-Sep) (ppb)
Restricted Activity Days	$PM_{2.5}$ (µg/m³)	—

The valuations resulting from AQBAT in 2010 dollars were converted to 2017 currency using the consumer price index (CPI) since the 2018 CPI was not available. Thus, all valuations are presented in 2017 Canadian dollars.

The valuation of each endpoint is based on the mean or central value as listed in AQBAT and is summarized in Table 4 . In AQBAT Version 2a there is no valuation for Cardiac Hospital Admissions and Respiratory Hospital Admissions. The valuations are based on social welfare, that is, they attempt to measure the impact on overall quality of life, not just the costs of medical care. Therefore, these valuation estimates include not only the costs of medication and the cost of medical treatment, but also the value of lost productivity, lost income due to illness, or an equivalent loss of "productivity" for students who miss school or for care givers that take care of sick children, and an estimate of the value of the pain and suffering of the illness. The valuation for mortality dominates the health effects valuation. For mortality, AQBAT uses the value of a statistical life (VSL). This is not a dollar value on individual lives. Rather, the estimate is generated from studies of how much people are willing to pay for small reductions in their risks of dying from adverse health conditions that may be caused by environmental pollution. This approach is also used by US EPA (e.g., US EPA, 2000).[1]

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[1] United States, Environmental Protection Agency, Guidelines for Preparing Economic Analyses, September 2000

Table 4 AQBAT Valuation per Health Outcome

Health Endpoint	Valuation per Outcome (2017 CAD)
Acute Exposure Mortality	7,612,582
Chronic Exposure Cerebrovascular Mortality	7,612,582
Chronic Exposure Ischemic Heart Disease Mortality	7,612,582
Chronic Exposure Lung Cancer Mortality	7,612,582
Chronic Exposure Respiratory Mortality	7,612,582
Adult Chronic Bronchitis Cases	390,036
Elderly Cardiac Hospital Admissions	7,505
Cardiac Emergency Room Visits	6,350
Respiratory Emergency Room Visits	2,886
Child Acute Bronchitis Episodes	455
Asthma Symptom Days	75
Restricted Activity Days	69
Minor Restricted Activity Days	32
Acute Respiratory Symptom Days	19
Cardiac Hospital Admissions	n/a
Respiratory Hospital Admissions	n/a

AQBAT was run by specifying changes to the annual average concentration metrics for each census division using a population projection scenario for the year 2018 that matches the 2016 Census closest[2]. AQBAT provided the annual valuation directly for the difference in the annual average concentration metric. The calculation of the number of health outcomes associated with a change in concentration and the appropriate valuation of each endpoint was calculated by AQBAT, accounting for the base rates of each endpoint and population in the Metro Vancouver and FVRD census divisions individually. The annual valuation for each endpoint and air contaminant was divided by the concentration difference to produce a valuation for a unit change in each associated metric. It was assumed in developing the health impact scale that the valuations are linear over a range of concentration differences for these values. The resulting valuations of health benefits associated with reductions in ambient air concentrations of air contaminants are given in Table 5 for both Metro Vancouver and FVRD. The valuations estimated for FVRD are much smaller than those for Metro Vancouver because of the small population size.

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[2] AQBAT Population Projection Scenario 1 has populations of 309,752 and 2,548,689 for FVRD and MV respectively.

Table 5 AQBAT Valuation per unit reduction in ambient concentration of Criteria Air Contaminants

AQBAT Pollutant	Annual Average Concentration Metric	Annual Valuation (2017 CAD) per Unit Reduction in Annual Average Concentration Metric		
		Metro Vancouver	FVRD	Units
CO	1h daily max CO (ppm)	3,586,640	494,060	(\$/yr) / (ppm CO)
CO	24 h CO (ppm)	292,561,156	38,914,658	(\$/yr) / (ppm CO)
NO_2	24 h NO_2 (ppb)	115,188,340	15,321,633	(\$/yr) / (ppb NO_2)
O_3	1h daily max O_3 (ppb)	129,196,017	17,184,847	(\$/yr / (ppb O_3)
O_3	1h daily max O_3 (May-Sep) (ppb)	57,815,261	7,785,684	(\$/yr) / (ppb O_3 (May-Sep))
$PM_{2.5}$	24 h $PM_{2.5}$ ($\mu g/m^3$)	1,158,407,202	153,437,355	(\$/yr) / ($\mu g/m^3$ $PM_{2.5}$)
SO_2	24 h SO_2 (ppb)	70,708,570	9,405,212	(\$/yr) / (ppb SO_2)

2.1.2 Air Quality and Annual Emissions

In order to develop an impact scale in the units of dollars per tonne of air contaminant emitted, a relationship between emissions and ambient air quality is required. The scope of this project did not support detailed air quality dispersion modelling and photochemical modelling studies. Instead, the relationship was based on existing information and studies related to historical emission inventory and ambient air quality in the Canadian LFV and other jurisdictions. As a result, several simplifying assumptions were applied to provide a means of estimating the air quality changes associated with emission reduction.

Relationships between annual emissions in the Canadian LFV and ambient air concentrations were developed using linear regression with annual emission inventory data and ambient monitoring data for the period 1995 to 2015.

The study relied on emission inventories and ambient air quality data documented by Metro Vancouver. Emission inventories for the Canadian LFV were available from 1995 to 2015 at 5 year intervals (i.e. 1995, 2000, 2005, 2010 and 2015).[3] The data gaps between the emission inventory years were filled using the trends observed in the British Columbia emission inventories, which are available annually through the same period.[4] It was assumed that emissions in the Canadian LFV inventory will follow the same trend as changes in the BC inventory. Table 6 presents the emission inventories after data gaps have been filled.

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[3] John Lindner. Personal Communication, Metro Vancouver. 2018

[4] National Pollutant Release Inventory online search. Last accessed Dec 2018 https://pollution-waste.canada.ca/air-emission-inventory/?GoCTemplateCulture=en-CA

Table 6 Annual Emissions in the Canadian LFV (tonnes/year)

Year	CO	NO_x	$PM_{2.5}$	SO_x	VOC
1995	**504,550**	**94,983**	**10,496**	**11,181**	**102,290**
1996	444,875	85,982	12,608	8,759	99,347
1997	430,279	88,285	11,787	9,128	97,909
1998	425,688	89,595	10,766	9,131	96,897
1999	383,621	90,959	10,020	9,283	91,392
2000	**365,002**	**90,244**	**9,503**	**9,408**	**87,145**
2001	383,494	77,635	9,332	8,173	90,872
2002	359,615	79,256	8,747	7,366	88,561
2003	346,628	73,831	8,699	7,328	87,259
2004	341,541	76,167	8,734	7,718	85,839
2005	**285,384**	**71,508**	**8,511**	**7,269**	**77,381**
2006	265,098	66,189	7,092	6,739	79,503
2007	253,520	65,597	7,249	6,650	76,659
2008	230,961	63,456	7,176	6,679	72,723
2009	225,249	58,759	6,856	6,604	68,889
2010	**211,909**	**58,898**	**6,978**	**6,669**	**68,421**
2011	210,219	49,100	5,507	2,420	67,078
2012	217,801	49,484	6,064	2,479	66,206
2013	199,242	50,936	5,659	2,582	66,880
2014	199,562	51,116	5,616	2,579	65,440
2015	**193,007**	**49,807**	**5,267**	**1,619**	**65,408**

Ambient air concentrations for Metro Vancouver and FVRD were based on monitored data for the period 1995 to 2015. For each CD, ambient air concentrations were considered to be uniform across the CD. The concentration for each CD was assumed to be the average concentration reported by all monitoring stations within the CD.[3]

The Tapered Element Oscillating Microbalance (TEOM) was used to measure ambient $PM_{2.5}$ hourly concentrations from 1996 to 2012. Starting in 2013, ambient $PM_{2.5}$ concentrations were measured using the Federal Equivalent Method (FEM) Synchronized Hybrid Ambient Real-time Particulate (SHARP). These two methods yield different results. In order to use the data in a multi-year regression, consistent measurements are needed, so the 2013 to 2015 SHARP data was adjusted to TEOM equivalent $PM_{2.5}$ values. A recent study[5] completed by Ontario Ministry of the Environment and Climate Change (Ontario MOECC) and Environment and Climate Change Canada (ECCC) developed a simplified approach to correct historical TEOM data for reporting long-term $PM_{2.5}$ trends in Ontario. These results were assumed to be valid for the Canadian LFV, and were applied to 2013-2015 SHARP $PM_{2.5}$ measurement data. The following relationship was used:

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[5] Yushan Su, Uwayemi Sofowote,, Jerzy Debosz, Luc White and Anthony Munoz. Multi-Year Continuous PM2.5 Measurements with the Federal Equivalent Method SHARP 5030 and Comparisons to Filter-Based and TEOM Measurements in Ontario, Canada. Atmosphere 2018, 9, 191

- Warm months (May to September): $[\text{Concentration}]_{\text{TEOM}} = [\text{Concentration}]_{\text{SHARP}} / 1.08$
- Cold months (October to April): $[\text{Concentration}]_{\text{TEOM}} = ([\text{Concentration}]_{\text{SHARP}} - 1.09)/1.34$

Metro Vancouver conducted a separate comparison using ambient monitoring data from TEOM and SHARP units co-located at monitoring stations in Metro Vancouver; this comparison produced similar relationships using an annual basis.[3]

The average ambient concentrations for each air contaminant with health outcomes predicted by AQBAT are provided in Appendix A for both Metro Vancouver and FVRD for the period 1995 to 2015. For $PM_{2.5}$, ambient concentrations for 2013 to 2015 in the Appendix have been adjusted to reflect TEOM type measurement.

Linear regression was used to establish relationships between average ambient concentration in each CD (the dependent variable), and emissions of air contaminants in the Canadian LFV (the independent variable). Variables used in the regressions are given in Table 7. The slope of the resulting linear relationship represents the change in ambient concentration resulting from a unit change in emissions.

Ambient air concentrations of air contaminants in each CD were assumed to be a function of total emissions from both CDs rather than of the individual CDs. Emissions from Metro Vancouver are generally an order of magnitude higher than those of the Fraser Valley and, given the proximity, Fraser Valley concentrations cannot be considered to be independent of Metro Vancouver emissions.

Table 7 Regression Variables

Dependent Variable	Independent Variable	R^2 for MV	R^2 for FVRD
CO concentration	CO emission	0.92	0.62
NO_2 concentration	NO_X emission	0.84	0.63
$PM_{2.5}$ concentration	$PM_{2.5}$ emission	0.58	
SO_2 concentration	SO_2 emission	0.81	0.71
$(O_3 + NO_X)$ concentrations	$(NO_X + VOC)$ emissions	0.93	0.75
NO_X concentration	$(O_3 + NO_X)$ concentrations	0.65	0.54

Appendix A includes the emissions and concentration data that were used in each regression analysis. Refer to following tables and figures for the data and regression analysis:

- Table A 1 CO emissions and annual 24-hour average ambient CO concentrations
- Table A 2 SOx emissions and annual 24-hour average ambient SO_2 concentrations
- Table A 3 NOx emissions and annual average 24-hour ambient NO_2 concentrations
- Table A 4 $PM_{2.5}$ emissions and annual 24-hour average ambient $PM_{2.5}$ concentrations
- Table A 5 NOx and VOC emissions and annual average ambient 1-hour max O_3 + 24-hour NOx concentrations
- Figure A-1: CO emissions vs. annual 24-hour average ambient CO concentrations in MV
- Figure A-2 CO emissions vs. annual 24-hour average ambient CO concentrations in FVRD

- Figure A-3 SOx emissions vs. annual 24-hour average ambient SO_2 concentrations in MV
- Figure A-4 SOx emissions vs. annual 24-hour average ambient SO_2 concentrations in FVRD
- Figure A-5 NOx emissions vs. annual 24-hour average ambient NO_2 concentrations in MV
- Figure A-6 NOx emissions vs. annual 24-hour average ambient NO_2 concentrations in FVRD
- Figure A-7 $PM_{2.5}$ emissions vs. annual 24-hour average ambient $PM_{2.5}$ concentrations in Canadian LFV (MV and FVRD)
- Figure A-8 VOC + NOx emissions vs. annual average ambient 1-hour max O_3 + 24-hour NOx concentrations in MV
- Figure A-9 VOC + NOx emissions vs. annual average ambient 1-hour max O_3 + 24-hour NOx concentrations in FVRD
- Figure A-10 Annual average ambient 24-hour NOx concentrations vs. annual average ambient 1-hour max O_3 + 24-hour NOx concentrations in MV
- Figure A-11 Annual average ambient 24-hour NOx concentrations vs. annual average ambient 1-hour max O_3 + 24-hour NOx concentrations in FVRD

For each air contaminant, the relationship is plotted with resulting equation and R-squared value for the fit. The slope of the line represents the change of concentration for a unit change in emissions. R-square (coefficient of multiple determination for multiple regression) is a statistical measure to show how close the data are to the fitted regression line. 0% indicates that the model explains none of the variability of the response data around its mean while 100% indicates that the model explains all the variability of the response data around its mean. We aimed for R^2 greater than 0.5. In the case of $PM_{2.5}$ emissions and concentrations analysis, we could not obtain $R^2 > 0.5$ for FVRD, therefore combined Metro Vancouver and FVRD data was considered. Concentrations of CO and SO_2 result entirely from emissions of these air contaminants. The impact of background concentrations (i.e. resulting from emissions outside of the Canadian LFV) or atmospheric reactions creating or destroying these air contaminants were assumed to be negligible. Therefore, fits were forced through a Y-intercept of zero.

For CO, AQBAT requires both the annual average concentration and the annual average of the maximum daily 1-hour concentration. The available monitoring data did not include the latter metric. Therefore, it was assumed that both metrics would change equally with a change in emissions. That is, the same slope was assumed to apply for both.

Ambient ozone concentrations can be controlled only by controlling emissions of its precursors, such as VOC and oxides of nitrogen (NOx). Nitric oxide (NO) is both a precursor of O_3 and a scavenger of O_3. NO reacts with O_3 and removes O_3 from the air. However, nitrogen dioxide (NO_2) is formed in this process which is a direct precursor of O_3. This chemical transformation takes place while these air contaminants are transported by the winds, and continuously changing composition of O_3, NO and NO_2. Therefore, concentration of NO_2 is dependent on the emissions of NO_X and atmospheric reactions with O_3 that affect the equilibrium concentrations of NO_2 and NO.

NO_2 concentrations estimated as a function on NO_X emissions showed good agreement, therefore NO_2 concentrations in this study were estimated as a function of NOx emissions.

Ambient concentrations of $PM_{2.5}$ result from emissions of $PM_{2.5}$ and secondary formation of $PM_{2.5}$ from precursors such as VOC, NO_X, SO_X and ammonia (NH_3). For this study, insufficient information was available to estimate concentration of secondary $PM_{2.5}$ due to emissions of precursors, and $PM_{2.5}$ concentration was treated as a function of $PM_{2.5}$ emissions only. The regression resulted in a Y-intercept of 2.75 µg/m³, which suggests background concentration and secondary formation of $PM_{2.5}$ is about 2.75 µg/m³. Average background concentration in BC was reported to be ~2 µg/m³ in 2005[6] which seems consistent with the regression.

Concentration of $PM_{2.5}$ due to secondary formation is dependent on photochemical reactions and, like background concentration, will be relatively constant over the Canadian LFV and should be similar for both CDs. Using the same Y-intercept for both CDs did not result in an acceptable fit for FVRD (i.e. $R^2<0.5$). Therefore, a single regression was used that included data for both Metro Vancouver and FVRD.

Ozone concentration is dependent on complex atmospheric reactions between precursors, mainly NO_X and VOCs. For this study, a simplified approach was used to estimate the change in ozone concentration for any change in total precursor emissions (i.e. VOC and NO_X). The relationship was established in three steps:

1. Regression analysis between the sum of O_3 and NO_X ambient concentrations (O_3 + NO + NO_2) vs the sum of O_3 precursors NO_X and VOC emissions (NO_X + VOC). As discussed above, chemical transformation and composition of O_3 and NOx is continuously changing, and there is an inverse relationship between O_3 and local NOx concentrations. Therefore, first we determined O_3 and NOx concentrations present in the atmosphere for VOC and NOx emissions (i.e. O_3 precursors). See Appendix figures A-8 and A-9;
2. Regression analysis between NO_X ambient concentrations vs. the sum of O_3 and NO_X ambient concentrations (O_3 + NO_X) provided NOx concentration portions with a reasonable fit. See Appendix figures A-10 and A-11; and
3. Combine steps 1 and 2 to establish a direct relationship between O_3 concentrations and NOx + VOC emissions.

After establishing a direct relationship between O_3 formation and O_3 precursors emissions from statistical models, we used information derived from comprehensive photochemistry modelling studies in the Canadian LFV. For example, a study on ozone formation in the Lower Fraser Valley[7] found that the western part of the region is consistently VOC-limited while the eastern region is transitional. Based on this study it was assumed that the change in O_3 formation on an annual basis will be primarily driven by VOC emissions in Metro Vancouver and by both NOx and VOC emissions in FVRD.

Table 8 provides the relationships between change in emissions and ambient concentrations developed in this study.

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[6] Background Concentrations of PM2.5 and Ozone in British Columbia, Canada. Prepared by Ian G. McKendry, Geography/Atmospheric Science, The University of British Columbia. Prepared for the British Columbia Ministry of Environment March 2006. Last accessed December 2018 https://www2.gov.bc.ca/assets/gov/environment/air-land-water/air/reports-pub/background_pm25_ozone.pdf

[7] Regional Ground-Level Ozone Strategy for the Canadian Lower Fraser Valley Region, April 2014. Last accessed December 2018 http://www.metrovancouver.org/services/air-quality/AirQualityPublications/RGLOS2014.pdf

Table 8 Change in ambient CAC concentrations for a change in emissions

AQBAT Air contaminant	CD	Averaging Period (hours)	Change in Annual Average Concentration Metric per Change in Annual Emissions	
CO	MV	1	1.34E-06	ppm CO / (tonnes CO/yr)
CO	MV	24	1.34E-06	ppm CO / (tonnes CO/yr)
NO_2	MV	24	2.04E-04	ppb NO_2 / (tonnes NO_X/yr)
O_3	MV	1	0	ppb O_3 / (tonnes NO_X/yr)
O_3	MV	1	2.46E-04	ppb O_3 / (tonnes VOC/yr)
O_3 (May-Sep)	MV	1	0	ppb O_3 (May-Sep) / (tonnes NOX/yr)
O_3 (May-Sep)	MV	1	2.46E-04	ppb O_3 (May-Sep) / (tonnes VOC/yr)
$PM_{2.5}$	MV	24	2.72E-04	($\mu g/m^3$ $PM_{2.5}$) / (tonnes $PM_{2.5}$/yr)
SO_2	MV	24	2.72E-04	ppb SO_2 / (tonnes SO_X/yr)
CO	FVRD	1	9.08E-07	ppm CO / (tonnes CO/yr)
CO	FVRD	24	9.08E-07	ppm CO / (tonnes CO/yr)
NO_2	FVRD	24	1.45E-04	ppb NO_2 / (tonnes NO_X/yr)
O_3	FVRD	1	1.84E-04	ppb O_3 / (tonnes NO_X/yr)
O_3	FVRD	1	2.38E-04	ppb O_3 / (tonnes VOC/yr)
O_3 (May-Sep)	FVRD	1	1.84E-04	ppb O_3 (May-Sep) / (tonnes NO_X/yr)
O_3 (May-Sep)	FVRD	1	2.38E-04	ppb O_3 (May-Sep) / (tonnes VOC/yr)
$PM_{2.5}$	FVRD	24	2.72E-04	($\mu g/m^3$ $PM_{2.5}$) / (tonnes $PM_{2.5}$/yr)
SO_2	FVRD	24	6.50E-05	ppb SO_2 / (tonnes SO_X/yr)

2.2 Hazardous Air Pollutants

2.2.1 Health Benefits Valuation of Air Quality Reductions

AQBAT version 2a does not include hazardous air pollutants (HAPs). Therefore, a methodology for quantifying the health benefits associated with improved air quality from reductions in HAP concentrations was developed using data compiled for a previous risk assessment that was completed for Metro Vancouver (Sonoma, 2015).[8]

The following HAPs were selected for inclusion in the analyses based on Metro Vancouver priorities and information availability: 1,3-butadiene, benzene, carbon tetrachloride, chromium VI, diesel particulate matter (PM), and formaldehyde. These HAPs have been associated with some of the highest cancer risks in past risk assessments for the Metro Vancouver area (Sonoma, 2015; Levelton, 2007)[8,9].

The analysis focused on non-fatal cancer outcomes for the HAPs, as all the HAPs are primarily associated with cancer at lower air concentrations, this health outcome is also more easily quantified, and it is associated with the highest benefits (i.e., higher social welfare costs are associated with the treatment of non-fatal cancer). Other health outcomes such as neurological, reproductive, developmental may be associated with some of these HAPs, but the valuation for these health outcomes is not easily quantifiable. Fatal lung cancer impacts are evaluated for the CACs.

Unit risk factors associated with developing cancer for each target HAP were taken from the Sonoma (2015) study. The unit risk factor is the probability of developing cancer at an exposure of 1 $\mu g/m^3$ over a 70-year lifetime. Unit risk factors from Health Canada were used if available because they were considered most applicable in a Canadian context. Factors from the California Office of Environmental Health Hazard Assessment (OEHHA) were used for HAPs with no Health Canada factor.

Unit risk factors used in the development of the impact scale are shown in Table 9.

Table 9 Cancer Unit Risk Factors

Hazardous Air Pollutant	Unit Risk[a] $(\mu g/m^3)^{-1}$	Reference
1,3-Butadiene	0.00017	OEHHA
Benzene	0.0000033	Health Canada
Carbon tetrachloride	0.000042	OEHHA
Chromium VI	0.076	Health Canada
Diesel PM	0.0003	OEHHA
Formaldehyde	0.000006	OEHHA

Notes: (a) Unit risks are the individual lifetime (70 yr) cancer risk per 1 $\mu g/m^3$ exposure

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[8] Toxic Air Pollutants Risk Assessment. Prepared by Sonoma Technology, Inc, Prepared for Metro Vancouver 2015.

[9] Air Toxics Emission Inventory and Health Risk Assessment – Technical Appendix. Prepared by Levelton Consultants Ltd., Prepared for Greater Vancouver Regional District and Environment Canada 2007.

A valuation of $4,949,041 per cancer case, expressed in 2017 Canadian dollars, was used in the analysis. This value was derived from the valuation for non-fatal cancers used in the Bay Area Air Quality Management District Multi-Pollutant Evaluation Method (BAAQMD, 2016) and was obtained from a review by McCubbin and Delucchi (1996)[10]. Like the AQBAT valuation, this value includes all welfare costs of cancer, such as medical costs, pain and suffering to both patients and friends, and the loss of production to society.

The BAAQMD value of $3.7M (2015 USD) per cancer case was adjusted to 2017 CAD using the average 2015 exchange rate and the Canadian consumer price index.

The annual social welfare costs associated with a 1 µg/m³ change in HAPs concentrations were calculated by applying the unit risk factors to the total population in Metro Vancouver and the FVRD respectively and dividing the result by 70 years to get an annual cost.

$$Annual\ Valuation\ per\ \ \mu g/m^3 = Cost\ per\ cancer\ case \times Unit\ Risk\ (\mu g/m^3)^{-1} \times Total\ Population\ in\ CD\ \div 70yr$$

For consistency with the analysis for CACs, the populations for Metro Vancouver and FVRD used in the HAPs calculations were taken from AQBAT. An example calculation is provided for benzene for the Metro Vancouver census division:

$$Annual\ Valuation\ for\ benzene\ in\ MV = \$4{,}949{,}041 \times 0.0000033\ (\mu g/m^3)^{-1} \times\ 2{,}548{,}689 \div 70yr$$
$$Annual\ Valuation\ for\ benzene\ in\ MV = 594{,}639\ (\$/yr)\ per\ (\mu g/m^3)$$

The annual valuation per 1 µg/m³ reduction in long-term average HAPs concentrations for Metro Vancouver and FVRD associated with cancer, calculated using this equation, are shown in Table 10 .

Table 10 Valuation of Health Benefits Associated with Reduction in Ambient Concentrations

Hazardous Air Pollutant	Annual Health Benefits Valuation (2017 CAD) per µg/m³	
	Metro Vancouver	FVRD
1,3-Butadiene	$30,632,944	$3,722,940
Benzene	$594,639	$72,269
Carbon tetrachloride	$7,568,139	$919,785
Chromium VI	$13,694,727,820	$1,664,373,068
Diesel PM	$54,058,136	$6,569,894
Formaldehyde	$1,081,163	$131,398

2.2.2 Air Quality and Annual Emissions

The relationship between ambient concentrations and annual emissions in the Canadian Lower Fraser Valley for HAPs was derived using ambient concentration data and the HAP emission inventory.

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[10] McCubbin DR, and MA Delucchi, *The social cost of the health effects of motor vehicle air pollution*, Report #11 in the series: The annualized social cost of motor vehicle use in the United States, based on 1990-1991 data, Institute of Transportation Studies, University of California, Davis, August 1996

Long-term average concentrations of HAPs in the Canadian LFV were documented for 2010 in a previous study for Metro Vancouver, "Toxics Air Pollutants Risk Assessment" (Sonoma, 2015). Toxic air pollutant is an equivalent term for HAP. Data from individual stations were averaged to produce an average concentration for the Metro Vancouver and FVRD census divisions, respectively, and the results are shown in Table 11. Concentration estimates for hexavalent chromium and diesel particulate matter were developed in the 2015 Sonoma Study as they were not measured directly by the monitoring network.

Table 11 Long-term average concentration (2010) for Hazardous Air Pollutants

Hazardous Air Pollutant	Long-term Average Concentration (µg/m³)	
	Metro Vancouver	FVRD
1,3-Butadiene	0.08775	0.05116
Benzene	0.82119	0.47010
Carbon tetrachloride	0.55240	0.55012
Chromium VI	0.00004	0.00002
Diesel PM	0.77750	0.65000
Formaldehyde	2.19748	1.56929

The 2010 emission inventory for HAPs in the Canadian LFV, developed and documented in the same study (Sonoma, 2015), was used as this represented the most recent emission inventory for HAPs in the Canadian LFV (Table 12).

Table 12 Hazardous Air Pollutant Emission Inventory (2010) for Canadian Lower Fraser Valley

Hazardous Air Pollutant	Annual Emissions (tonnes/year)
1,3-Butadiene	104
Benzene	832
Carbon tetrachloride	2.8
Chromium VI	0.04172
Diesel PM	1451
Formaldehyde	512

Since only a single year of data was available, a regression analysis as was done for CACs could not be completed. Instead, background concentrations of the HAPs were taken into account by using remote concentration estimates (RCE) in the Northern Hemisphere for relevant HAPs from the Technical Support Document for EPA's 2011 National-scale Air Toxics Assessment[11]. These values represent an estimate of the background concentration of HAPs in the Northern Hemisphere in 2011 and for the purposes of this study were assumed to be applicable to the Canadian LFV in 2010. The RCE values are given in Table 13 .

[11] Table 7, 2011 NATA TSD, Accessed Nov 2018, https://digital.library.unt.edu/ark:/67531/metadc949443/m2/1/high_res_d/2011-nata-tsd.pdf

Table 13 2011 Remote Concentration Estimate for Northern Hemisphere

Hazardous Air Pollutant	Long-term average background concentration (μg/m³)
1,3-Butadiene	2.00E-03
Benzene	0.12
Carbon tetrachloride	0.55
Chromium VI	1.50E-06
Diesel PM	No background
Formaldehyde	0.43

The RCE was subtracted from the long-term average concentrations presented above and the difference was assumed to be the ambient concentration attributable to the emissions in the Canadian LFV. The change in long-term average concentration per change in Canadian LFV annual emissions was derived by then dividing this result for each census division by the annual emissions from Table 12 . The results are presented in Table 14 .

Table 14 Derived concentration change per change in annual emissions

Hazardous Air Pollutant	Concentration change per change in annual emissions (μg/m³) / (tonne/year)	
	MV	FVRD
1,3-Butadiene	8.24E-04	4.73E-04
Benzene	8.43E-04	4.21E-04
Carbon tetrachloride	8.55E-04	4.32E-05
Chromium VI	9.74E-04	4.88E-04
Diesel PM	2.70E-04	2.70E-04
Formaldehyde	3.45E-03	2.23E-03

2.3 Health Impact Scale

The relationships developed in the preceding sections were combined to provide the health impact scale.

The valuation ($/tonne) for contaminant emitted resulting ambient air contaminant, and CD are presented in Table 15 . The valuations associated with emissions of each air contaminant for the two CDs are summed to define the health impact scale valuation for the Canadian LFV in Table 16 .

Table 15 Summary of Valuations by Ambient Air Contaminant

CD	Change in Annual Average Concentration		Health Benefits Valuation ($/yr) per Change		Valuation ($/tonne)	
	Value	Units	Value	Units	Value	Units
MV	1.34E-06	ppm CO / (tonnes CO/yr)	296,147,796	($/year)/(ppm CO)	397	$/tonne CO
	2.04E-04	ppb NO2 / (tonnes NOX/yr)	115,188,340	($/year)/(ppb NO2)	23,498	$/ tonne NOx
	-	ppb O3 / (tonnes NOX/yr)	129,196,017	($/year)/(ppb O3)	-	$/ tonne NOx
	2.46E-04	ppb O3 / (tonnes VOC/yr)	129,196,017	($/year)/(ppb O3)	31,790	$/tonne VOC
	-	ppb O3 (May-Sep) / (tonnes NOX/yr)	57,815,261	($/year)/(ppb O3 (May-Sep))	-	$/ tonne NOx
	2.46E-04	ppb O3 (May-Sep) / (tonnes VOC/yr)	57,815,261	($/year)/(ppb O3 (May-Sep))	14,226	$/tonne VOC
	2.72E-04	(µg/m^3 PM2.5) / (tonnes PM2.5/yr)	1,158,407,202	($/year)/(µg/m^3 PM2.5)	315,087	$/tonne PM2.5
	2.72E-04	ppb SO2 / (tonnes SOX/yr)	70,708,570	($/year)/(ppb SO2)	19,233	$/tonne SOX
	8.24E-04	(µg/m^3) / (tonne/yr)	30,632,944	($/year)/(µg/m^3)	25,256	$/ tonne 1,3-Butadiene
	8.43E-04	(µg/m^3) / (tonne/yr)	594,639	($/year)/(µg/m^3)	501	$/ tonne Benzene
	8.55E-04	(µg/m^3) / (tonne/yr)	7,568,139	($/year)/(µg/m^3)	6,474	$/ tonne Carbon tetrachloride
	9.74E-04	(µg/m^3) / (tonne/yr)	13,694,727,820	($/year)/(µg/m^3)	13,340,400	$/ tonne ChromiumVI
	2.70E-04	(µg/m^3) / (tonne/yr)	54,058,136	($/year)/(µg/m^3)	14,596	$/ tonne Diesel PM
	3.45E-03	(µg/m^3) / (tonne/yr)	1,081,163	($/year)/(µg/m^3)	3,732	$/ tonne Formaldehyde
FVRD	9.08E-07	ppm CO / (tonnes CO/yr)	39,408,718	($/year)/(ppm CO)	36	$/tonne CO
	1.45E-04	ppb NO2 / (tonnes NOX/yr)	15,321,633	($/year)/(ppb NO2)	2,222	$/ tonne NOx
	1.84E-04	ppb O3 / (tonnes NOX/yr)	17,184,847	($/year)/(ppb O3)	3,168	$/ tonne NOx
	2.38E-04	ppb O3 / (tonnes VOC/yr)	17,184,847	($/year)/(ppb O3)	4,095	$/tonne VOC
	1.84E-04	ppb O3 (May-Sep) / (tonnes NOX/yr)	7,785,684	($/year)/(ppb O3 (May-Sep))	1,435	$/ tonne NOx
	2.38E-04	ppb O3 (May-Sep) / (tonnes VOC/yr)	7,785,684	($/year)/(ppb O3 (May-Sep))	1,855	$/tonne VOC
	2.72E-04	(µg/m^3 PM2.5) / (tonnes PM2.5/yr)	153,437,355	($/year)/(µg/m^3 PM2.5)	41,735	$/tonne PM2.5
	6.50E-05	ppb SO2 / (tonnes SOX/yr)	9,405,212	($/year)/(ppb SO2)	611	$/tonne SOX
	4.73E-04	(µg/m^3) / (tonne/yr)	$3,722,940	($/year)/(µg/m^3)	1,760	$/ tonne 1,3-Butadiene
	4.21E-04	(µg/m^3) / (tonne/yr)	$72,269	($/year)/(µg/m^3)	30	$/ tonne Benzene
	4.32E-05	(µg/m^3) / (tonne/yr)	$919,785	($/year)/(µg/m^3)	40	$/ tonne Carbon tetrachloride
	4.88E-04	(µg/m^3) / (tonne/yr)	$1,664,373,068	($/year)/(µg/m^3)	812,600	$/ tonne ChromiumVI
	2.70E-04	(µg/m^3) / (tonne/yr)	$6,569,894	($/year)/(µg/m^3)	1,774	$/ tonne Diesel PM
	2.23E-03	(µg/m^3) / (tonne/yr)	$131,398	($/year)/(µg/m^3)	292	$/ tonne Formaldehyde

Table 16 Health Impact Scale by Air Contaminant Emitted

Air Contaminant	Air Contaminant Type	Health Impact Scale	
		Valuation ($/tonne)	Relative Scale
Chromium VI	HAP	$14,153,000	32,714.5
PM$_{2.5}$	CAC	$356,822	824.8
VOC	CAC	$51,967	120.1
NOx	CAC	$30,324	70.1
1,3-Butadiene	HAP	$27,016	62.4
SOx	CAC	$19,844	45.9
Diesel PM	HAP	$16,370	37.8
Carbon tetrachloride	HAP	$6,514	15.1
Formaldehyde	HAP	$4,025	9.3
Benzene	HAP	$532	1.2
CO	CAC	$433	1.0

It may be useful at times to rank actions using the scale, but without presenting dollar valuations. For this purpose, a relative scale is also provided in the table. For the relative scale, a value of 1 is assigned to the lowest valuation, and all others are assigned proportionally. The use of the relative scale will result in the same ranking as the valuation.

It is important to understand that the impact of any action is the product of the impact scale and the reduction in emissions associated with action. While it may appear that the scale is dominated by Chromium VI, total emissions of chromium VI were less than 42kg in 2010, and potential reductions of emissions are expected to be in the order of 1kg or less. In contrast, the potential reductions of other air contaminants are 3 to 6 orders of magnitude higher. The DPM valuation only included non-fatal cancer. The impact evaluation, however, would include both reductions in $PM_{2.5}$ and DPM.

The valuation estimates compare with some of the estimates and contrasts with others that are reported in the Bay Area Air Quality Management District Clean Air Plan 2017 “Spare the Air – Cool the Climate, A Blueprint for Clean Air and Climate Protection in the Bay Area.” For example, we estimated much higher NOx and VOC valuations and much lower valuations for benzene and PM (both direct and DPM), but similar estimates for SOx, 1,3-butadiene and formaldehyde. These differences are likely due to the different methodology used in producing these estimates compared to the methods used by BAAQMD.

The methods employed in this analysis are associated with a number of uncertainties, which primarily include:

- Assumptions regarding the association between emissions and concentrations, which are based on only a single point in time. A more rigorous modeling approach to derive the relationship between emissions and concentrations would reduce these uncertainties.
- Assumptions regarding the associations between air pollutants and health impacts, which are based primarily on observations epidemiological associations.
- Assumptions related to the valuation of health impacts, which can vary depending on the methodology for determining these values.

The valuations in this study were compared to a similar analysis completed by the BAAQMD in 2017 (Table 17), which produced valuations on the same order of magnitude, and also indicated that PM2.5 is the criteria contaminant with the most impact per tonne.

Table 17 Comparison between BAAQMD and Ramboll studies

Pollutant	Ramboll valuation ($CDN/tonne)	BAAQMD valuation ($US/ton)	BAAQMD valuation ($CDN/tonne)	Ratio Ramboll/ BAAQMD	Notes
CO	$ 433	-	-	-	-
NOx	$ 30,324	$6,000	$8,461	3.58	Same order of magnitude
$PM_{2.5}$	$ 356,822	$ 558,400	$ 787,424	0.45	Same order of magnitude
VOC	$ 51,967	$3,400	$4,794	10.84	Poor agreement - Differences may be due to ozone chemistry
SOx	$ 19,844	$ 18,700	$ 26,370	0.75	Same order of magnitude
1,3-butadiene	$ 27,016	$ 32,400	$ 45,689	0.59	Same order of magnitude
Benzene	$ 532	$9,200	$ 12,973	0.04	Poor agreement - different unit risks used[12]
Carbon tetrachloride	$6,514	-	-	-	-
Chromium 6	$14,153,000	-	-	-	-
DPM	$ 16,370	$4,200	$5,923	2.76	Same order of magnitude
Formaldehyde	$4,025	$1,400	$1,974	2.04	Same order of magnitude
Acetaldehyde		$ 600	$ 846	-	-

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[12] The benzene valuation in Scenario 5 of the Sensitivity Analysis (see Section 3) is $4671/tonne, based on the more conservative OEHHA unit risk

3. SENSITIVITY ANALYSIS OF IMPACT SCALE

3.1 Example Policies and Program Actions

Metro Vancouver provided ten (10) example actions that are included in the IAQGGMP or by other regulatory authorities in the region. Metro Vancouver estimated the annual emission reductions that would result from implementation of these actions, generally based on expected reductions and the 2015 emissions inventory. They were developed for demonstration purposes and do not necessarily represent expected Metro Vancouver policy directions. The health impact scale was applied to these example actions to illustrate how the scale would value and prioritize the actions. It is assumed that the actions would be prioritized based on ranking from highest to lowest valuation.

The list of actions and the associated reductions in annual emissions are provided in Table 18. The table includes the overall impact valuation and ranking after application of the health impact scale described in Section 2. The breakdown in valuation resulting from each air contaminant is provided in Table 19 .

Action 8, restriction on wood burning appliances, is ranked highest priority, with an annual valuation of about $438M based on the health impact scale. This action has been studied in detail, and results were documented in "Health and Economic Impacts of Residential Wood Burning in Metro Vancouver", Hemmera Envirochem Inc., 2017[13]. The study estimated annual cost benefits from emission reductions ranging from $282M to $869M depending on assumptions. The health impact scale valuation falls well within this range, suggesting reasonable agreement with the Hemmera methodology.

3.2 Sensitivity Analysis and Scenarios

An analysis was conducted to investigate the sensitivity of the overall impacts to changes in the impact scales of specific air contaminants. Metro Vancouver identified nine (9) alternate scenarios that each represent a change in the impact scale compared to the "Base Case" health impact scale documented in Section 2. The nine scenarios are described in Table 20.

The impact scales for the Base Case and the nine alternate scenarios are presented in Table 21. These alternate scales were applied to the example actions described in Table 18, and the resulting valuation and ranking of the actions are provided in Table 22 and Table 23.

3.3 Discussion of Sensitivity

Comparing the alternate scenarios to the Base Case in Table 22 and Table 23 leads to a number of notable observations.

For this particular set of example actions, the ranking of actions is not particularly sensitive to the changes in the scale covered by the alternate scenarios. Only 2 of the 9 scenarios resulted in any change to the ranking whatsoever. Scenario 3 (specified values for NO_X VOC, and $PM_{2.5}$) resulted only in a reversal of the 1st and 2nd ranked actions. Scenario 2 (zero valuation for

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[13] Health and Economic Impacts of Residential Wood Burning in Metro Vancouver, Prepared by Hemmera Envirochem Inc., Prepared for Metro Vancouver, 2017

mortality) had more effect on ranking, though it should be noted that the three actions ranked highest were unaffected, with minor rearrangement of the lower rankings.

It is notable that eliminating the impact of toxics, or HAPs, (Scenario 4) or doubling the impact of toxics (Scenario 8) did not affect the ranking of the actions. With the exception of Action 10, which affects Chromium 6 emissions only, eliminating or doubling the impact of toxics had negligible effect on the valuation of the action.

Comparison of Scenario 2 (zero valuation for a statistical life) to the Base Case shows that valuations are dominated by the value of a statistical life. For all actions other than Scenario 2, which affects Cr VI only, the value of mortalities represented more than 93% of the total impact. Nevertheless, as mentioned above, the value of mortalities had limited effect on ranking of actions.

The impact scale does not currently account for secondary formation of PM, so it is possible that the impact scale of precursors (NO_X, VOC, SO_2) may be different than presented in this study. Scenario 7 (25% increase in valuation of NO_X, VOC, and SO_2) was included to investigate the potential effect of higher impact scales for these precursors. This change did not affect the ranking of the 10 actions, though the valuation of the actions increased between 0 and 25%, depending on the air contaminants involved. This suggests that ignoring secondary formation in the scale has moderate to negligible effect on the impacts predicted.

Table 18 Example Actions and Associated Emission Reductions (tonnes/year)

Actions		Projected Emission Reductions (tonnes/year)											Overall Impact	
Action No.	Description of Action	CO	NOX	PM2.5	VOC	SOX	1,3-Butadiene	Benzene	Carbon tetrachloride	ChromiumVI	Diesel PM	Formaldehyde	Valuation	Rank
8	Restrictions on wood burning appliances	5,427	72.0	1,117	650	11.0	8.79	41.4	-	-	-	67.2	$ 438,000,000	1
10	Implementation of ECA	-	-	279	-	4,433	-	-	-	-	316	-	$ 193,000,000	2
9	Impact of Nonroad Diesel Engine Emissions Regulation	300	500	50.0	40.0	1.00	0.242	2.83	-	6.01E-05	50.0	0.713	$ 36,100,000	3
6	Zero emission vehicle mandate	3,380	185	5.71	248	9.45	1.31	8.09	-	2.92E-04	2.82	3.47	$ 22,300,000	4
4	Heavy duty vehicle inspection maintenance program	609	350	10.0	43.2	1.10	0.125	0.495	-	-	10.0	0.665	$ 16,900,000	5
5	Residential wood stove incentives	48.9	0.125	7.98	8.68	0.0749	0.117	0.553	-	-	-	0.898	$ 3,330,000	6
1	Apply California-style VOC limits for automotive refinishing products	-	-	-	50.0	-	-	-	-	-	-	-	$ 2,600,000	7
3	Voluntary program to replace gasoline lawn eqmt	136	2.32	0.484	10.0	0.00626	0.0606	0.708	-	5.82E-07	0.484	0.178	$ 832,000	8
7	Heavy duty vehicle licensing program	3.60	19.1	0.450	0.43	-	1.25E-03	4.93E-03	-	-	0.450	0.0299	$ 772,000	9
2	Phase out CrVI emissions from miscellaneous industrial sources	-	-	-	-	-	-	-	-	1.58E-03	-	-	$ 22,400	10

Table 19 Breakdown of Base Case Impact Valuations for Example Actions

Actions		Valuation ($) with Base Case Health Impact Scale											Overall Impact	
Action No.	Description of Action	CO	NOX	PM2.5	VOC	SOX	1,3-Butadiene	Benzene	Carbon tetrachloride	ChromiumVI	Diesel PM	Formaldehyde	Valuation	Rank
8	Restrictions on wood burning appliances	$ 2,347,835	$ 2,183,314	$ 398,569,861	$ 33,778,393	$ 218,285	$ 237,478	$ 21,983	$ -	$ -	$ -	$ 270,560	$ 438,000,000	1
10	Implementation of ECA	$ -	$ -	$ 99,660,711	$ -	$ 87,968,762	$ -	$ -	$ -	$ -	$ 5,178,750	$ -	$ 193,000,000	2
9	Impact of Nonroad Diesel Engine Emissions Regulation	$ 129,786	$ 15,161,905	$ 17,841,086	$ 2,078,670	$ 19,844	$ 6,550	$ 1,506	$ -	$ 851	$ 818,478	$ 2,871	$ 36,100,000	3
6	Zero emission vehicle mandate	$ 1,462,098	$ 5,605,508	$ 2,038,522	$ 12,905,529	$ 187,487	$ 35,354	$ 4,300	$ -	$ 4,133	$ 46,214	$ 13,959	$ 22,300,000	4
4	Heavy duty vehicle inspection maintenance program	$ 263,610	$ 10,613,334	$ 3,568,217	$ 2,242,548	$ 21,858	$ 3,389	$ 263	$ -	$ -	$ 163,696	$ 2,678	$ 16,900,000	5
5	Residential wood stove incentives	$ 21,138	$ 3,796	$ 2,849,052	$ 451,306	$ 1,486	$ 3,173	$ 294	$ -	$ -	$ -	$ 3,615	$ 3,330,000	6
1	Apply California-style VOC limits for automotive refinishing products	$ -	$ -	$ -	$ 2,598,338	$ -	$ -	$ -	$ -	$ -	$ -	$ -	$ 2,600,000	7
3	Voluntary program to replace gasoline lawn eqmt	$ 58,794	$ 70,207	$ 172,580	$ 519,668	$ 124	$ 1,638	$ 376	$ -	$ 8	$ 7,917	$ 718	$ 832,000	8
7	Heavy duty vehicle licensing program	$ 1,557	$ 579,791	$ 160,570	$ 22,346	$ -	$ 34	$ 3	$ -	$ -	$ 7,366	$ 120	$ 772,000	9
2	Phase out CrVI emissions from miscellaneous industrial sources	$ -	$ -	$ -	$ -	$ -	$ -	$ -	$ -	$ 22,362	$ -	$ -	$ 22,400	10

Table 20 Alternate Impact Scale Scenarios for Sensitivity Analysis

Scenario No.	Change from Base Case Impact Scale	Purpose
1	50% reduction in the value of a statistical life - i.e. valuation of mortalities reduced by 50%	Vary impact of value of statistical life
2	100% reduction in the value of a statistical life - i.e. mortalities not included in impact scale	Vary impact of value of statistical life
3	Impact Scale for NO_X, VOC and $PM_{2.5}$ set to $6,600/tonne, $10,000/tonne, and $75,000/tonne, respectively	Comparison to impact scales provided in private communication with Health Canada staff
4	Toxics (HAPs) removed from Impact Scale - i.e. valuations set to zero	Impact of CACs vs. HAPs
5	Unit risks for air toxics changed from prioritized value, to the maximum value reported by STI	Worst case impact of HAPs
6	Valuation for benefits in FVRD not included in impact scale - i.e. scale includes Greater Vancouver only	Impact only for Metro Vancouver
7	Impact Scale for NO_X, VOC and $PM_{2.5}$ each increased by 25% - as test of importance of considering secondary PM formation	Test for considering secondary PM
8	100% increase in the air toxics (HAPs) impact scale - i.e. valuations increased by factor of 2	Impact if HAP impact scale under-estimated
9	50% decrease in the air toxics (HAPs) impact scale - i.e. valuations decreased by factor of 2	Impact if HAP impact scale over-estimated

Table 21 Alternate Impact Scales ($/tonne)

Pollutant	Base Case	Scenario 1	Scenario 2	Scenario 3	Scenario 4	Scenario 5	Scenario 6	Scenario 7	Scenario 8	Scenario 9
CO	433	219	5	433	433	433	397	433	433	433
NOx	30,324	15,180	36	6,600	30,324	30,324	23,498	37,905	30,324	30,324
PM$_{2.5}$	356,822	190,329	23,836	75,000	356,822	356,822	315,087	356,822	356,822	356,822
VOC	51,967	26,202	437	10,000	51,967	51,967	46,016	64,958	51,967	51,967
SOx	19,844	9,922	0	19,844	19,844	19,844	19,233	24,805	19,844	19,844
1,3-Butadiene	27,016	27,016	27,016	27,016	0	27,016	25,256	27,016	54,033	13,508
Benzene	532	532	532	532	0	4,671	501	532	1,063	266
Carbon tetrachloride	6,514	6,514	6,514	6,514	0	6,514	6,474	6,514	13,027	3,257
ChromiumVI	14,153,000	14,153,000	14,153,000	14,153,000	0	27,933,552	13,340,400	14,153,000	28,305,999	7,076,500
Diesel PM	16,370	16,370	16,370	16,370	0	16,371	14,596	16,370	32,739	8,185
Formaldehyde	4,025	4,025	4,025	4,025	0	8,720	3,732	4,025	8,049	2,012

Table 22 Impact Valuations of Sample Actions for Alternate Impact Scale Scenarios

Actions		Action Valuation ($/year)									
Action No.	Description of Action	Base Case	Scenario 1	Scenario 2	Scenario 3	Scenario 4	Scenario 5	Scenario 6	Scenario 7	Scenario 8	Scenario 9
1	Apply California-style VOC limits for automotive refinishing products	$ 2,600,000	$ 1,310,000	$ 21,800	$ 500,000	$ 2,600,000	$ 2,600,000	$ 2,300,000	$ 3,250,000	$ 2,600,000	$ 2,600,000
2	Phase out CrVI emissions from miscellaneous industrial sources	$ 22,400	$ 22,400	$ 22,400	$ 22,400	$ -	$ 44,100	$ 21,100	$ 22,400	$ 44,700	$ 11,200
3	Voluntary program to replace gasoline lawn eqmt	$ 832,000	$ 430,000	$ 27,400	$ 221,000	$ 821,000	$ 836,000	$ 731,000	$ 980,000	$ 843,000	$ 827,000
4	Heavy duty vehicle inspection maintenance program	$ 16,900,000	$ 8,660,000	$ 443,000	$ 3,950,000	$ 16,700,000	$ 16,900,000	$ 13,800,000	$ 20,100,000	$ 17,000,000	$ 16,800,000
5	Residential wood stove incentives	$ 3,330,000	$ 1,770,000	$ 201,000	$ 716,000	$ 3,330,000	$ 3,340,000	$ 2,950,000	$ 3,450,000	$ 3,340,000	$ 3,330,000
6	Zero emission vehicle mandate	$ 22,300,000	$ 11,300,000	$ 373,000	$ 5,890,000	$ 22,200,000	$ 22,400,000	$ 19,200,000	$ 27,000,000	$ 22,400,000	$ 22,300,000
7	Heavy duty vehicle licensing program	$ 772,000	$ 395,000	$ 19,100	$ 173,000	$ 764,000	$ 772,000	$ 619,000	$ 922,000	$ 779,000	$ 768,000
8	Restrictions on wood burning appliances	$ 438,000,000	$ 233,000,000	$ 27,500,000	$ 93,800,000	$ 437,000,000	$ 438,000,000	$ 386,000,000	$ 447,000,000	$ 438,000,000	$ 437,000,000
9	Impact of Nonroad Diesel Engine Emissions Regulation	$ 36,100,000	$ 19,100,000	$ 2,060,000	$ 8,430,000	$ 35,200,000	$ 36,100,000	$ 30,200,000	$ 40,400,000	$ 36,900,000	$ 35,600,000
10	Implementation of ECA	$ 193,000,000	$ 102,000,000	$ 11,800,000	$ 114,000,000	$ 188,000,000	$ 193,000,000	$ 178,000,000	$ 215,000,000	$ 198,000,000	$ 190,000,000

Table 23 Impact Rankings of Sample Actions for Alternate Impact Scale Scenarios

Actions		**Action Ranking**									
Action No.	**Description of Action**	**Base Case**	**Scenario 1**	**Scenario 2**	**Scenario 3**	**Scenario 4**	**Scenario 5**	**Scenario 6**	**Scenario 7**	**Scenario 8**	**Scenario 9**
1	Apply California-style VOC limits for automotive refinishing products	7	7	9	7	7	7	7	7	7	7
2	Phase out CrVI emissions from miscellaneous industrial sources	10	10	8	10	10	10	10	10	10	10
3	Voluntary program to replace gasoline lawn eqmt	8	8	7	8	8	8	8	8	8	8
4	Heavy duty vehicle inspection maintenance program	5	5	4	5	5	5	5	5	5	5
5	Residential wood stove incentives	6	6	6	6	6	6	6	6	6	6
6	Zero emission vehicle mandate	4	4	5	4	4	4	4	4	4	4
7	Heavy duty vehicle licensing program	9	9	10	9	9	9	9	9	9	9
8	Restrictions on wood burning appliances	**1**	**1**	**1**	2	**1**	**1**	**1**	**1**	**1**	**1**
9	Impact of Nonroad Diesel Engine Emissions Regulation	3	3	3	3	3	3	3	3	3	3
10	Implementation of ECA	2	2	2	**1**	2	2	2	2	2	2

4. DISCUSSION AND RECOMMENDATIONS

The health impact scale for the Canadian LFV has been developed based on empirical relationships between air contaminant emissions within the Canadian LFV and monitored ambient air concentrations. The scope of this project did not support detailed air quality dispersion modelling and photochemical modelling studies to develop the relationships. Instead, the relationships were based on linear correlations between historical emission and ambient concentration data. This involved applying simplifying assumptions to provide a means of estimating the air quality changes associated with emission reduction. The most significant assumptions include:

1) Ambient concentrations were assumed to be uniform across census divisions;
2) Concentrations of NO_2, SO_2, CO, and VOCs within the Canadian LFV were considered a function of emissions in the Canadian LFV only, with no significant contribution from long range transport or atmospheric reactions;
3) Concentrations of $PM_{2.5}$ and ozone were assumed to have contributions from background including long range transport (i.e. resulting from emissions outside of the Canadian LFV) and photochemical or other atmospheric reactions. These background levels were treated as constants;
4) Concentrations of HAPs were assumed to be a function of emissions within the Canadian LFV, and constant background concentrations based on Remote Concentration Estimates (RCE) in the Northern Hemisphere;
5) Ozone was assumed to be VOC limited in the Metro Vancouver CD throughout the year, and primarily a function of VOC emissions in that CD. In the FVRD, ozone can be both VOC and NO_X limited over any year, and was assumed to be a function of both VOC and NO_X; and
6) Valuations of impacts of air toxics were assumed to be dominated by non-fatal cancer risk. Other health outcomes were not considered in the analysis.

Concentrations of $PM_{2.5}$ that result from secondary formation in the atmosphere from precursor air contaminants (e.g. NO_X, SO_2, NH_3) are included in ambient monitored concentration, and included with "background" for development of correlations. However, the impact scale does not account for changes in $PM_{2.5}$ concentrations that result from reductions in the emission of precursors. It is recommended that:

- Air contaminants formation and algorithms relating changes in ambient concentrations to changes in emissions used in this study should be further investigated.
- Formation of secondary $PM_{2.5}$ should be further investigated. Findings of the sensitivity analysis suggest that this could have an impact on valuations or rankings produced by the scale.
- Updating the health impact scale with outputs from the new version of AQBAT should be considered for the future. Changes in the AQBAT model may improve accuracy of CAC valuation, and provide more thorough treatment of a few HAPs. This project relied heavily on the current version (2a) of AQBAT for valuation of health impacts of CACs. Health Canada has indicated they are very close to release of a new version of AQBAT that includes modified concentration response functions and valuations. In addition, the new version includes some HAPs.

APPENDIX A
EMISSION – CONCENTRATION CORRELATIONS

Table A-24 CO emissions and annual 24-hour average ambient CO concentrations in Canadian LFV

Year	CO Emissions in Cdn LFV (tonnes)	Annual average ambient CO concentrations (ppm)		
		MV	FVRD	Cdn LFV
1995	504,550	0.68	0.36	0.65
1996	444,875	0.63	0.35	0.6
1997	430,279	0.63	0.32	0.58
1998	425,688	0.56	0.34	0.53
1999	383,621	0.54	0.32	0.49
2000	365,002	0.58	0.39	0.54
2001	383,494	0.51	0.36	0.49
2002	359,615	0.5	0.39	0.48
2003	346,628	0.48	0.38	0.46
2004	341,541	0.45	0.34	0.43
2005	285,384	0.44	0.32	0.42
2006	265,098	0.37	0.32	0.36
2007	253,520	0.29	0.27	0.29
2008	230,961	0.27	0.26	0.27
2009	225,249	0.26	0.21	0.25
2010	211,909	0.23	0.19	0.22
2011	210,219	0.22	0.18	0.21
2012	217,801	0.21	0.18	0.21
2013	199,242	0.21	0.18	0.21
2014	199,562	0.21	0.17	0.2
2015	193,007	0.22	0.2	0.22

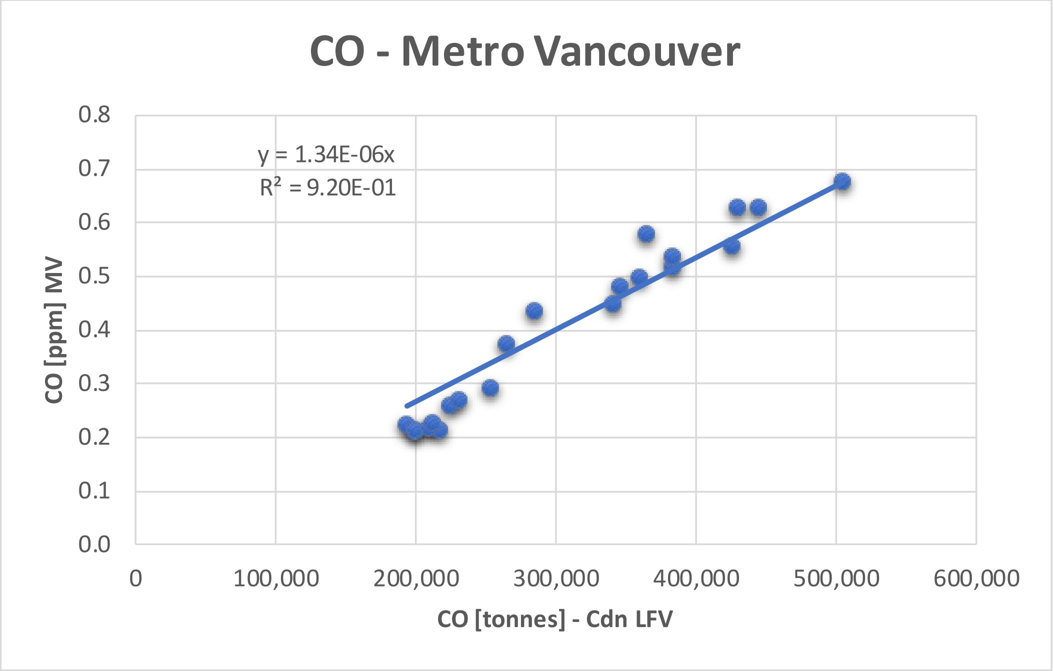


Figure A-1: CO emissions vs. annual 24-hour average ambient CO concentrations in MV

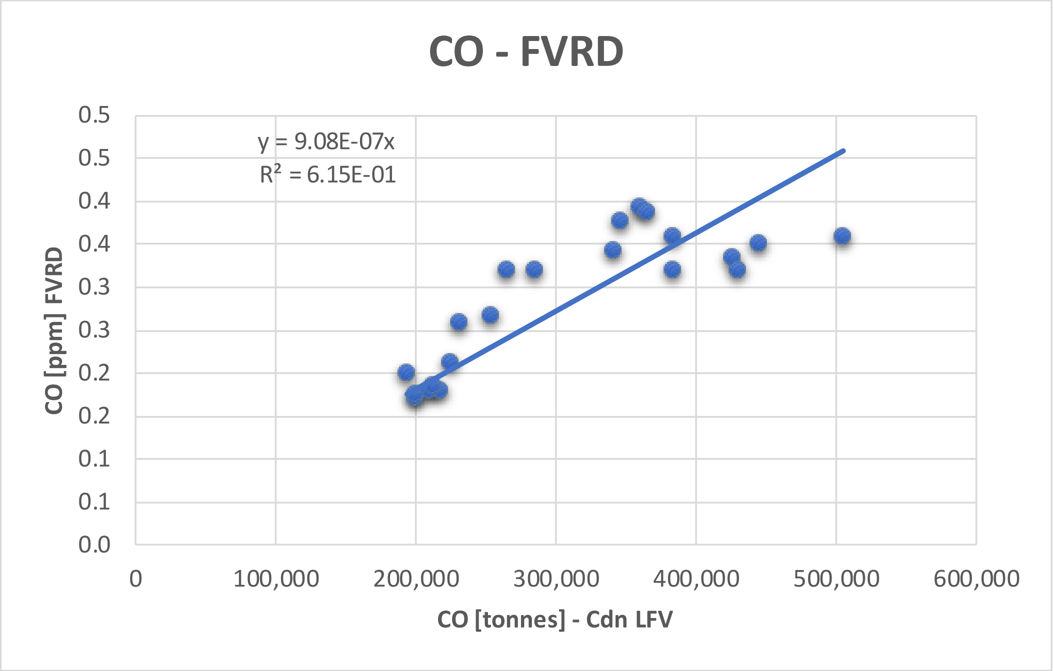


Figure A-2 CO emissions vs. annual 24-hour average ambient CO concentrations in FVRD

Table A-25 SO_x emissions and annual 24-hour average ambient SO_2 concentrations in Canadian LFV

Year	SO_X Emissions in Cdn LFV (tonnes)	Annual average ambient SO_2 concentrations (ppb)		
		MV	FVRD	Cdn LFV
1995	11,181	3.23	—	3.23
1996	8,759	2.9	—	2.9
1997	9,128	2.96	—	2.96
1998	9,131	2.56	—	2.56
1999	9,283	2.3	0.6	2.15
2000	9,408	2.67	0.5	2.49
2001	8,173	2.19	0.6	2.06
2002	7,366	2.05	0.55	1.82
2003	7,328	1.98	0.35	1.75
2004	7,718	1.75	0.7	1.61
2005	7,269	1.73	0.45	1.56
2006	6,739	1.7	0.35	1.52
2007	6,650	1.45	0.4	1.31
2008	6,679	1.49	0.5	1.36
2009	6,604	1.62	0.5	1.47
2010	6,669	1.26	0.3	1.12
2011	2,420	1.18	0.2	1.05
2012	2,479	1.26	0.15	1.12
2013	2,582	0.94	0.2	0.81
2014	2,579	0.82	0.17	0.7
2015	1,619	0.51	0.23	0.46

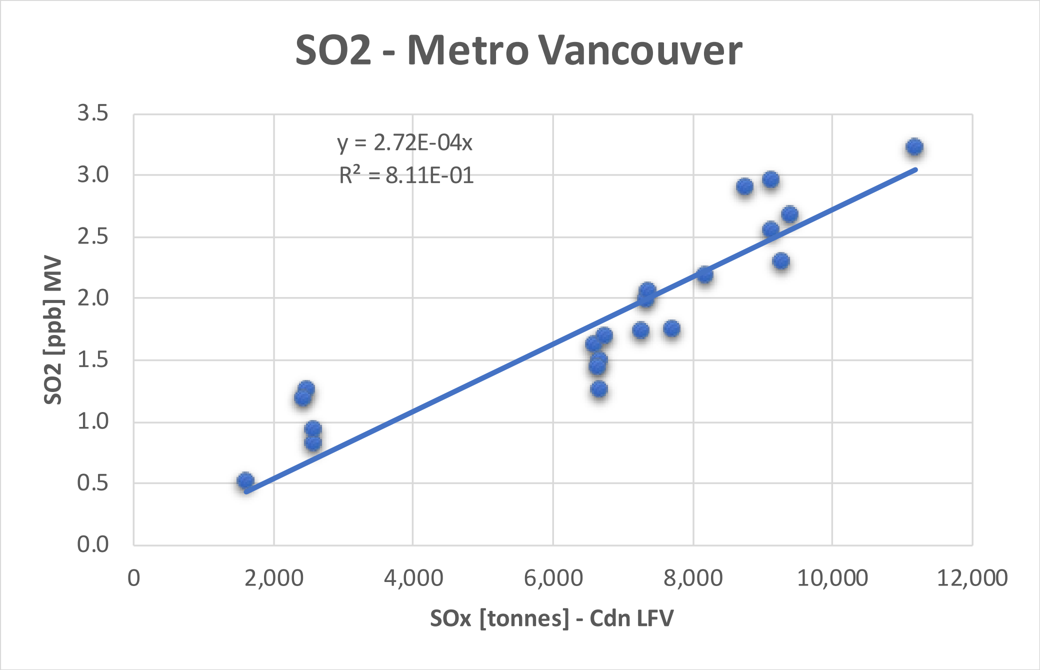


Figure A-3 SO$_X$ emissions vs. annual 24-hour average ambient SO$_2$ concentrations in MV

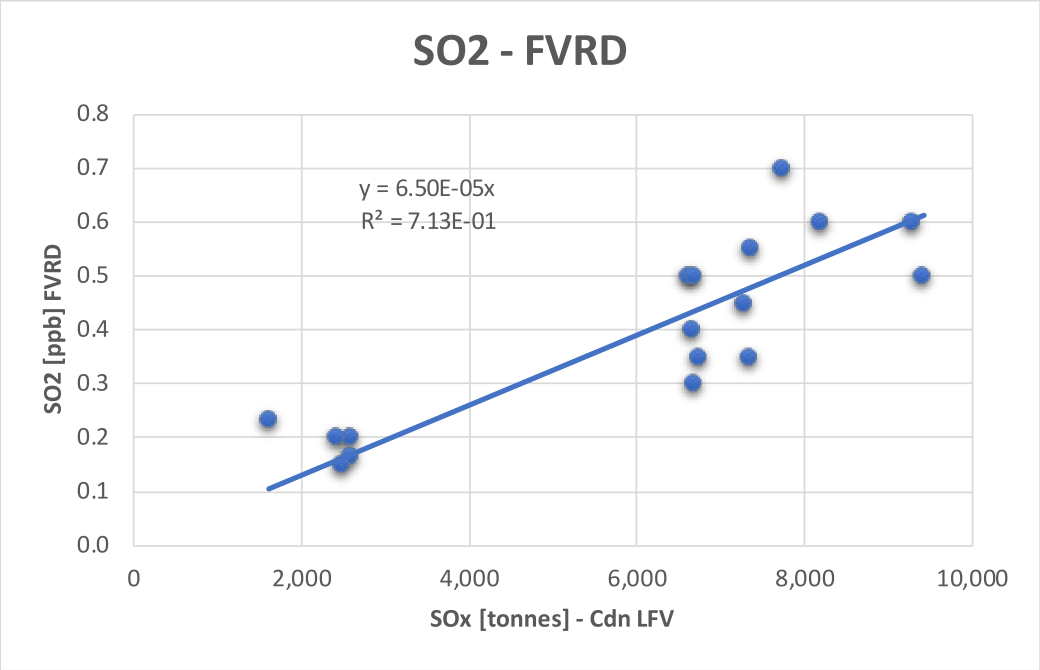


Figure A-4 SO$_X$ emissions vs. annual 24-hour average ambient SO$_2$ concentrations in FVRD

Table A-26 NOx emissions and annual average 24-hour ambient NO_2 concentrations in Canadian LFV

Year	NOx Emissions in Cdn LFV (tonnes)	Annual average ambient NO_2 concentrations (ppb)		
		MV	FVRD	Cdn LFV
1995	94,983	18.24	11.6	17.73
1996	85,982	18.25	10.6	17.66
1997	88,285	18.1	11.55	17.16
1998	89,595	17.25	13.4	16.8
1999	90,959	16.25	12	15.54
2000	90,244	17.11	12.2	16.29
2001	77,635	15.91	15.4	15.88
2002	79,256	15.89	12.97	15.38
2003	73,831	16.01	11.6	15.28
2004	76,167	14.93	10.83	14.28
2005	71,508	14.97	10.2	14.22
2006	66,189	14.39	10.13	13.72
2007	65,597	13.81	9.63	13.15
2008	63,456	13.34	8.87	12.64
2009	58,759	13.38	8.8	12.65
2010	58,898	11.19	7.93	10.65
2011	49,100	10.99	8	10.52
2012	49,484	11.6	8.07	11.07
2013	50,936	11.56	7.7	10.82
2014	51,116	11.07	7.08	10.12
2015	49,807	11.14	8.13	10.32

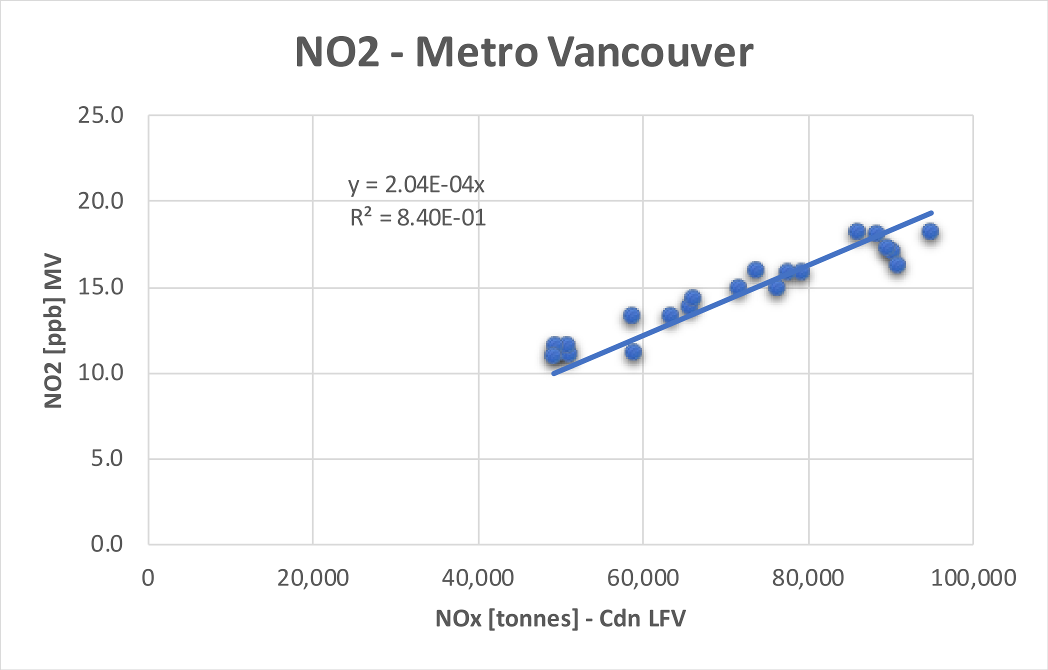


Figure A-5 NOx emissions vs. annual 24-hour average ambient NO_2 concentrations in MV

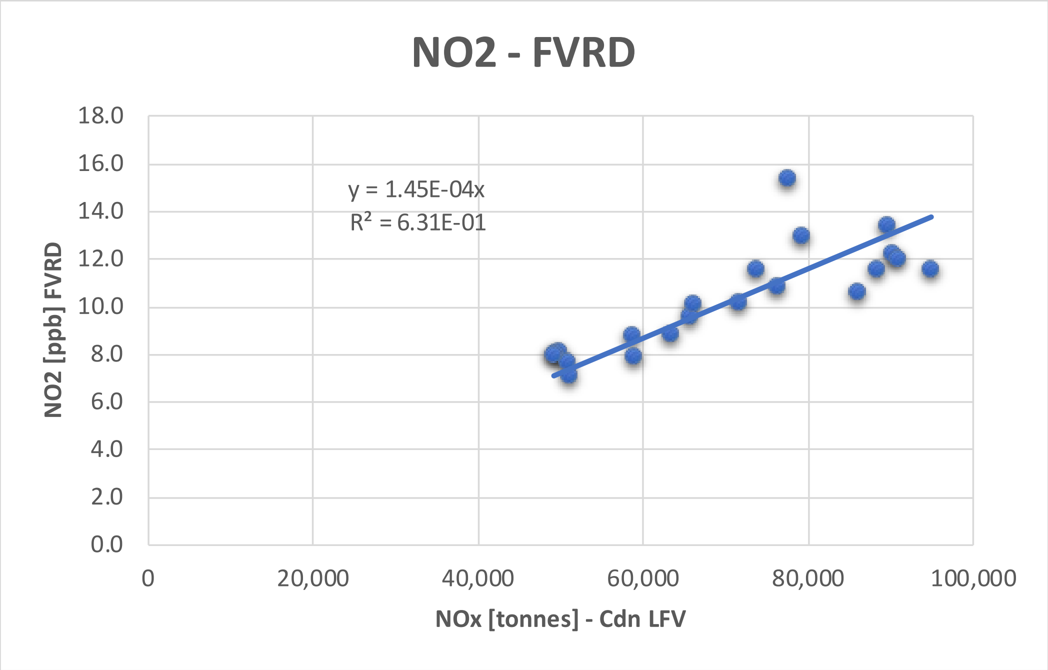
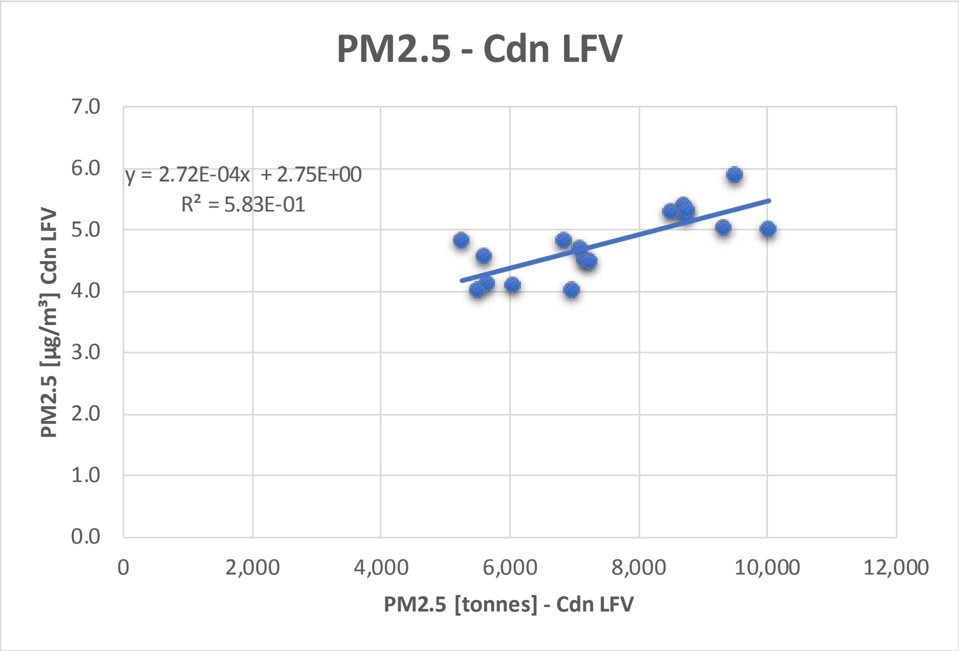


Figure A-6 NOx emissions vs. annual 24-hour average ambient NO_2 concentrations in FVRD

Table A-27 $PM_{2.5}$ emissions and annual 24-hour average ambient $PM_{2.5}$ concentrations in Canadian LFV

Year	$PM_{2.5}$ Emissions in Cdn LFV (tonnes)	Annual average ambient $PM_{2.5}$ concentrations (µg/m³)		
		MV	FVRD	Cdn LFV
1995	10,496	—	—	—
1996	12,608	—	5	—
1997	11,787	—	4.4	—
1998	10,766	—	5.3	—
1999	10,020	5.25	4.5	5
2000	9,503	6.2	5.3	5.9
2001	9,332	5.3	4.5	5.03
2002	8,747	5.47	4.9	5.33
2003	8,699	5.48	4.9	5.38
2004	8,734	5.34	4.9	5.25
2005	8,511	5.49	4.55	5.3
2006	7,092	4.88	4	4.7
2007	7,249	4.56	4.15	4.48
2008	7,176	4.58	4.2	4.5
2009	6,856	4.98	4.2	4.84
2010	6,978	4.04	3.9	4.01
2011	5,507	4.08	3.77	4.01
2012	6,064	4.11	4	4.09
2013	5,659	3.97	4.49	4.13
2014	5,616	4.55	4.57	4.55
2015	5,267	4.72	5.04	4.83



Note: Intercept represents background (including secondary) $PM_{2.5}$ concentrations in Cdn LFV

Figure A-7 PM2.5 emissions vs. annual 24-hour average ambient PM2.5 concentrations in Canadian LFV (MV and FVRD)

Table A-28 NOx and VOC emissions and annual average ambient 1-hour max O_3 + 24-hour NOx concentrations (ppb)s in Canadian LFV

Year	NOx + VOC Emissions in Cdn LFV (tonnes)	Annual average ambient 1-hour max O_3 + 24-hour NOx concentrations (ppb)		
		MV	FVRD	Cdn LFV
1995	197,273	67.16	56.07	66.62
1996	185,328	65.21	54.91	64.72
1997	186,194	65.78	53.72	64.06
1998	186,492	61.95	55.38	61.16
1999	182,351	61.79	55.73	60.78
2000	177,389	64.07	55.2	62.57
2001	168,507	61.27	62.07	61.68
2002	167,817	60.31	57.53	59.75
2003	161,090	62.18	57.13	61.29
2004	162,005	57.44	53.08	56.76
2005	148,889	58.2	50.58	56.99
2006	145,692	56.34	51.64	55.6
2007	142,257	53.89	47.77	52.93
2008	136,179	52.34	46.24	51.37
2009	127,649	54.34	47.94	53.33
2010	127,319	48.57	43.82	47.77
2011	116,178	48.12	45.67	47.72
2012	115,690	49.8	45.24	49.19
2013	117,816	49.38	43.46	48.35
2014	116,556	50.27	44.22	48.89
2015	115,215	48.38	45.59	47.53

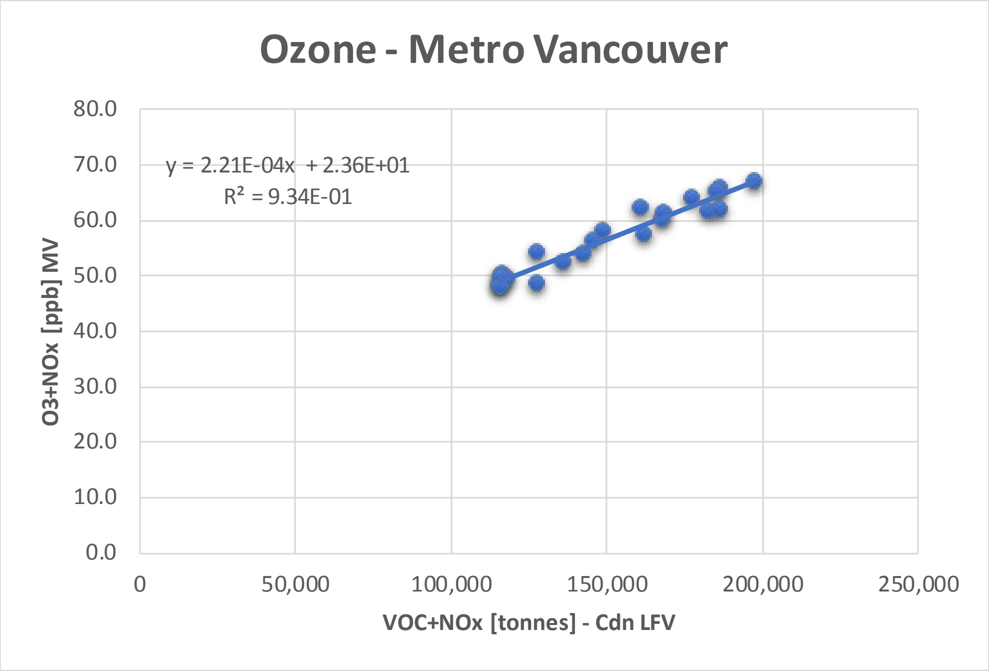


Figure A-8 VOC + NOx emissions vs. annual average ambient 1-hour max O_3 + 24-hour NOx concentrations in MV

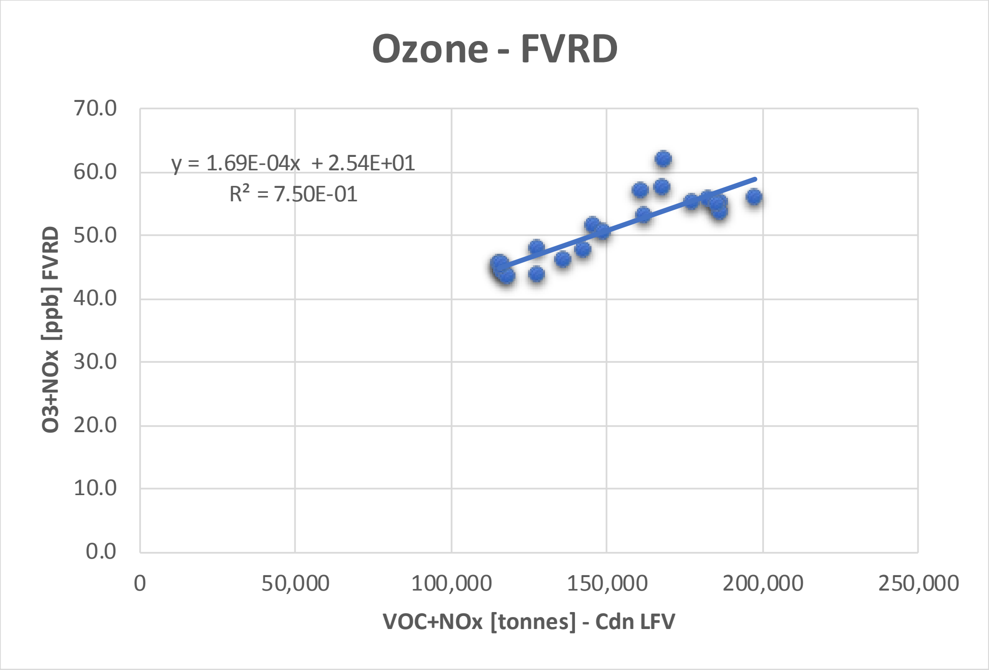


Figure A-9 VOC + NOx emissions vs. annual average ambient 1-hour max O_3 + 24-hour NOx concentrations in FVRD

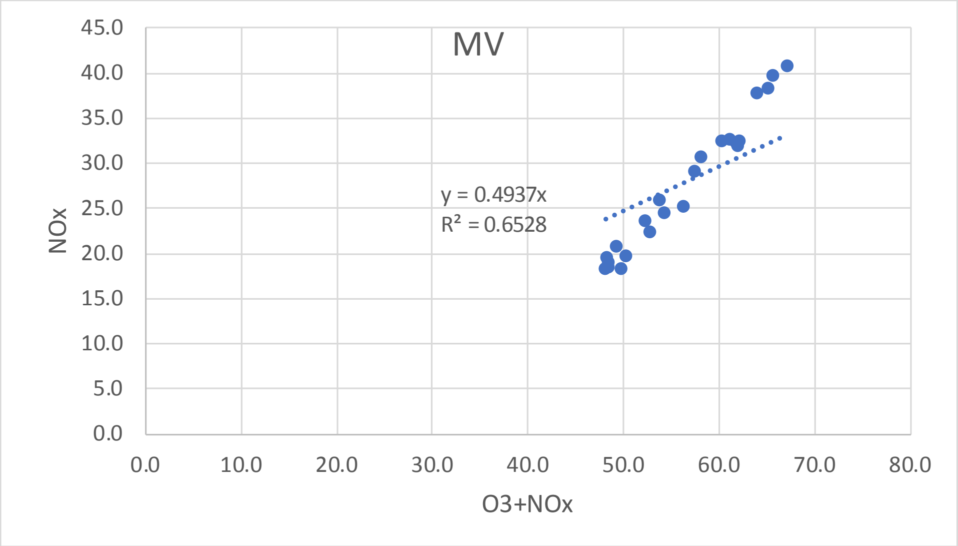


Figure A-10 Annual average ambient 24-hour NOx concentrations vs. annual average ambient 1-hour max O_3 + 24-hour NOx concentrations in MV

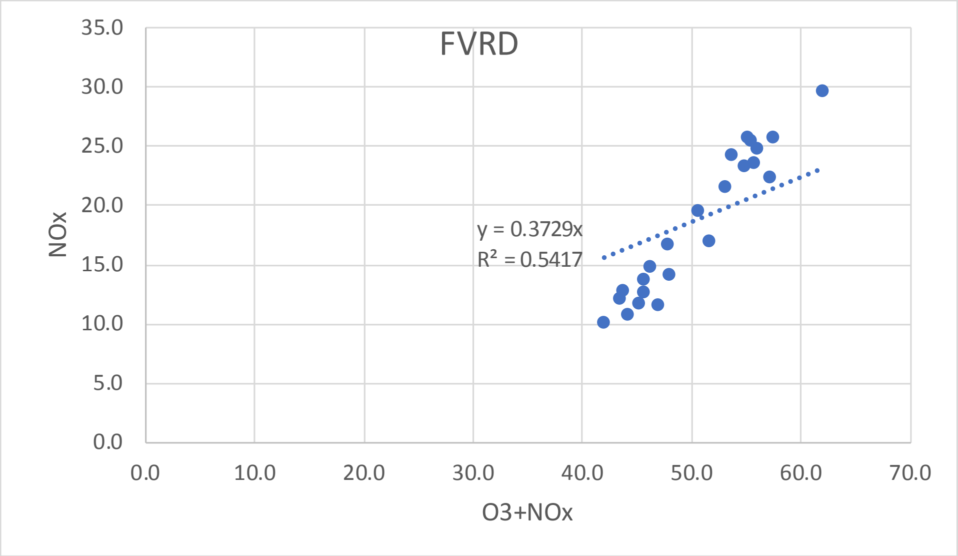


Figure A-11 Annual average ambient 24-hour NOx concentrations vs. annual average ambient 1-hour max O_3 + 24-hour NOx concentrations in FVRD