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To: Climate Action Committee

From: Francis Ries, Senior Project Engineer
Planning, Policy and Environment Department

Date: February 27, 2015 Meeting Date: March 26, 2015

Subject: **Toxic Air Pollutants Risk Assessment and Emissions Inventory for the Lower Fraser Valley**

RECOMMENDATION

That the Climate Action Committee receive for information the report dated February 27, 2015, titled "Toxic Air Pollutants Risk Assessment and Emissions Inventory for the Lower Fraser Valley".

PURPOSE

To provide a summary of a health risk assessment and emissions inventory performed for toxic air pollutants in the Lower Fraser Valley.

BACKGROUND

Goal 1 of Metro Vancouver's *Integrated Air Quality and Greenhouse Gas Management Plan* (IAQGGMP) is to "protect public health and the environment". This goal includes strategies that focus on reduction of emissions from diesel engines, industrial, commercial, institutional (ICI) and agricultural sources, residential sources, and cars, trucks, and buses. Metro Vancouver prepares regular emissions inventories of criteria air pollutants (carbon monoxide, nitrogen oxides, fine particulate matter, volatile organic compounds, sulphur oxides, ammonia) and greenhouse gases to track the progress of these strategies. Additionally, progress is tracked through continuous monitoring of criteria air pollutants at 28 stations throughout the Lower Fraser Valley, with summary reporting conducted annually.

In addition to the commonly known criteria air pollutants, emissions from sources managed under IAQGGMP Goal 1 also include small amounts of other substances, many of which are harmful to human health. These substances are referred to as Toxic Air Pollutants (TAPs), and include diesel particular matter (DPM), benzene and lead, as well as many other organic compounds and metals. A 2007 consulting study prepared for Metro Vancouver titled "Air Toxics Emission Inventory and Health Risk Assessment" identified DPM as the key driver of human health risk (primarily cancer) associated with TAPs in the Lower Fraser Valley. This result formed a key rationale for a number of air quality management actions, including the adoption of Metro Vancouver's Non-Road Diesel Engine Emission Regulation.

In 2013, Metro Vancouver convened an ad hoc working group to prepare an updated TAP health risk assessment and emissions inventory. This group included staff from Metro Vancouver, Fraser Valley Regional District, Vancouver Coastal Health and Fraser Health. Vancouver Coastal Health also provided partner funding for the work. In early 2014, Sonoma Technology Inc. was selected to conduct the "Metro Vancouver Toxic Air Pollutants Risk Assessment" study.

Study Methodology

This study followed screening health risk assessment approaches as articulated by the United States Environment Protection Agency (US EPA) and Health Canada. The time range over which the study assessed TAP risks and emissions was centered on 2010, in order to provide a risk estimate 10 years after that which was provided in the previous study, and to align emissions estimates with Metro Vancouver's emissions inventory for 2010. The list of TAPs assessed in the study included 17 pollutants identified in the previous study as representing the majority for regional TAP health risk. An additional 23 pollutants were included based on working group input and results from similar studies in other North American cities.

Human health risks associated with the 40 pollutants assessed in the study were calculated based on measured concentrations of these pollutants in Metro Vancouver and the Fraser Valley Regional District, and dose-response factors that relate pollutant concentrations to health risks. Pollutant measurement data were taken primarily from Environment Canada's National Air Pollutant Surveillance stations, which are collocated with Metro Vancouver's own ambient monitoring stations. Pollutant dose-response factors were drawn from databases maintained by three agencies: Health Canada, US EPA, and the California Office of Environmental Health Hazard Assessment. Depending on availability of appropriate dose-response factors, risks of both cancer and non-cancer health outcomes were determined.

Health Risk Assessment

Figure 1 presents the range of cancer risks estimated for the study pollutants across all Canadian Lower Fraser Valley (CLFV) measurement locations. Cancer risks are shown as lifetime excess cancer risk, which is the number of additional cancer cases expected if a million people were exposed to ambient levels of a given pollutant for a 70 year lifetime. Health Canada considers incremental cancer risks below 10 in a million to be "essentially negligible", so only pollutants with risk ranges exceeding 10 in a million are identified in the figure.

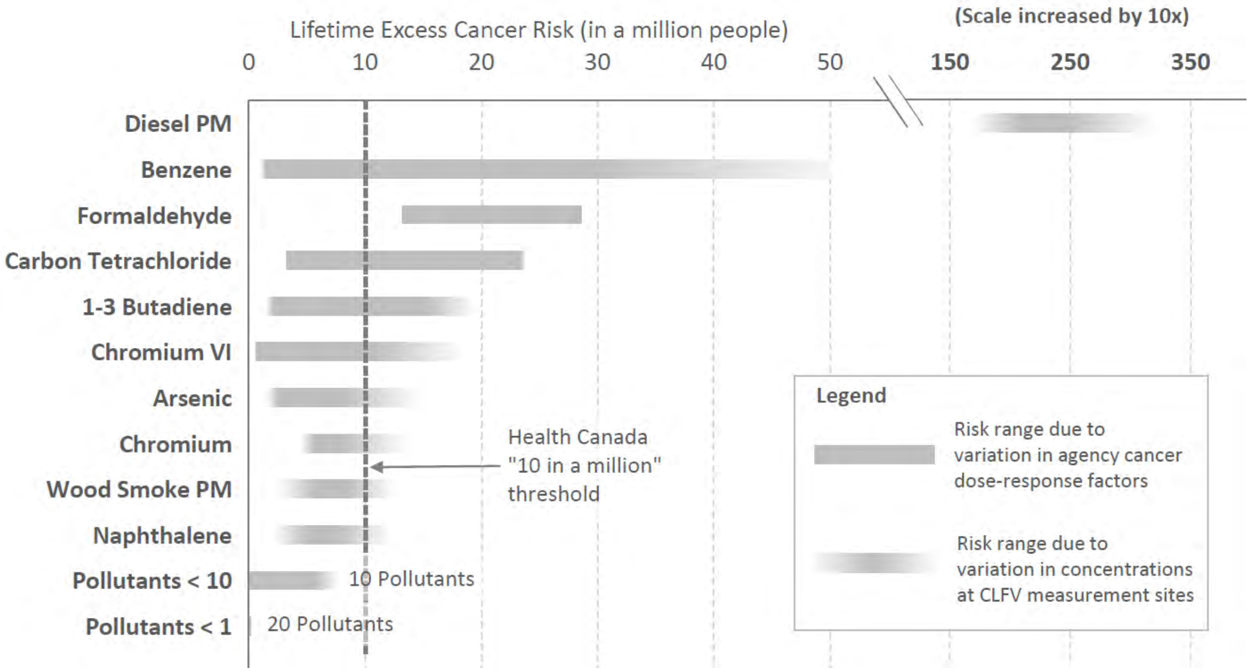

Figure 1: Lifetime Excess Cancer Risk for Toxic Air Pollutants in the Lower Fraser Valley

As with the 2007 study, the current study indicates that DPM poses far and away the highest cancer risk associated with TAPs, with a risk range at ten times higher than the remainder of the pollutants studied and larger than the sum of the cancer risks from all the other TAPs combined. The remainder of the top 10 includes a number of pollutants associated with incomplete combustion of fossil fuels or wood (benzene, formaldehyde, 1-3 butadiene, wood smoke PM, naphthalene), and three metals predominantly from fugitive dust and industrial emissions. Carbon tetrachloride stands out among the top 10 as a pollutant with no significant sources in our airshed. The international Montreal Protocol on Substances that Deplete the Ozone Layer banned the production and use of carbon tetrachloride after 2010, but it breaks down very slowly in the atmosphere. As such, atmospheric levels of carbon tetrachloride and associated cancer risks are expected to decrease slowly over the coming decades.

Non-cancer health risks (e.g. organ, neurologic, reproductive and developmental effects) were also estimated for the 40 pollutants included in the study. Similar to cancer risks, non-cancer risks were based on the assumption of a chronic lifetime exposure to ambient levels of a given pollutant. Non-cancer risks are expressed as a hazard quotient (HQ), a ratio of measured pollutant concentration to the level at which no adverse effects are expected. Health Canada considered HQs less than 0.2 to represent “essentially negligible” risk. For the CLFV, non-cancer risks were dominated by acrolein, which is associated primarily with incomplete combustion of fossil fuels or wood. Formaldehyde and DPM had lower non-cancer risks, but still greater than Health Canada’s risk threshold of 0.2, as did acetaldehyde, ammonia and trichloroethylene. Acetaldehyde is emitted naturally by vegetation, while ammonia is associated with incomplete combustion and agricultural emissions, and trichloroethylene is a solvent.

Toxic Air Pollutant Emissions

In addition to characterizing the health risks for the 40 pollutants included in the study, emissions of these pollutants were also characterized by source and municipality throughout the CLFV. In order to identify key sources of TAPs, the mass emissions of each pollutant were “risk-weighted” by applying the average cancer and non-cancer dose-response factors used in the study. The figure below indicates the percentage contribution of various source groups to the total risk-weighted TAP emissions in the CLFV. Due to the very high risk weighting of DPM, and because its sources are already well-known, DPM is excluded from the figure.

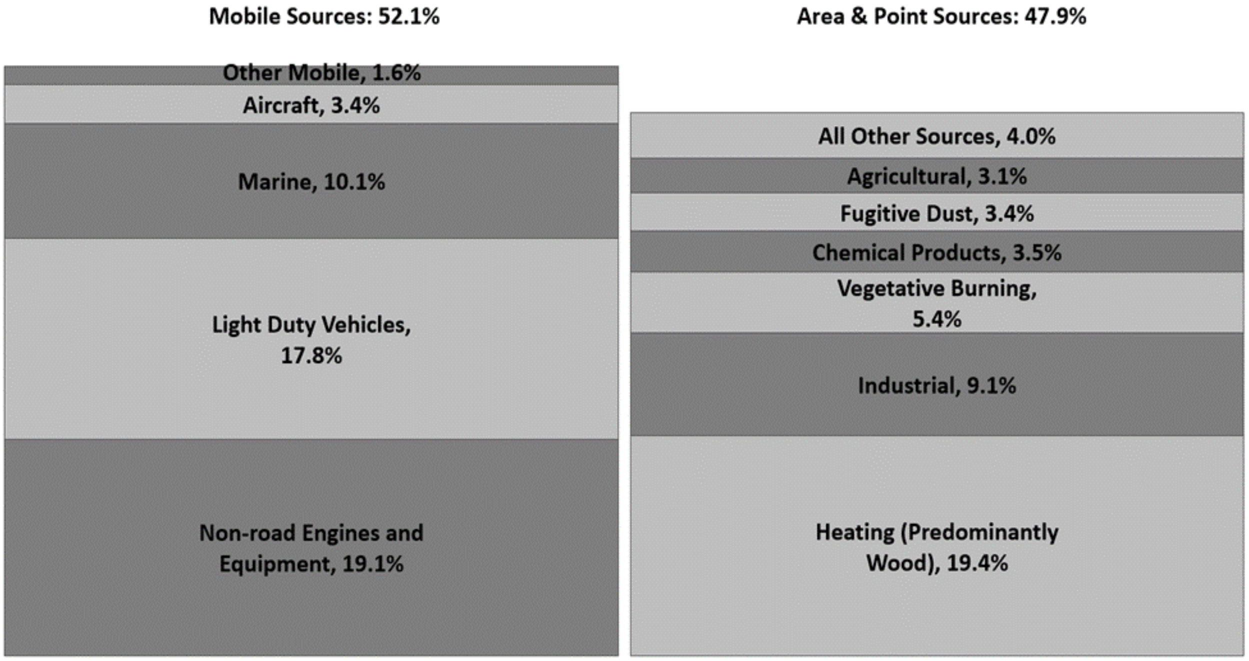

Figure 2: Sector Contributions of "Risk-Weighted" Emissions (Diesel PM Emissions not included)

Figure 2 illustrates the dominance of mobile sources, even with DPM excluded from the risk weighted totals. Non-road engines, light duty vehicles and marine emissions all contribute significantly to the mobile source total, while heavy duty vehicles represent a very small fraction of the total non-DPM TAP emissions. Heating emissions (predominantly from residential wood burning), vegetative burning (land clearing and open burning), and industrial emissions represent the other significant sources of non-DPM TAP emissions.

ALTERNATIVES

This report is provided for information only. No alternatives are presented.

FINANCIAL IMPLICATIONS

The cost of the "Metro Vancouver Toxic Air Pollutants Risk Assessment" was $107,945 USD, with a portion of the funding provided by Vancouver Coastal Health. Continued work by Metro Vancouver to act on the findings of this analysis and address sources of high priority TAP emissions has been included in the Board approval 2015 Air Quality budget.

SUMMARY / CONCLUSION

This report summarizes a health risk assessment and emissions inventory performed for toxic air pollutants in the Lower Fraser Valley. As in a previous assessment, diesel PM remains the key driver of health risk associated with toxic air pollutants in the Canadian Lower Fraser Valley. This finding supports Metro Vancouver's continued focus on sources of diesel PM emissions, including existing regulatory programs such as the Non-Road Diesel Engine Regulatory Program, as well as further investigation of measures to reduce diesel PM emissions from on-road heavy diesel vehicles. It also emphasizes the importance of the recently implemented International Maritime Organization North American Emission Control Area, as marine emissions were major sources of diesel PM and other TAPs in 2010.

Aside from diesel PM, the large contribution of light duty vehicles to total risk-weighted TAP emissions indicates the importance of implementing measures to prevent backsliding in vehicle emissions post-AirCare, while the contribution of non-road engines and equipment indicates that regulatory actions targeting gasoline non-road engines may be warranted. The prominence of risk-weighted TAP emissions associated with wood burning support Metro Vancouver's efforts to reduce emissions from residential wood burning, and emphasize the importance of reducing open burning and land clearing burning in the CLFV. Finally, Metro Vancouver's regulatory program for permitted industrial emitters should continue to focus on reductions of TAP emissions in addition to criteria pollutant emissions.

The study also identified a number of areas where monitoring of TAPs could be improved. Metro Vancouver will work with Environment Canada, the Fraser Valley Regional District and others to identify possible modifications to existing monitoring programs.

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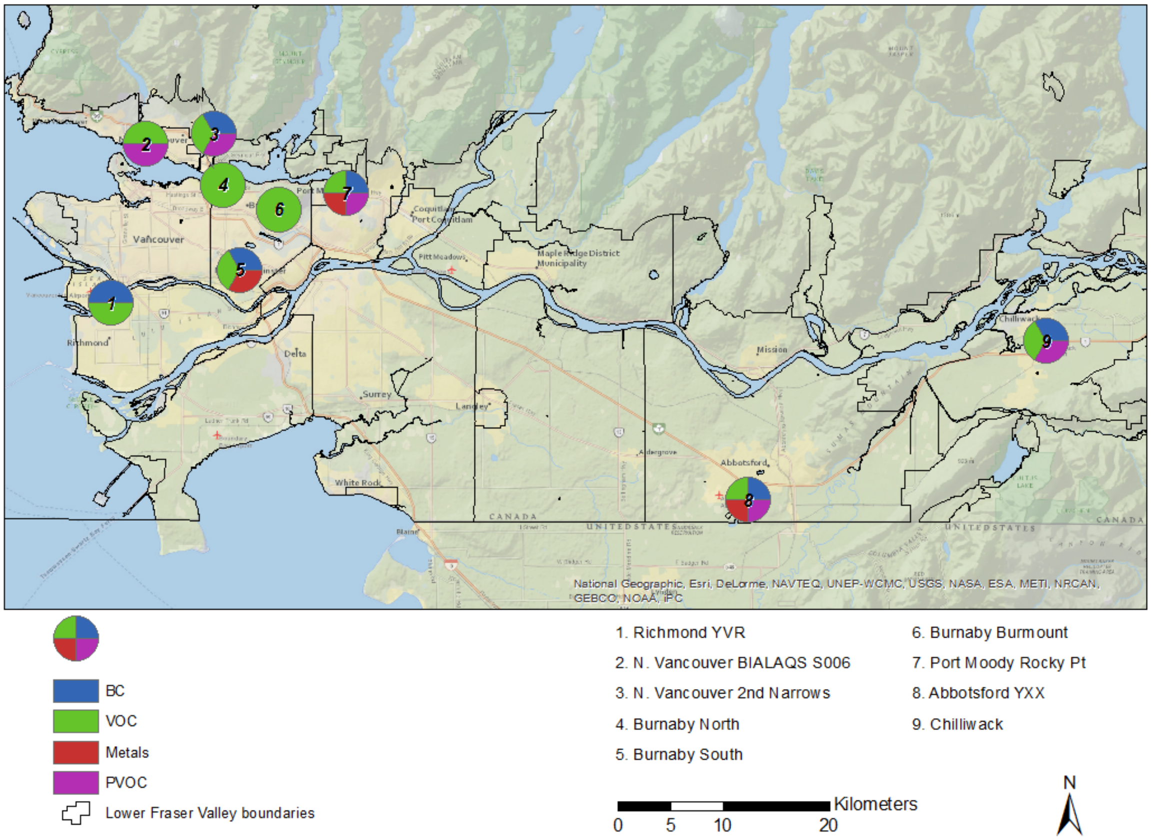
Sonoma Technology, Inc.
Environmental Science and Innovative Solutions

Toxic Air Pollutants Risk Assessment



National Geographic, Esri, DeLorme, NAVTEQ, UNEP-WCMC, USGS, NASA, ESA, METI, NRCAN, GEBCO, NOAA, IPC

Executive Summary (Final)
Prepared for

Metro Vancouver
Burnaby, British Columbia

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Toxic Air Pollutants Risk Assessment

Executive Summary (Final)

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This report was commissioned by Metro Vancouver and Vancouver Coastal Health, with development of study scope guided by an ad hoc stakeholder group that also included the Fraser Valley Regional District and Fraser Health. This report has been reviewed by representatives from each organization in this stakeholder group, but the findings and conclusions expressed in the report are the opinion of the authors of the study and may not be supported by Metro Vancouver, Vancouver Coastal Health, and the other stakeholder organizations. Any use by a third party of the information presented in this report, or any reliance on or decisions made based on such information, is solely the responsibility of such third party.

Cover graphic illustrates where toxic air pollutants are measured throughout the study region.

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Glossary

Term	Definition
BC	Black carbon
CAP	Criteria air pollutant
CAPMoN	Canadian Air and Precipitation Monitoring Network
CASN	Chemical Abstracts Service Number
CLFV	Canadian Lower Fraser Valley
DNPH	Dinitrophenylhydrazine
DPM	Diesel particulate matter
EC	Elemental carbon
EI	Emissions inventory
EPA	U.S. Environmental Protection Agency
FVRD	Fraser Valley Regional District
HCan	Health Canada
HQ	Hazard Quotient
HRA	Health Risk Assessment
IARC	International Agency for Research on Cancer
ICP-MS	Inductively coupled plasma mass spectrometry
IRIS	Integrated Risk Information System
MATES IV	Multiple Air Toxics Exposure Study IV
MV	Metro Vancouver
NAPS	National Air Pollution Surveillance
NH_3	Ammonia
NPRI	National Pollutant Release Inventory
OAQPS	U.S. EPA Office of Air Quality, Planning, and Standards
OEHHA	Office of Health Hazard Assessment
PAHs	Polycyclic aromatic hydrocarbons
PCDDs	Polychlorinated dibenzodioxins, or dioxins
PCDFs	Polychlorinated diobenzofurans, or furans
PM	Particulate matter
PVOC	Polar volatile organic compound (oxygenated)
SCC	Source classification code
STI	Sonoma Technology, Inc.
SVOCs	Semi-volatile organic compound
TAPs	Toxic air pollutants
UV	Ultraviolet
VOCs	Volatile organic compounds
WOE	Weight-of-evidence

1. Study Overview: Toxic Air Pollutants Risk Assessment

Metro Vancouver (MV) and its stakeholders initiated a study to develop updated information concerning toxic air pollutants (TAPs) in the Canadian Lower Fraser Valley (CLFV). TAPs are associated with a wide variety of adverse health effects, including cancer, neurologic effects, reproductive effects, and developmental effects. TAPs include volatile and semi-volatile organic compounds (VOCs, SVOCs), polycyclic aromatic hydrocarbons (PAHs), heavy metals, and carbonyl compounds. The information developed through this study is intended to be useful for air quality managers, who must determine which specific TAPs or emissions sources may require more detailed study, or become subject to management actions.

In this study, Sonoma Technology, Inc. (STI) completed (1) a screening Health Risk Assessment (HRA) of 40 TAPs and (2) an update of the TAP emissions inventory for a total of 51 TAPS. Through the HRA, STI evaluated ambient TAP data collected in the CLFV and applied the data to characterize the associated chronic inhalation health risks from those pollutants for the year 2010. Then, STI updated the TAP emissions inventory, focusing primarily on TAPs identified as high-priority during the HRA.

Diesel particulate matter (PM) was the dominant contributor to the total chronic inhalation cancer risk and risk-weighted emissions in the CLFV, with an average excess cancer risk of 224 per million, which exceeds the Canadian agency risk threshold of 10 per million by more than an order of magnitude (Health Canada, 2010). The next most important air pollutants related to cancer risk are formaldehyde, carbon tetrachloride, and benzene, which contribute 23, 10, and 8 average excess risk per million, respectively. The cumulative sum total excess cancer risk for the other targeted TAPs is 56 excess risk per million.

Acrolein was the dominant contributor to the chronic inhalation noncancer hazard and hazard-weighted emissions in the CLFV, with an average noncancer hazard quotient of 15.2, which exceeds Health Canada (HCan) hazard thresholds of 1 and 0.2 by more than an order of magnitude. None of the other targeted pollutants exceeded a hazard quotient value of 1, though hazard quotients for formaldehyde and acetaldehyde exceed 0.2.

There are significant uncertainties associated with health risk assessments. Each risk, hazard, and emissions estimate is dependent on multiple assumptions and uncertainties. To the extent possible, a consistent methodological approach following health risk assessment guidelines were applied to each pollutant to ensure internal consistency of study results. However, application of different assumptions or approaches, or introduction of different dose-response or pollutant concentration data will result in changes to estimated health risks for individual pollutants or sites. Thus, this study should be considered a screening-level exercise to prioritize among a large number of potential pollutants that may affect human health.

2. Health Risk Assessment

2.1 Approach

HRA comprises four steps, as adopted by the California Office of Health Hazard Assessment (OEHHA) and first laid out by the U.S. National Research Council (1983; California Environmental Protection Agency, 2001; British Columbia Ministry of Environment, 2012; Health Canada, 2010). The first step is hazard identification, which is used to identify pollutants of potential concern and their associated health impacts. The second step is dose-response assessment, which provides quantitative benchmark levels for assessing risk. The third step is exposure assessment, which involves assessing how people are exposed to the pollutant, for how long, and at what levels. Finally, the fourth step is risk characterization, where the three previous steps are synthesized into a quantitative evaluation of the pollutant's potential to cause illness or disease in the population.

2.1.1 Hazard Identification

STI worked with MV stakeholders to identify 40 priority TAPs on which to focus the study based on a candidate pollutant list containing 84 substances. MV stakeholders specified in the Request for Proposal that "the select TAPs list shall include all 17 pollutants identified to have a cancer risk greater than 1 per million or non-cancer hazard quotient greater than 0.1 by the previous risk assessment conducted for MV and EC (Levelton 2007). The select TAPs should also include wood smoke particulate and up to 22 additional pollutants (for a total of 40) based on input from project stakeholders, the proponent's review of relevant literature and, a review of ambient TAP monitoring data available for the CLFV."

The priority TAPs included in the study, as well as all pollutants not selected for inclusion, are listed in **Table 2-1**. Pollutants selected were based on a consideration of:

- Risk screening from previous studies (Levelton Consultants, 2007; McCarthy et al., 2009; U.S. Environmental Protection Agency, 2011; South Coast Air Quality Management District, 2008)
- A screening-level emissions inventory (EI) that identified pollutants that would be expected to occurred in the CLFV based on the emissions sources located in the airshed (see section 3.1 for more details).
- Availability and quality of ambient monitoring data in the CLFV, including whether sufficient data are available to create annual averages.
- Dose-response factor availability from four agencies: Health Canada (HCan), U.S. Environmental Protection Agency (EPA) Integrated Risk Information System (IRIS), U.S. EPA Office of Air Quality, Planning, and Standards (OAQPS), and California's OEHHA.
- Stakeholder priorities

The previous CLFV toxic air pollutant risk assessment included 45 pollutants (Levelton Consultants, 2007). We also note that the hazard identification step did not weight pollutant selection based on any sensitive subpopulations (e.g., children, seniors, ethnicity) to any pollutants and focused on chronic exposures. For further details and comments concerning the rationale for selection or inclusion of specific TAPs in the health risk assessment analyses, see **Appendix A**.

Table 2-1. Pollutants that were included, or considered but not included, for the HRA. Pollutants are ordered by screening ranking, with the highest priority pollutants listed first.

Pollutants Selected for HRA	Pollutants Not Selected for HRA
Acrolein	Chlorine
Formaldehyde	Hydrogen chloride
Benzene	Toluene diisocyanates
Diesel Particulate Matter	Chlorine dioxide
1,3-butadiene	Chloropicrin
Manganese	n-Hexane
Carbon tetrachloride	Hexachlorobenzene
Chromium VI	Beryllium
Acetaldehyde	Polychlorinated Biphenyls (PCBs)
Arsenic	Cobalt
Naphthalene	1,3-dichloropropene
1,4-dichlorobenzene	Propylene
1,2-dichloroethane	Propionaldehyde
Tetrachloroethylene	Copper
Ammonia	Zinc
1,2-dibromoethane	2,4,6-Trichlorophenol
Benzyl chloride	Ethylene
Ethylene oxide	1-Butene/Isobutene
Cadmium	Isoprene
Vinyl Chloride	2-Methyl-2-butene
Ethylbenzene	Isopentane
Nickel	trans-2-Butene
Dichloromethane	a-Pinene
Wood smoke Particulate Matter	trans-2-Pentene
Vanadium	d-Limonene
Acrylonitrile	1,2,4-Trimethylbenzene
1,1,2,2-tetrachloroethane	Butane
Methanol	cis-2-Butene
Antimony	Pentane
Toluene	hydrogen fluoride (hydrofluoric acid)
Trichloroethylene	Pentachlorophenol
PAHs – polycyclic aromatic hydrocarbons[a]	Thallium
Chloroform	2-Methyl-1-butene

Pollutants Selected for HRA	Pollutants Not Selected for HRA
Xylenes	Radon
Mercury	Ethylene glycol
Lead	Isopropanol
Styrene	White phosphorus
Chromium	Sulfuric acid [& oleum, acute only]
Dioxins (polychlorinated dibenzodioxins, PCDDs) and furans (polychlorinated dibenzofurans, PCDFs)[b]	Glutaraldehyde
	Asbestos
	Diethylene glycol monobutyl ether
	Tellurium
	Coal dust
	2-Methyl-1-Pentene

[a] PAHs were treated as a composite species group. The species list for the PAHs is provided in Appendix B.

[b] PCDDs/PCDFs were treated as a composite species group. The species list for the PCDDs/PCDFs is provided in Appendix B.

2.1.2 Dose-Response Assessment

Upon selection of target pollutants, inhalation dose-response factors were compiled from four agencies in March 2014:

- HCan[1]
- U.S. EPA OAQPS[2]
- U.S. EPA (IRIS)[3]
- California OEHHA[4]

Chronic inhalation dose-response benchmarks for cancer risk and noncancer hazard for target pollutants are listed in **Table 2-2**. The 1-in-a-million cancer risk level was determined for each agency value based on unit-risk estimates and a linear extrapolation of risk levels based on 70-year exposures. Blank cells in Table 2-2 indicate that a given agency does not provide a risk value for that pollutant. For example, no diesel PM risk value is available from EPA IRIS, OAQPS, nor HCan because none of these agencies consider any currently available benchmark value to be viable (despite the fact that they do recognize diesel PM as a carcinogen). Wood smoke PM is not recognized as a separate carcinogenic mixture by any of the four reference agencies used in this study; instead, a scientific literature estimate was used to provide a benchmark estimate for this assessment. For Table 2-2, the weight-of-evidence

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[1] *Part II: Health Canada Toxicological Reference Values (TRVs) and Chemical-Specific Factors Version 2.0* (2010); http://www.hc-sc.gc.ca/ewh-semt/pubs/contamsite/part-partie_ii/index-eng.php.

[2] Available at http://www2.epa.gov/sites/production/files/2014-05/documents/table1.pdf. Note that 5/21/2012 version values were used.

[3] Available at http://www.epa.gov/IRIS/.

[4] Available at http://www.oehha.ca.gov/air/allrels.html and http://www.oehha.ca.gov/air/hot_spots/pdf/CPFs042909.pdf.

(WOE) from the International Agency for Research on Cancer (IARC) is also provided to show the certainty that a given compound is carcinogenic to humans.

We note that antimony in its trioxide state (antimony trioxide) has a chronic noncancer hazard dose-response factor from both EPA IRIS and OAQPS. However, typical measurements of antimony only provide total antimony, so the exposure assessment for this pollutant is a conservative upper limit of noncancer hazard based on the assumption that all antimony is in the trioxide form. Secondly, vanadium dose-response factors were not available for chronic exposures from any of the four agency sources; they were available only for the acute 1-hr time-scale. Available measurements of vanadium in the CLFV are 24-hr duration, and therefore the comparison to acute noncancer hazard threshold should be considered qualitative.

There are significant differences across the four agencies' choices of pollutant dose-response factors. Evaluating the differences in the dose-response factors goes beyond the scope of this study. For display purposes, we have made two key assumptions for many of the figures displayed in this report.

- A dose response factor is more certain if a greater number of agencies have designated a value than if fewer agencies have designated a value. Thus, diesel PM with a single agency dose-response factor for cancer risk is more uncertain than benzene, which has four different values from the four different agencies.
- Each agency dose-response factor is equally valid for use in preliminary risk screening. This assumption eventually leads to a decision point in the risk characterization section on how to use multiple dose-response factors in presenting the preliminary risk data. We could choose the most conservative dose-response factor, choose to use a single agency set of values, have a ranked hierarchy for available dose-response factors (e.g., HCan, then OEHHA, then IRIS, then OAQPS), or weight each dose-response factor equally and use an average of the risk estimates. We used an average of the risk estimates for summary displays, but also show the individual risk values based on each dose-response value to show the sensitivity of the results to this assumption.

Table 2-2. Chronic inhalation dose-response factors for cancer risk and noncancer hazard for target pollutants.

Name	Cancer Risk (µg/m^3)						Noncancer Hazard (µg/m^3)				
	CASN[e]	IARC WOE[a]	HCan	IRIS	OAQPS	OEHHA	HCan	IRIS	OAQPS	OEHHA	Type of Effect
1,1,2,2-Tetrachloroethane	79-34-5	3			1.7E-02	1.7E-02					
1,2-Dibromoethane	106-93-4	2A	1.70E-03	3.30E-03	1.70E-03	1.40E-02	9.3	9	9	0.8	Respiratory
1,2-Dichloroethane	107-06-2	2B		3.80E-02	3.80E-02	4.80E-02			2400	400	Liver
1,3-Butadiene	106-99-0	1		3.30E-02	3.30E-02	5.90E-03		2	2	2	Reproductive
1,4-Dichlorobenzene	106-46-7	2B			9.10E-02	9.10E-02	95	800	800	800	Liver
Acetaldehyde	75-07-0	2B		4.50E-01	4.50E-01	3.70E-01		9	9	140	Respiratory
Acrolein	107-02-8							0.02	0.02	0.35	Respiratory
Acrylonitrile	107-13-1	2B		1.50E-02	1.50E-02	3.40E-03		2	2	5	Respiratory
Ammonia	7664-41-7							100		200	Respiratory
Antimony[b]	7440-36-0							0.2	0.2		Respiratory
Arsenic	7440-38-2	1	1.60E-04	2.30E-04	2.30E-04	3.00E-04			0.015	0.015	Developmental
Benzene	71-43-2	1	3.00E-01	4.50E-01[f]	1.30E-01	3.40E-02		30	30	60	Immune system
Benzyl chloride	100-44-7	2B			2.04E-02	2.04E-02					
Beryllium	7440-41-7							0.02	0.02	0.007	Respiratory, Immune system
Cadmium	7440-43-9	1	1.00E-04	5.60E-04	5.60E-04	2.40E-04			0.01	0.02	Reproductive, Developmental
Carbon tetrachloride	56-23-5	2B		1.70E-01	1.70E-01	2.40E-02		100	100	40	Liver
Chloroform	67-66-3	2B		4.30E-02		1.90E-01			98	300	Liver, Kidney, Developmental
Chromium	16065-83-1	3	9.09E-05								
Chromium VI	18540-29-9	1	1.30E-05	8.30E-05	8.30E-05	6.70E-06		0.008	0.1	0.2	Respiratory
Cobalt	7440-48-4								0.1		Respiratory
Dichloromethane	75-09-2	2B	4.30E+01	1.00E+02	1.00E+02	1.00E+00		600	600	400	Cardiovascular
Diesel PM	Diesel Emis.	1				3.30E-03			5	5	Respiratory

Name	Cancer Risk ($\mu g/m^3$)						Noncancer Hazard ($\mu g/m^3$)				
	CASN[e]	IARC WOE[a]	HCan	IRIS	OAQPS	OEHHA	HCan	IRIS	OAQPS	OEHHA	Type of Effect
Ethylbenzene	100-41-4	2B			4.00E-01	4.00E-01	1000	1000	1000	2000	Liver
Ethylene oxide	75-21-8	1			1.10E-02	1.10E-02			30	30	Nervous system
Formaldehyde	50-00-0	1		7.70E-02	7.70E-02	1.70E-01			9.8	9	Respiratory
Lead	7439-92-1	2B				8.30E-02			0.15		Developmental
Manganese	7439-96-5							0.05	0.05	0.09	Developmental
Mercury	7439-97-6							0.3	0.3	0.03	Nervous system
Methanol	67-56-1							20000	4000	4000	Developmental
Naphthalene	91-20-3	2B			2.90E-02	2.90E-02		3	3	9	Respiratory
Nickel	7440-02-0	1				3.80E-03			0.09		Respiratory
PAHs as Benzo[a]pyrene	50-32-8	1	3.20E-02		9.10E-04	9.10E-04					
PCDDs/PCDFs as 2,3,7,8-TCBD	1746-01-6	1			3.00E-08	2.60E-07			4.00E-05	4.00E-05	Immune system, developmental
Styrene	100-42-5						92	1000	1000	900	Nervous system
Tetrachloroethylene	127-18-4	2A		3.80E+00	3.80E+00	1.70E-01	360	40	40	35	Nervous system/Respiratory
Toluene	108-88-3						3750	5000	5000	300	Nervous system
Trichloroethylene	79-01-6	2A	1.60E+00	2.40E-01	2.40E-01	5.00E-01		2	2	600	Nervous system
Vanadium	7440-62-2									30[c]	Respiratory
Vinyl chloride	75-01-4	1		1.14E-01	1.14E-01	1.28E-02					
Wood smoke PM	Wood smoke	2A	1.0E-01[d]								
Xylenes	1330-20-7						180	100	100	700	Respiratory

[a] Weight-of-evidence for carcinogenicity in humans (1 - carcinogenic; 2A - probably carcinogenic; 2B - possibly carcinogenic; 3 - not classifiable).

[b] Values provided for antimony's trioxide form.

[c] Acute. [d] Lewtas J. (1988). [e] Chemical Abstract Services Number

[f] Benzene has a range of dose-response factors in IRIS. The more conservative number is used by OAQPS. We show only the upper end of the range for IRIS in the table and in the figures. This may slightly underweight risk from benzene (~3%).

2.1.3 Exposure Assessment

To estimate inhalation exposure, data were gathered from Environment Canada's National Air Pollution Surveillance (NAPS) monitoring sites in the CLFV for the period 2009–2012 (see **Figure 2-1** for monitoring locations). Annual mean concentrations were calculated, with at least one complete year (>75%) of monitoring data required for a pollutant to be included. STI used multi-year annual average concentration as an estimate of the “central tendency” of exposures over long time periods to best reflect concentrations available over the 2009-2012 time period.[5] It is also possible to use 90% or 95% upper confidence limits in the annual mean concentration as a more conservative approach which will generate “high end” risk levels. We note that two methods were used to sample and analyze for metals, which were averaged for the summary risk estimates where both methods were available at the same site and pollutant.

Due to the large percentage of time spent indoors, inhalation exposure is more closely associated with indoor concentrations than outdoor (i.e., ambient) concentrations. Outdoor to indoor pollutant infiltration is generally high in the Pacific Northwest and British Columbia South Coast (Hystad et al., 2009), but presence of indoor sources and sinks leads to highly variable indoor concentrations from building to building. As such, the ambient concentrations used in this study are not a direct surrogate for exposures, but rather provide an estimate of the outdoor “background” onto which indoor concentration variability is superimposed.

A gaseous mercury concentration estimate was developed from data from the Canadian Air and Precipitation Monitoring Network (CAPMoN) site on Saturna Island. Metro Vancouver developed concentration estimates using other NAPS monitoring sites in Canada (for PCDD/PCDFs and PAHs). All final annual concentrations are available in Excel spreadsheet format.

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[5] http://www.epa.gov/risk_assessment/exposure.htm.

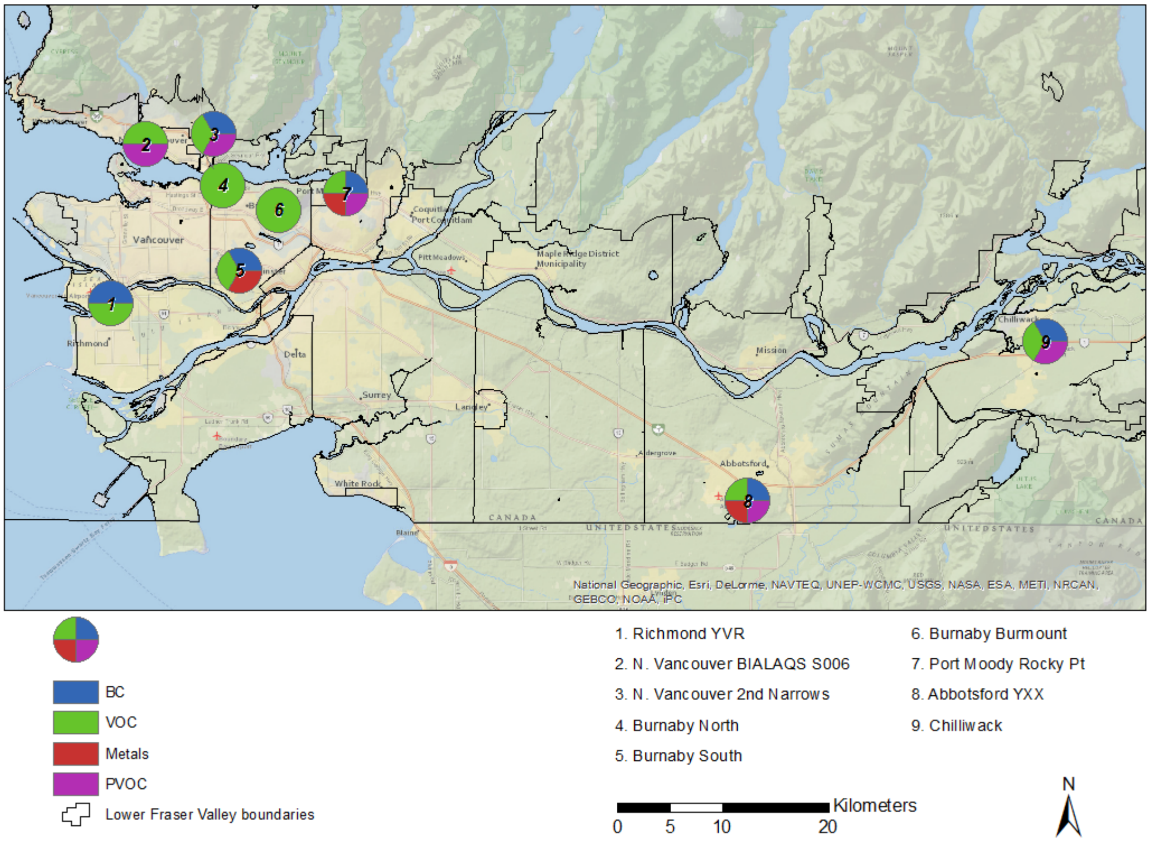

Figure 2-1. Measurement sites for key pollutant groups used in the HRA. Each site with measurements of at least one of the four key pollutant groups is shown with a colored pie.

Assessing the contributions of wood smoke PM and diesel PM required a more sophisticated analysis because these pollutants cannot be measured directly. Measurements of BC by aethalometer, elemental carbon (EC), and levoglucosan in PM were used to apportion the wood smoke and diesel PM (Fine et al., 2002a, 2002b; Mazzoleni et al., 2007; Sandradewi et al., 2008b; Turner et al., 2009).

The primary approach used to estimate wood smoke and diesel PM concentrations quantified differences in the UV and BC channels of the Magee AE22 aethalometer measurements. The difference between these two channels can be used as a proxy for “brown carbon” or wood smoke PM concentrations. An analysis of the five sites with at least 22 months of aethalometer data generated estimates of wood smoke PM concentrations that ranged from 0.15 μg/m^3 at North Vancouver – 2nd Narrows, ~0.37-0.49 μg/m^3 at other MV sites, and 0.86 μg/m^3 at the Chilliwack site. After accounting for the wood smoke PM portion of BC based on the UV-BC differential, we assumed that all remaining BC was attributable to diesel PM. Recent work has attributed a value of 0.82 for the EC:diesel PM ratio based on work done in Southern California and documented in the South Coast Air Quality Management District Multiple Air Toxics Exposure Study IV (MATES IV) draft report[6]; note that this study assumed all elemental

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[6] http://www.aqmd.gov/home/library/air-quality-data-studies/health-studies/mates-iv.

carbon was attributable to diesel PM. Using empirically derived BC:EC ratios from sites at Abbotsford and Burnaby South, we converted BC to EC by multiplying the BC by 1.63. This approach led to estimates of diesel PM concentrations at sites ranging from 0.56 μg/m^3 (Abbotsford YXX and Port Moody Rocky Pt.) to 1.02 μg/m^3 (North Vancouver – 2nd Narrows) across the CLFV. The wood smoke PM and diesel PM concentrations estimates derived using this method were used to calculate the health risk estimates presented in Section 2.2.

A second approach to corroborate these values used measurements of levoglucosan, BC, and EC as surrogates to estimate the concentrations of wood smoke PM and diesel PM. Both wood smoke PM and diesel PM contain BC and EC, but only wood smoke PM contains levoglucosan. The amount of levoglucosan emitted in smoke is highly dependent on the type of wood burned. Survey and literature data were used to estimate levoglucosan:EC ratios and a levoglucosan:wood smoke PM ratio (see **Appendix B**). Those assumptions led to estimated annual wood smoke PM concentrations of 0.65 μg/m^3 and 1.26 μg/m^3 in the MV area and the Fraser Valley Regional District (FVRD), respectively. This approach provides estimates that are slightly higher but very similar to the estimates made using the primary approach. Finally, a source apportionment modeling approach attempted to assess wood smoke and diesel PM concentrations, but did not yield usable results; this work is documented in **Appendix C**.

Finally, we note that the most common exposure pathway for gaseous toxic air pollutants is by inhalation; however, other exposure pathways such as ingestion through drinking water, soil, food, or dermal exposures from soils may contribute to total lifetime pollutant exposure. Exposures from these alternate pathways were not quantified under the scope of this study, which focused on inhalation exposure to ambient (i.e., outdoor) air. The exclusion of alternate exposure pathways may impact estimated risks for pollutants that are likely to be deposited from the atmosphere to the aquatic or terrestrial environments. For example, mercury exposure is often associated with eating fish, which depends on the location and types of fish consumed. Persistent organic pollutants like PCDDs can also bioaccumulate in food chains. The ambient concentrations used in this study do not provide an estimate of the full multipathway exposures expected from these pollutants. However, the use of ambient concentrations is consistent with other toxic air pollutant risk assessments, including the previous assessment for the CLFV and other U.S. risk screenings (Levelton Consultants, 2007; McCarthy et al., 2009; U.S. Environmental Protection Agency, 2011; South Coast Air Quality Management District, 2008).

2.1.4 Risk Characterization

Using the dose-response factors detailed in Section 2.1.2 and the ambient pollutant concentration estimates detailed in Section 2.1.3, chronic inhalation cancer risks and non-cancer hazard quotients were calculated according to the equations shown below. Incremental lifetime cancer risks and hazard quotients were calculated using the average concentrations for each different measurement location, each TAP, and each dose-response factor (detailed in Section 2.1.1, 2.1.2, and 2.1.3).

- Incremental Lifetime Cancer Risk = Concentration / Cancer Dose-Response Factor (units of per million residents)

- Hazard Quotient (HQ) = Exposure Concentration / Non-cancer Dose-Response Factor. Values above 1 and 0.2 are considered above thresholds of concern.

For the summary figures, risk/hazard averaging and spatial averaging were then applied. First, cancer risk (or noncancer hazard) estimates at a given site for different agency dose-response factors were averaged. Subsequently, the average site risks for a given pollutant were averaged across all sites with measurements to generate a single CLFV risk or hazard estimate. The sensitivity of the summary CLFV figures to this approach is shown in later figures where individual sites and dose-response factors are shown individually.

More details regarding the exposure assessment and risk characterization are provided in Appendix B. Results of the risk characterization calculations are presented in Section 2.2 following, along with discussion of the results.

2.2 Health Risk Assessment Results and Discussion

Figure 2-2 summarizes the chronic inhalation cancer risk results for key risk drivers in the CLFV for the 2009-2012 time period across all sites and dose-response factors. Risks are shown in units of excess cancer cases per million residents as a result of a 70-yr exposure. Total risk for each pollutant is provided as an average of both spatial and agency-specific dose-response risk estimates across the entire CLFV and rank ordered from highest to lowest risk. Averages are made across all sites equally, with no population weighting. Pollutants with an average risk of at least 1-in-a-million are shown; those with risk levels below 1-in-a-million are combined in the bottom category. In addition to the risk estimates, the right y-axis provides a description of the certainty in the concentration estimate and the dose-response factor. Concentration certainty reflects the number and quality of measurements for a given pollutant. Dose-response certainty reflects the agreement between agencies about the quantitative benchmark for cancer risk; low certainty indicates wider discrepancies between agencies in risk values or fewer agencies providing quantitative benchmarks.

Diesel PM contributes the greatest cancer risk—larger than the sum of the cancer risks from all the other target pollutants combined. This result is consistent with the previous HRA for the CLFV (Levelton Consultants, 2007). However, there is low certainty in the diesel PM concentration estimates as a result of the assumptions needed to convert BC measurements to diesel PM concentrations. Moreover, only California's OEHHA provides a quantitative risk benchmark for diesel PM; the U.S. EPA considers the available evidence insufficient to quantify cancer risk from diesel PM. Thus, the 224-per-million diesel PM risk estimate should be considered highly uncertain as a quantitative value. Even so, the qualitative understanding that diesel PM causes very high risk is appropriate, despite the uncertainty in the quantitative estimate of its risks.[7] The 224-per-million diesel PM risk estimate is significantly lower than the 350-per-million diesel PM risk estimate from the previous CLFV HRA (Levelton Consultants, 2007), but this change is almost entirely a result of differences in the methodology for estimating diesel PM concentrations. In the previous study, elemental carbon concentrations were

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[7] Diesel PM is a mixture of compounds, including metals, PAHs, hydrocarbons, and elemental carbon. There may be some "double-counting" of risk as a result of overlap, but it is not possible to quantify the extent.

multiplied by 1.24 as a surrogate for diesel PM concentrations, with a CLFV average of 1.11 μg/m^3. If consistent methodology with the current study were applied where a EC:DPM ratio of 0.82 was applied, the previous study would yield DPM concentrations of 0.79 μg/m^3 and a DPM cancer risk estimate of 237-per-million, which is basically consistent with the current study estimate.

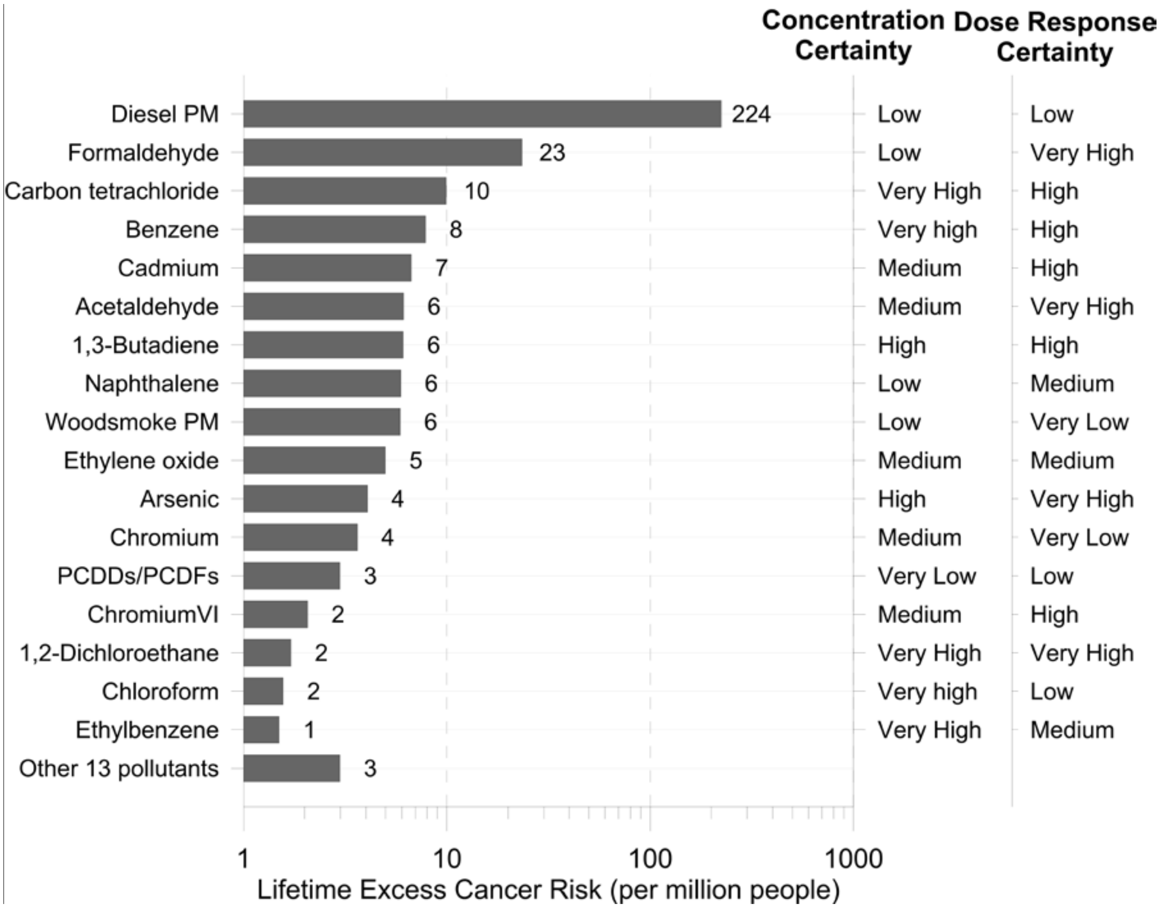

Figure 2-2. Summary of the average lifetime excess cancer risk per million people for residents of the CLFV based on average concentrations for 2009–2012.

The next highest cancer risk drivers are a group of oxygenated hydrocarbons: formaldehyde, acetaldehyde, and ethylene oxide. The cancer estimate for formaldehyde is based on data from a single site and single year of measurements, but the result is consistent with risk estimates derived from ambient measurements and modelling results from across the United States and in the Los Angeles Basin (McCarthy et al., 2009; South Coast Air Quality Management District, 2008). Acetaldehyde and ethylene oxide risk estimate results are based on data from only two sites, but results are comparable to available United States measurements (see Figure 2-10 and Figure 1 from McCarthy et al., 2009).

Hydrocarbons that are important drivers of cancer risk include benzene, naphthalene, 1,3-butadiene, and ethylbenzene. Chlorocarbons that are important drivers of cancer risk include carbon tetrachloride, 1,2-dichloroethane, chloroform, and PCDDs/PCDFs. Metals that are important drivers of cancer risk include arsenic, cadmium, chromium, and hexavalent chromium.

The wood smoke PM risk estimate is based on UV-BC estimates of wood smoke PM mass. The cancer risk estimate for wood smoke PM is the likely the least reliable value in our study because none of the four agencies from which dose response data were drawn identifies wood smoke PM as a human carcinogen nor provides a benchmark. The International Agency for Research on Cancer has listed wood smoke PM as a “probable” human carcinogen.

Figure 2-3 shows the average chronic noncancer hazard quotients for pollutants with values greater than 0.05; the other 30 pollutants are shown as a lump sum value. Some Canadian jurisdictions, including British Columbia, consider hazard quotient values below 1.0 to represent a negligible hazard, while other such as Alberta and Ontario consider values below 0.2 to represent a negligible hazard (Health Canada 2010). Of the pollutants measured in the CLFV, acrolein alone has a quotient exceeding 1 and is the dominant pollutant for noncancer hazard, while formaldehyde and acetaldehyde exceed the 0.2 threshold. The rank-order of key pollutants is largely consistent with the findings of Levelton Consultants (2007), although acrolein hazard is almost ten times higher in this study; this is due to higher measured acrolein concentrations. Diesel PM is the next highest ranked pollutant, but does not have a quotient above either threshold. Only two sites measured acrolein in the CLFV for a limited period of time and scientific studies indicate that the measurement method for acrolein in canisters may have sampling issues that can cause both positive and negative biases in concentrations.[8]

Figure 2-4 shows the spatial variability in cancer risk estimates across measurement sites in the CLFV. Variations in concentrations across the CLFV are pollutant specific. Of note, diesel PM risk was highest at the N. Vancouver – 2nd Narrows site, which is located in an industrial area near the port and the associated marine traffic. Hydrocarbons were typically highest at Burnaby North, which is near an oil refinery. Wood smoke PM was higher at inland sites Abbotsford YXX and Chilliwack than at the Metro Vancouver sites. Each site is shown with a different symbol; estimates for PAHs and PCDD/PCDFs are based on a Canadian average of four other cities since no recent measurements for the CLFV are available. Cancer risk estimates are based on the average risk using each of the agency dose-response factors. Note that some pollutants were measured at only one or two sites, while others may have been measured at eight; those measured at larger numbers of sites are more likely to represent the true spatial variability in the CLFV.

Noncancer hazard spatial variability is shown in **Figure 2-5**. Acrolein and acetaldehyde are both higher at Abbotsford YXX than at Burnaby South. Both acrolein and acetaldehyde are emitted directly through incomplete combustion processes and formed secondarily in the atmosphere from photo-oxidation of hydrocarbons. Higher concentrations downwind of urban areas are not uncommon for these pollutants (McCarthy and Hafner, 2004; Seinfeld and Pandis, 1998; Grosjean et al., 1983; Parrish et al., 2012).

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[8] http://www.epa.gov/ttnamti1/files/2009conference/Acrolein.pdf.

Figure 2-6 shows the spatial variability in pollutant concentrations between the MV area and FVRD sites. Most of the key pollutants' concentrations are higher in MV. Only wood smoke PM and arsenic concentrations are higher in the FVRD.

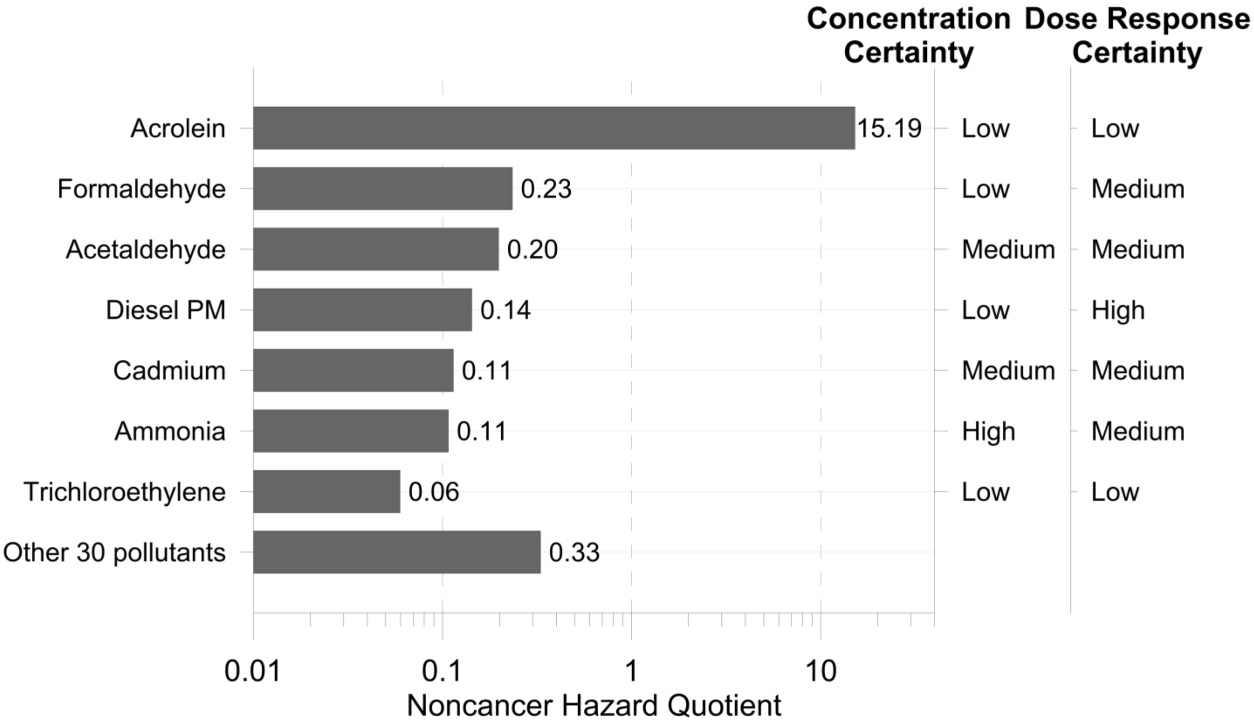

Figure 2-3. Summary of the average chronic noncancer hazard quotients for key pollutants in the CLFV for the years 2009–2012.

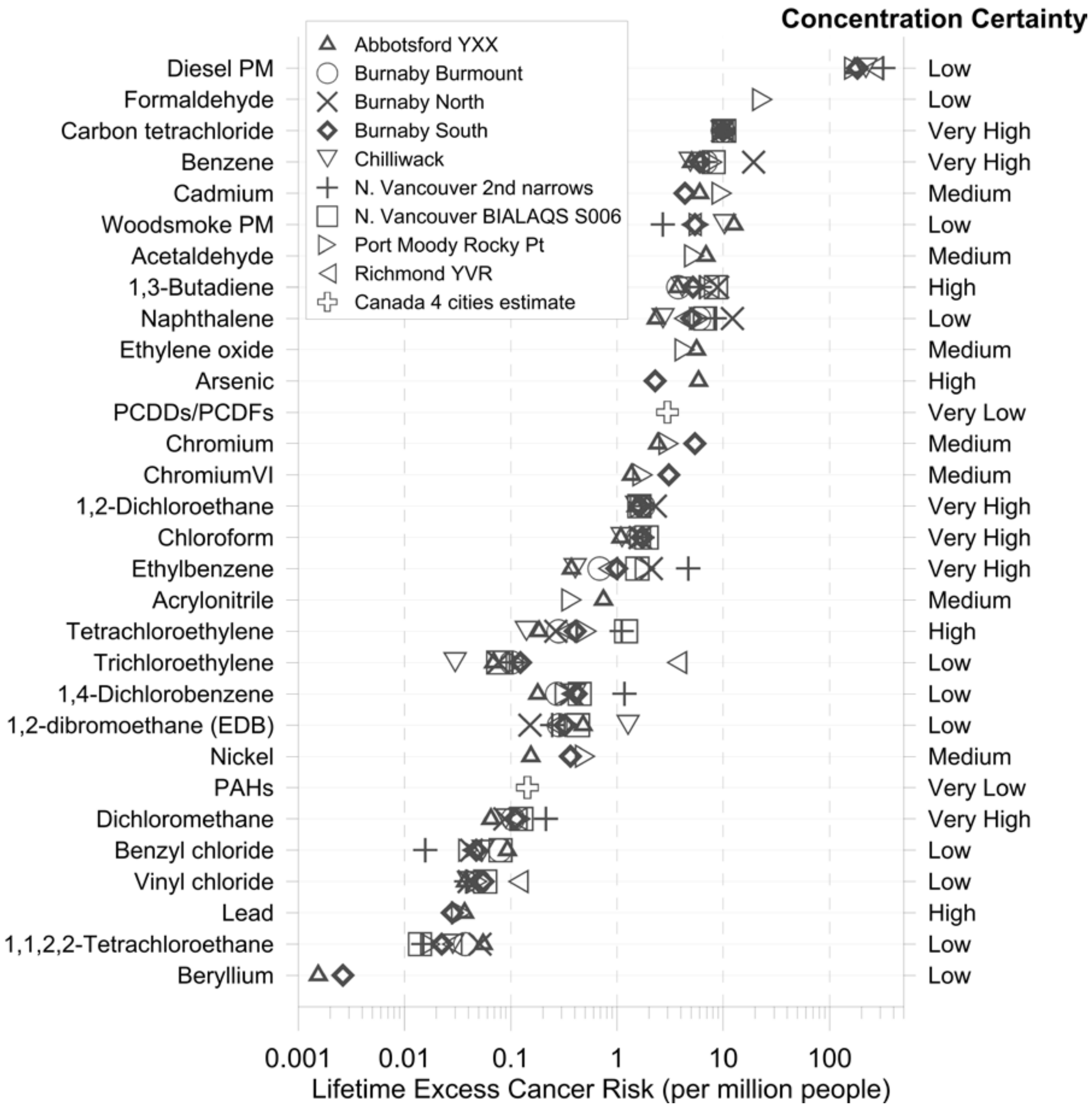

Figure 2-4. Site specific cancer risk estimates (per million people) for pollutants measured from 2009–2012.[9]

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[9] The Canada city estimate for PAHs and PCDD/PCDFs is an average from measurements at four Canadian cities, since no recent measurements of these pollutants are available in the CLFV.

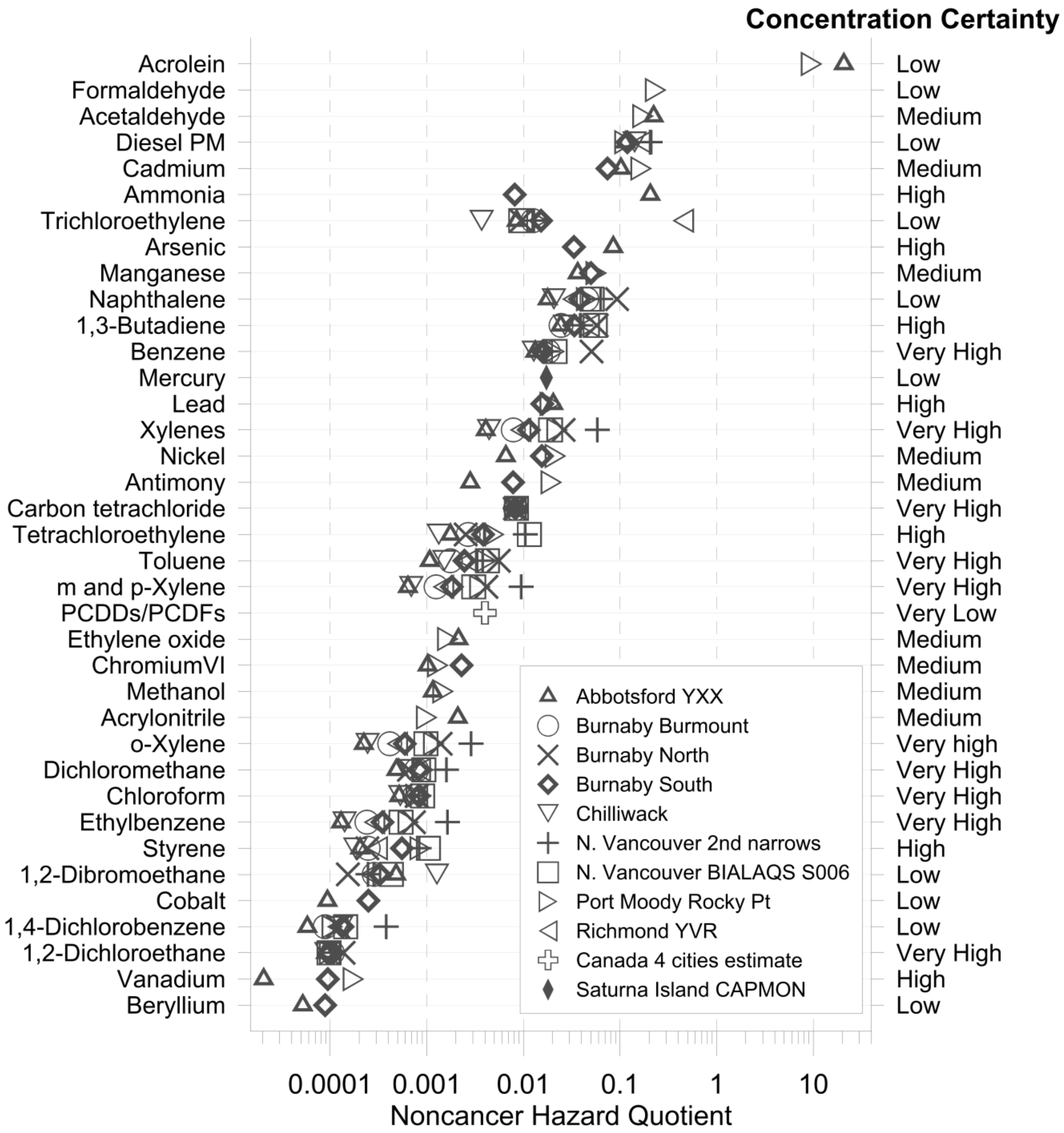

Figure 2-5. Site-specific noncancer hazard quotients for pollutants measured from 2009–2012.[10]

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[10] The Canada city estimate for PAHs and PCDD/PCDFs is an average from measurements at four Canadian cities since no recent measurements of these pollutants are available in the CLFV.

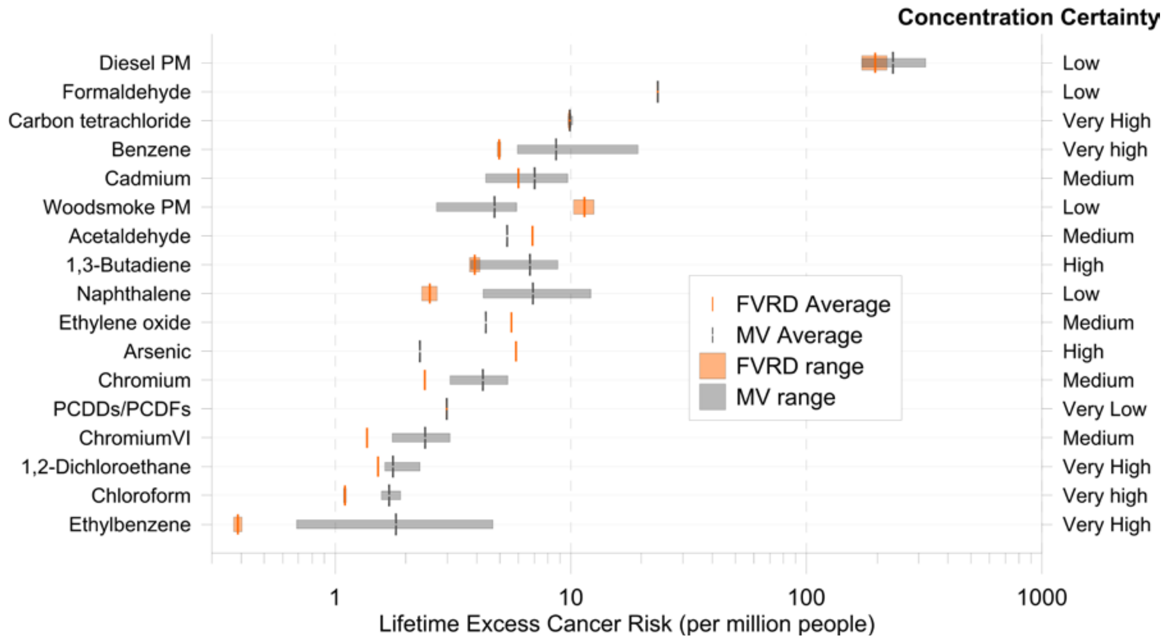

Figure 2-6. Comparison of risk ranges for key pollutants between MV area (grey) and the FVRD (orange). The average, minimum, and maximum values are shown for each pollutant; if only one site measured a pollutant, only a line is shown.[11]

Pollutants transported across the Pacific Ocean with long residence times in the atmosphere can contribute significantly to local concentrations. **Figure 2-7** shows the percentage of cancer risk attributable to northern hemisphere remote background concentrations based on a literature survey of measured concentrations for 2011 done as part of the U.S. EPA's National Air Toxics Assessment (http://www.epa.gov/nata/). Remote background concentrations were divided by CLFV average concentrations to generate a percentage estimate. Based on this approach, carbon tetrachloride is estimated to be more than 99% attributable to the global remote background contribution, meaning CLFV ambient concentrations of this pollutant cannot be mitigated through local emissions reduction measures. Chloroform, acrolein, chromium, and dichloromethane concentrations are estimated to be more than 30% attributable to background, meaning local emissions reduction measure can only influence a portion of their total ambient concentrations and associated health risk. In contrast, 1,3-butadiene, diesel PM, and ethylbenzene contributions are estimated to be less than 5% attributable to background concentrations, meaning CLFV ambient concentrations of these pollutants are almost exclusively due to emissions from local sources.

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[11] Note that MV has a possible six sites for a pollutant range while FVRD only has two; greater spatial variability should be expected for MV.

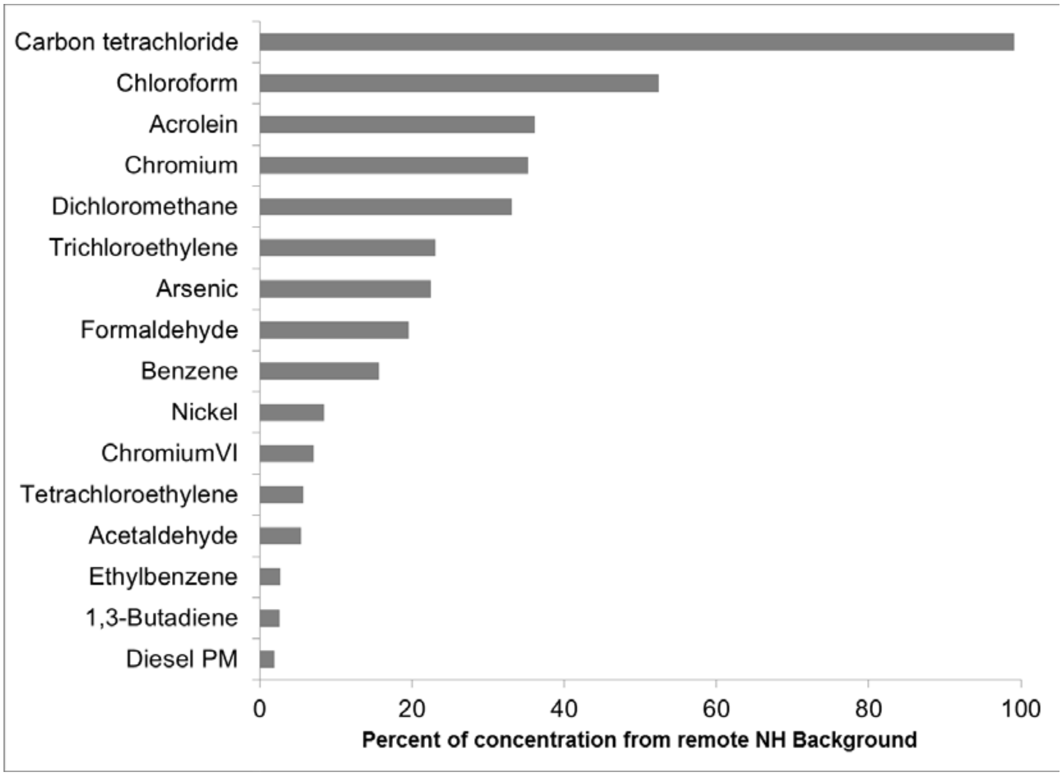

Figure 2-7. Percentage of concentrations attributable to northern hemisphere (NH) remote background concentrations based on a literature survey of measured concentrations for 2011 done as part of the U.S. EPA's National Air Toxics Assessment (http://www.epa.gov/nata/).

Figure 2-8 shows the range of uncertainty in cancer risk estimates for pollutants due to differences between agency dose-response factors for pollutants measured at the Burnaby South site, which had measurements for the largest number of cancer-risk pollutants of any CLFV site. Longer bars indicate pollutants with greater uncertainty in the quantitative dose-response values for cancer risk. Fewer dose-response factors indicates greater uncertainty in providing a quantitative value for risk estimation, as is the case for diesel PM. For many of the key pollutants, dose-response uncertainty is a bigger contributor to uncertainty in the final cancer risk estimate than the concentration uncertainty. In fact, the rank-ordering of some key pollutants according to their estimated risks is sensitive to the choice of dose-response factor such that a different result would be obtained by selecting maximum dose-response factors rather than averages. Also note that we show the results from two different sampling and analysis methods that were used to collect and analyze metals in the CLFV: "dichot" indicates the dichotomous sampling method and ICP-MS indicates inductively coupled plasma mass spectrometry.

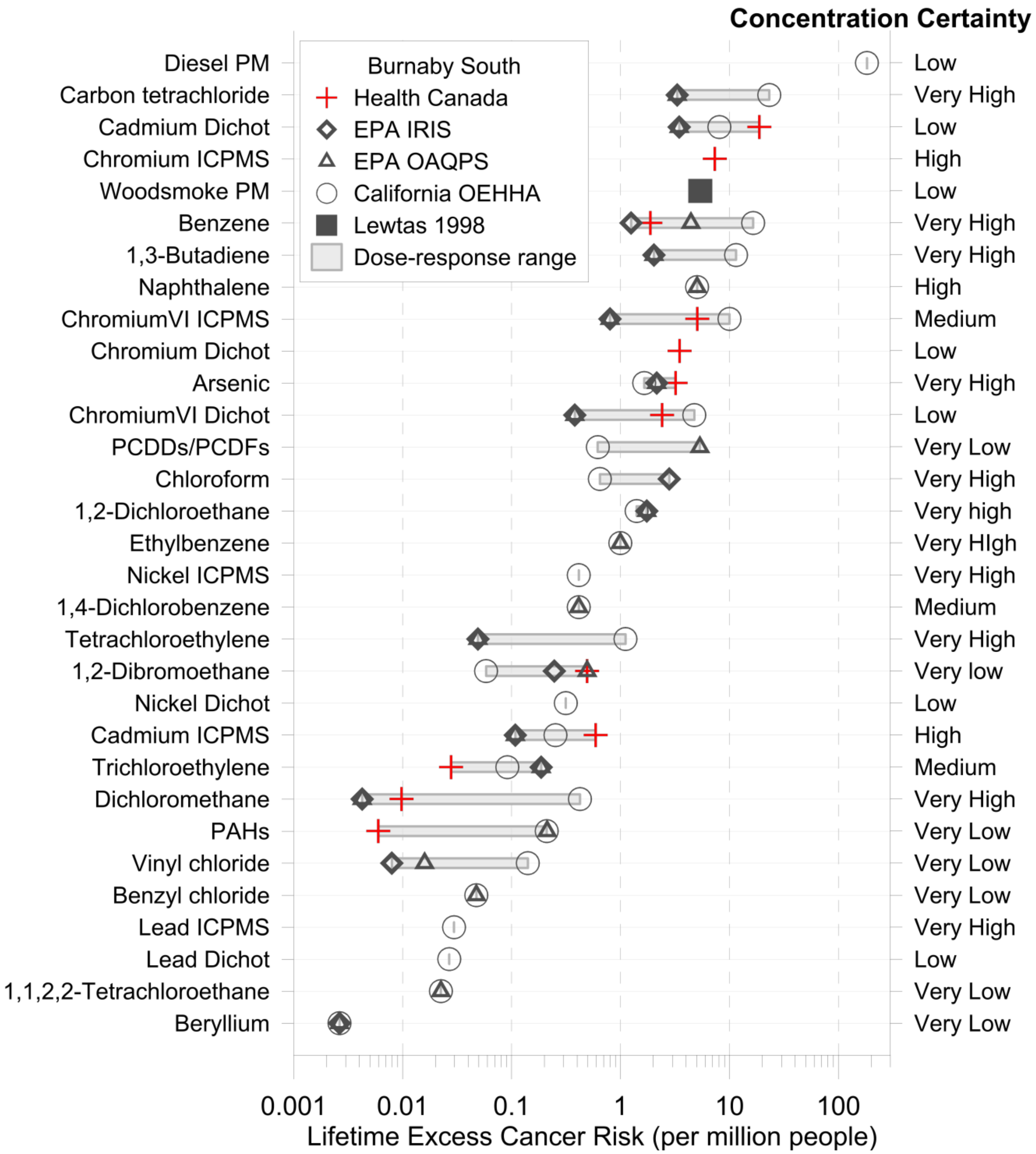

Figure 2-8. Range of cancer risk estimates for pollutants measured at Burnaby depending on choice of dose-response factor.[12]

Figure 2-9 shows the dose-response uncertainty for noncancer hazard at the Abbotsford site, which had measurements for the largest number of non-cancer hazard pollutants of any CLFV site. Acrolein has a wide range of dose-response factors, but both the high and low resultant hazard quotient values are over the thresholds of 1. All other pollutants' quotients are

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[12] Dose-response factors are only shown if the agency provides a quantitative dose-response value for that pollutant.

below the threshold of 1, but formaldehyde, acetaldehyde, diesel PM, ammonia, and trichloroethylene are above the 0.2 hazard threshold if the higher dose-response factor is used.

Comparisons of Vancouver average risk/hazard to other major metropolitan areas on the west coast of North America are shown in **Figure 2-10**. Data for other North American cities were selected from the U.S. EPA's Air Toxics Monitoring Archive.[13] Annual averages for 2010-2012 were generated by site for each of the pollutants available. Sites were then averaged up to the metropolitan area level based on metropolitan statistical area boundaries. Each of the other cities is also located on or near the Pacific Ocean. Metro Vancouver cancer risk values were higher than Seattle's, Portland's, and San Francisco's, but lower than Los Angeles' for most pollutants. Metro Vancouver hazard values for acrolein and total chromium were lower than those for the four U.S. cities. FVRD risk levels were usually lower than those for the four U.S. cities. We note that a comparison of only urban sites in Metro Vancouver to urban/suburban sites throughout the four U.S. cities may ignore the lower downwind concentrations included in metropolitan area definitions (like the FVRD, for example). We also speculate that the PVOC methodology may not be directly comparable to carbonyl measurements made routinely by dinitrophenylhydrazine (DNPH) cartridges in the U.S. cities, and this dissimilarity may account for some of the differences.

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[13] http://www.epa.gov/ttnamti1/toxdat.html#data.

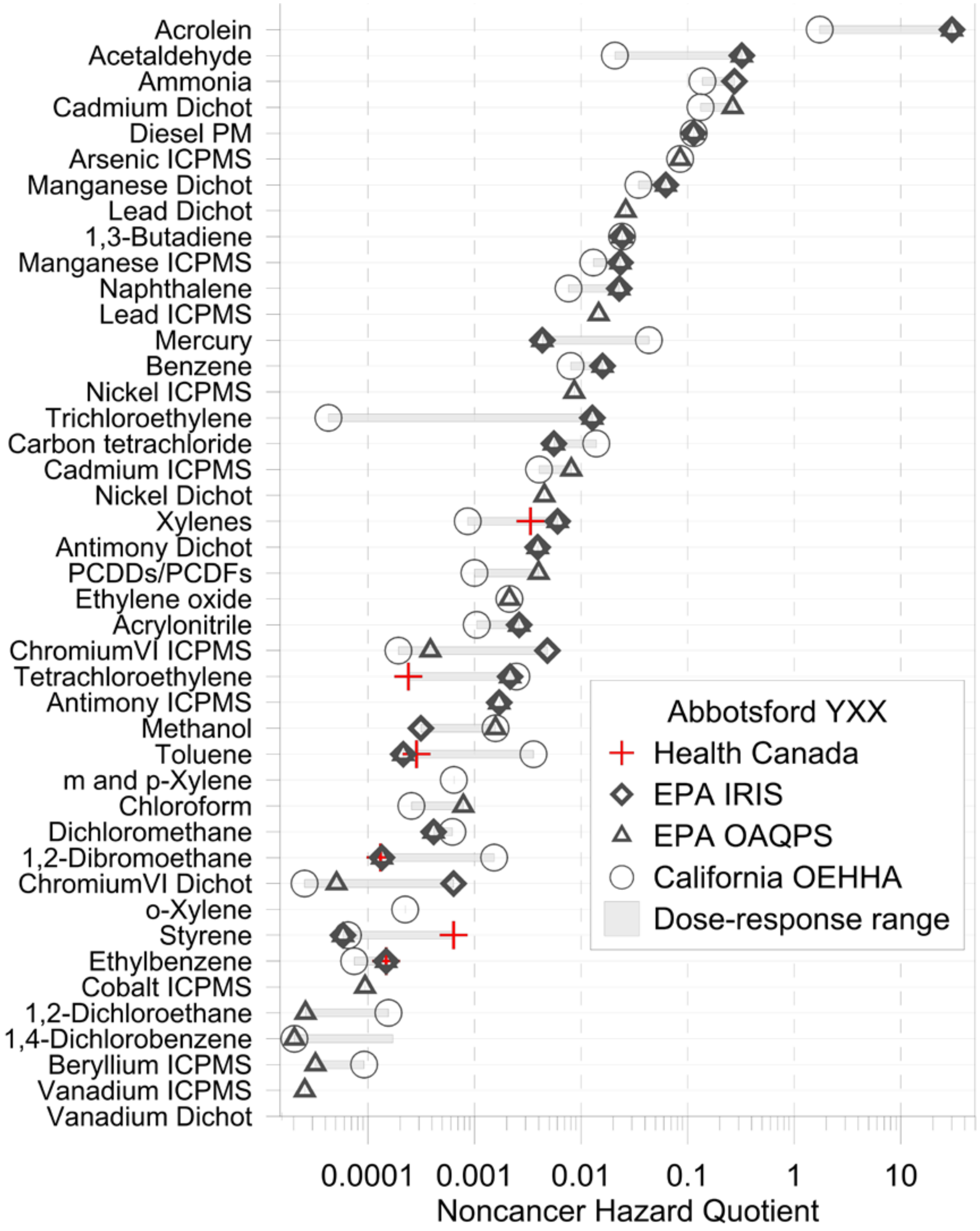

Figure 2-9. Range of noncancer hazard estimates for pollutants measured at Abbotsford YXX site depending on choice of dose-response factor.[14]

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[14] If a no dose-response factor is shown, the agency does not provide a quantitative dose-response value for that pollutant.

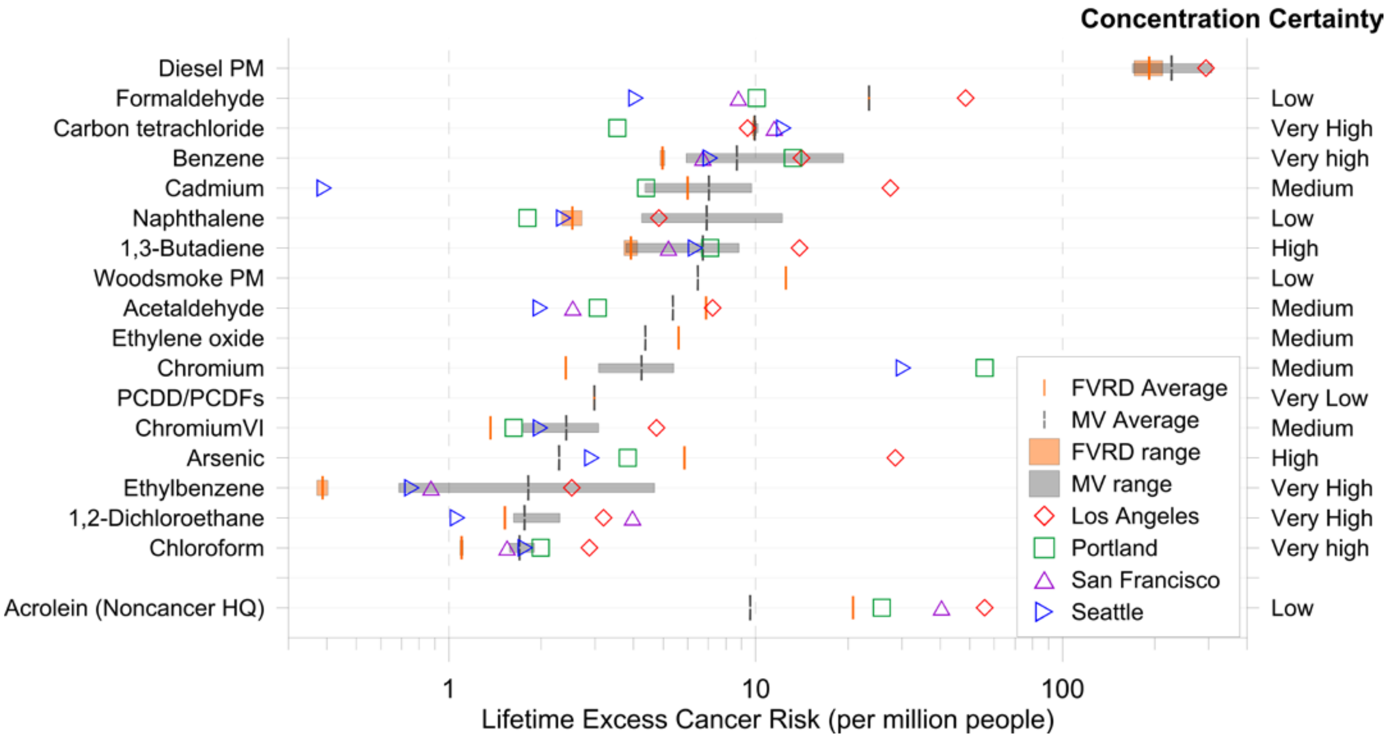

Figure 2-10. Individual pollutant cancer risk (or noncancer hazard for acrolein) from the greater Vancouver metropolitan area compared to average risk based on annual average concentrations in Los Angeles, Portland, San Francisco, and Seattle for 2010-2012.

3. Emissions Inventory

3.1 Approach

As part of the process to identify and finalize the list of 40 priority TAPs, a screening-level emissions inventory (EI) was developed. The screening EI represented a preliminary estimate of total risk/hazard-weighted emissions in the CLFV area, composed of MV and the FVRD, for each pollutant on the candidate pollutant list. The screening EI was developed using two sources of data provided by MV:

1. Readily available TAP emissions estimates, including data from the Canadian National Pollutant Release Inventory (NPRI) and MOVES model outputs containing TAP emissions
2. Criteria air pollutant (CAP) emissions estimates for 2010, which were disaggregated into individual chemical compounds (including TAPs) using chemical speciation profiles available from the U.S. EPA's SPECIATE 4.4 database and a previous VOC study for the CLFV (SNC Lavalin, 2012)

After the screening EI was reviewed to finalize the list of priority TAPs, a refined EI was developed using a hybrid method, with the data/approach selection hierarchy ordered as follows:

1. Existing TAP emissions estimates (e.g., MOVES outputs, NPRI data).
2. An emissions factor-based approach, in which emissions were estimated by multiplying available TAP emissions factors from the U.S. EPA and other data sources by corresponding activity data provided by MV.
3. A TAP augmentation approach (U.S. Environmental Protection Agency, 2013), in which TAP emissions were calculated by multiplying the appropriate CAP emissions by established TAP-to-CAP ratios for various pollutants. These ratios vary among sources and were applied based on source classification codes (SCCs).
4. Chemical speciation of CAP emissions, as was done during the development of the screening EI.

The refined EI contains risk/hazard-weighted emissions for each of 32 municipalities in the CLFV and the 39 priority TAPs[15], and 11 additional TAPs[16]. To be conservative, the risk/hazard factor applied to each TAP was the minimum value in the chronic dose-response factors assembled from various sources for the HRA task. For emissions ranking purposes, only the cancer risk factor was applied if a TAP has both cancer risk factor and noncancer hazard factor. More details about the approaches to assembling the emissions inventories are provided in **Appendix D**.

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[15] There are 40 pollutants on the finalized priority TAP list. However, the emissions of dioxins and furans are reported together as 2,3,7,8-TCDD equivalent according to communication with MV staff. Please refer to Table D-2 in Appendix D for the list of dioxin/furans compounds included.

[16] Metro Vancouver designated 12 additional TAPs while chlorine dioxide is not emitted in the CLFV.

3.2 Emissions Inventory Results and Discussion

Table 3-1 summarizes the mass-only and risk/hazard-weighted emissions, respectively, for MV, FVRD, and total CLFV[17]. As expected, MV has higher emissions than FVRD for every TAP except ammonia (NH_3), because MV includes most of the region's population and industrial sources, while the FVRD includes most of the region's agricultural lands. The ranks of each TAP based on their cancer risk and noncancer hazard weighted 2010 emissions for the CLFV are also provided. Diesel PM[18] and acrolein top the lists by substantial margins. **Table 3-2** compares the mass-only emissions of TAPs with those estimated for year 2000 for a previous study (Levelton Consultants, 2007). In general, these TAP emissions decreased significantly (30% to 94%) from 2000 to 2010. The only exception to this trend was toluene, for which emissions increased by 9% from 2000 to 2010.[19] However, one should note that inventory methods may vary between this study and the Levelton 2007 study, and no backcasting to 2000 was performed. Thus, the comparison may be subject to significant uncertainty.

Figure 3-1 through **Figure 3-5** illustrate the relative contributions of different emissions source categories to TAP emissions. Figure 3-1 provides an overview of source contributions for each TAP based on CLFV total emissions. The remaining figures analyze the source contributions to diesel PM and acrolein emissions—i.e., the TAPs with highest risk/hazard-weighted emissions—at different aggregation levels. Figure 3-2 shows that marine vessels are the dominant emission sources of diesel PM, while on-road vehicles and non-road engines and equipment are also key contributors. Also, non-road engines and equipment emit more diesel PM than onroad vehicles. This finding is consistent with the diesel particulate matter (DPM) emissions trends and percentage distribution across sectors in 2010 Lower Fraser Valley Air Emissions Inventory Forecast and Backcast (Metro Vancouver, 2013). Acrolein is emitted from a broader array of source categories, with aircraft being the largest contributor (Figure 3-3). Figure 3-4 and Figure 3-5 show the variability in source contributions between jurisdictions.

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[17] For emissions ranking purpose, the TAP emissions from "Natural Sources" are not accounted for in any table/figures in this report, though they are retained in the underlying TAP EI database.

[18] Specifically referred to diesel $PM_{2.5}$.

[19] This increase in emissions for toluene was largely due to the TAP augmentation approach applied to area sources such as consumer products. The TAP augmentation approach generally assumes a higher toluene-to-VOC ratio than would be derived from the speciation profiles applied to area sources in the 2000 inventory.

Table 3-1. 2010 TAP emissions for MV and FVRD, actual mass and cancer/noncancer risk weighted mass (tonnes).

Pollutant	Actual Emissions (tonnes)			Risk-weighted emissions (tonnes)				Rank	Rank
	MV	FVRD	Total	Risk/Hazard	MV	FVRD	Total	Cancer	Noncancer
1,1,2,2-tetrachloroethane	9.43E-05	0	9.43E-05	Cancer	0.01	0	0.01	29	
1,2-dibromoethane	1.8	0.06	1.9	Cancer	1,093	35	1,128	12	
1,2-dichloroethane	2.5	0.05	2.5	Cancer	65	1.3	66	25	
1,3-butadiene	88	15	104	Cancer	14,990	2,627	17,617	5	
1,4-dichlorobenzene	89	10	99	Cancer	977	115	1,092	14	
Acetaldehyde	230	87	318	Cancer	622	236	858	16	
Acrolein	38	6.0	44	Non-Cancer	1,875	300	2,176		1
Acrylonitrile	2.5	0.03	2.5	Cancer	716	8.5	725	17	
Aluminum	79	27	106	N/A	--	--	--		
Ammonia	4,451	6,149	10,600	Non-Cancer	45	61	106		2
Antimony	0.34	0.07	0.41	Non-Cancer	1.7	0.37	2.1		7
Arsenic	0.07	0.01	0.08	Cancer	421	75	497	18	
Benzene	703	129	832	Cancer	20,386	3,748	24,134	2	
Benzyl Chloride	1.8	0.02	1.8	Cancer	87	1.0	88	24	
Cadmium	0.39	0.18	0.57	Cancer	3,843	1,769	5,612	8	
Carbon tetrachloride	2.8	0.03	2.8	Cancer	118	1.3	119	23	
Chlorine	22	8.0	30	N/A	--	--	--		
Chlorobenzenes	1.9	0.10	2.0	N/A	--	--	--		
Chloroform	2.5	0.05	2.6	Cancer	58	1.2	59	26	
Chlorophenols	1.27E-04	6.86E-05	1.96E-04	N/A	--	--	--		
Chromium	1.7	0.16	1.8	Cancer	18,330	1,782	20,112	4	
Chromium VI	0.04	1.72E-03	0.04	Cancer	5,811	257	6,068	7	
Copper	1.1	0.22	1.3	N/A	--	--	--		
Dichloromethane	1,020	112	1,132	Cancer	1,020	112	1,132	11	
Diesel Particulate Matter	1,265	186	1,451	Cancer	898,343	132,006	1,030,349	1	
Dioxins & Furans (PCDD/PCDF as TCDD TEQ)	9.58E-07	3.78E-07	1.34E-06	Cancer	32	12	44	27	

Pollutant	Actual Emissions (tonnes)			Risk-weighted emissions (tonnes)				Rank	Rank
	MV	FVRD	Total	Risk/Hazard	MV	FVRD	Total	Cancer	Noncancer
Ethylbenzene	393	67	460	Cancer	982	167	1,149	10	
Ethylene oxide	1.6	0.02	1.6	Cancer	142	1.6	143.7	21	
Formaldehyde	433	79	512	Cancer	5,633	1,023	6,656	6	
Hexachlorobenzene	9.54E-04	6.86E-04	1.64E-03	N/A	--	--	--		
Hydrogen chloride	55	0.67	55	N/A	--	--	--		
Hydrogen fluoride	0.01	0	0.01	N/A	--	--	--		
Lead	2.3	0.58	2.9	Cancer	28	6.9	35	28	
Manganese	2.0	0.65	2.7	Non-Cancer	41	13	54		3
Mercury	0.08	3.59E-03	0.09	Non-Cancer	2.7	0.12	2.9		6
Methanol	1,535	237	1,771	Non-Cancer	0.38	0.06	0.44		8
Naphthalene	43	6.9	50	Cancer	1,469	234	1,703	9	
Nickel	4.1	0.09	4.2	Cancer	1,077	24	1,101	13	
Phosphorus	29	2.0	31	N/A	--	--	--		
Polychlorinated biphenyls (PCBs)	1.86E-05	8.69E-06	2.73E-05	N/A	--	--	--		
Polycyclic aromatic hydrocarbons (PAH as B[a]P TEF)	0.18	0.03	0.2	Cancer	195	37	231	19	
Styrene	34	3.6	37	Non-Cancer	0.36	0.04	0.40		9
Tetrachloroethylene	150	18	168	Cancer	887	104	991	15	
Toluene	5,068	724	5,792	Non-Cancer	17	2.4	19		5
Trichloroethylene	48	5.0	53	Cancer	198	20	218	20	
Vanadium	8.6	0.14	8.8	Non-Cancer	0.29	4.80E-03	0.29		10
Vinyl Chloride	1.8	0.03	1.8	Cancer	140	2.7	142	22	
Woodsmoke PM	1,571	459	2,030	Cancer	15,709	4,589	20,298	3	
Xylenes	1,722	280	2,002	Non-Cancer	17	2.8	20		4
Zinc	14	1.0	15	N/A	--	--	--		

🟧 Top cancer and noncancer risk weighted emissions
🟩 Top 2-10 ranked cancer weighted emissions
-- no risk-weighted emissions, no dose-response factor were available

Table 3-2. Comparison of mass-only TAP emissions by source category: 2010 vs. 2000 (tonnes).

Pollutant	2010 TAP EI - STI (tonnes)					2000 Levelton EI (tonnes)				
	Point	Area	On-road	Non-road	Total	Point	Area	On-road	Non-road	Total
Acetaldehyde	0.09	108	124	86	318	26	982	109	265	1,382
Aluminum	6.6	94	0.5	4.2	106	12	883	927	2.8	1,824
Ammonia	87	10,049	429	36	10,600	673	16,517	1,065	88	18,342
Benzene	12.2	109	266	445	832	21	176	654	322	1,173
Diesel PM	—	0.15	366	1,085	1,451	8.2	—	385	2,175	2,568
Ethylbenzene	3.3	41	186	228	460	7	54	552	152	766
Formaldehyde	16	146	144	207	512	22	71	325	622	1,040
Methanol	77	1,682	—	12	1,771	77	6,329	—	—	6,406
Toluene	128	3,620	1,062	982	5,792	142	825	3,620	730	5,317
Xylenes	73	378	684	867	2,002	130	732	2,022	580	3,464

Note: Only those TAPs that were apportioned by source category in the Levelton report (2007) are included in this table.

—: No emissions for the source category.

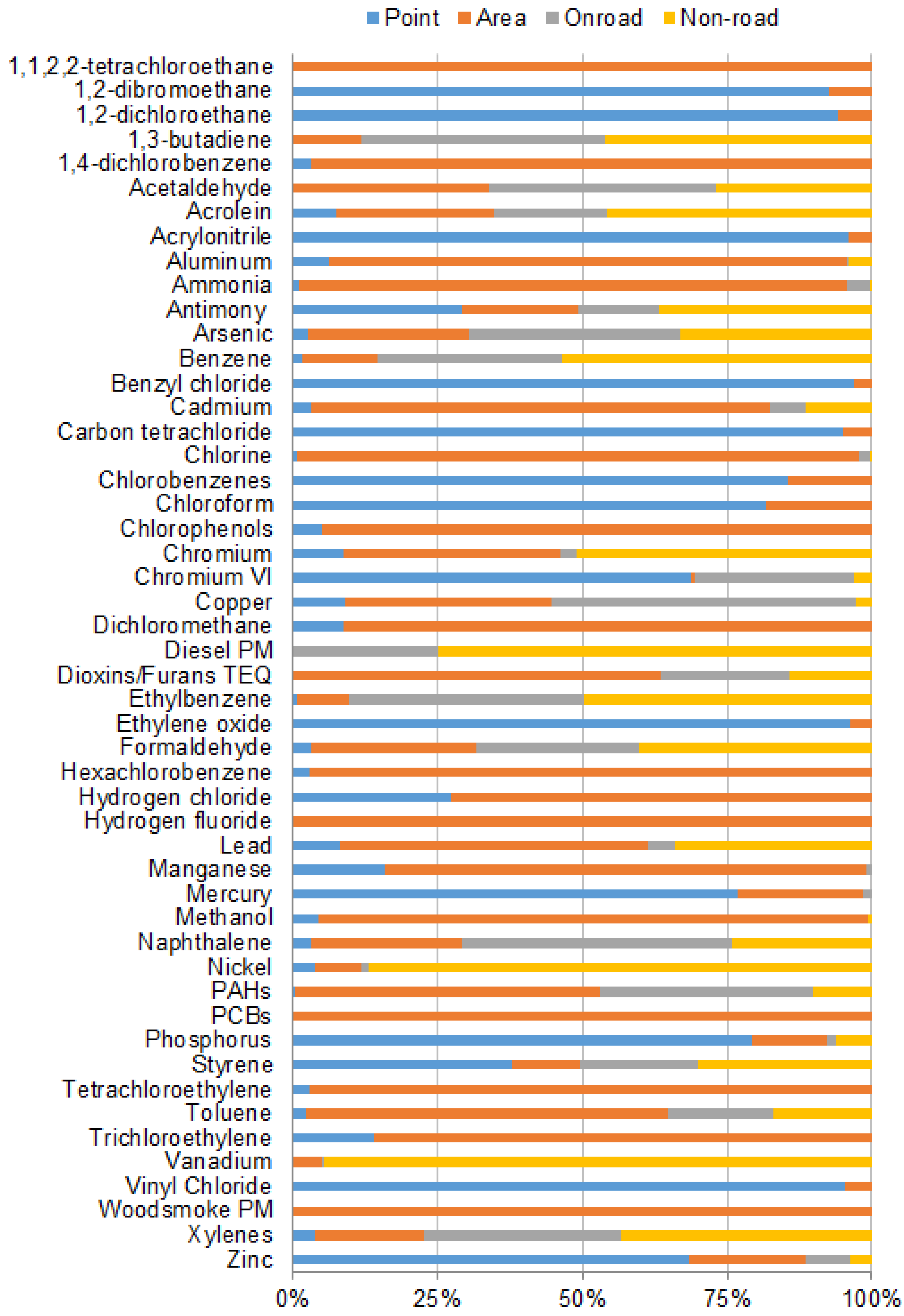

Figure 3-1. Contributions from each source sector[20] to 2010 CLFV total emissions (mass-only) by TAP.

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[20] The "Non-road" sector here includes non-road engines and equipment, aircraft, marine, and rail sources.

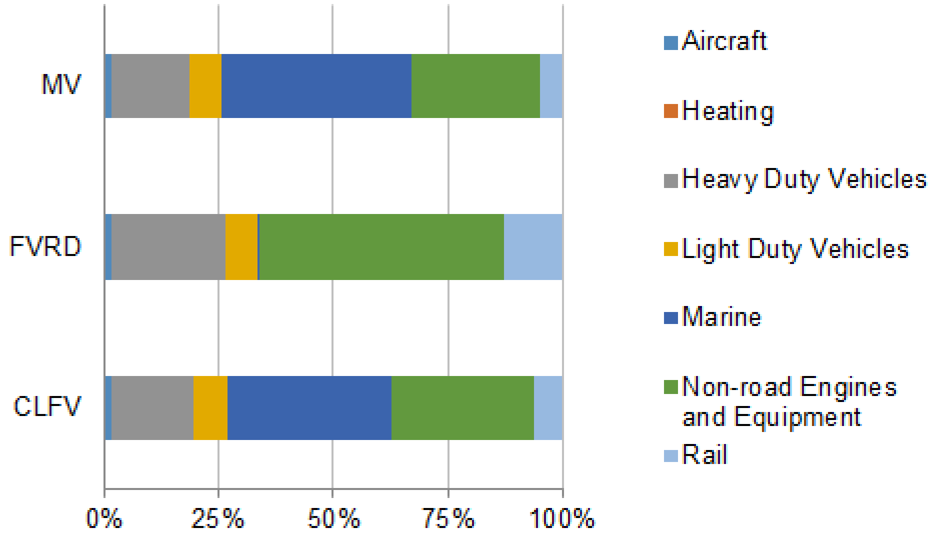

Figure 3-2. 2010 diesel PM emissions contributions from each source category for MV, FVRD, and CLFV.

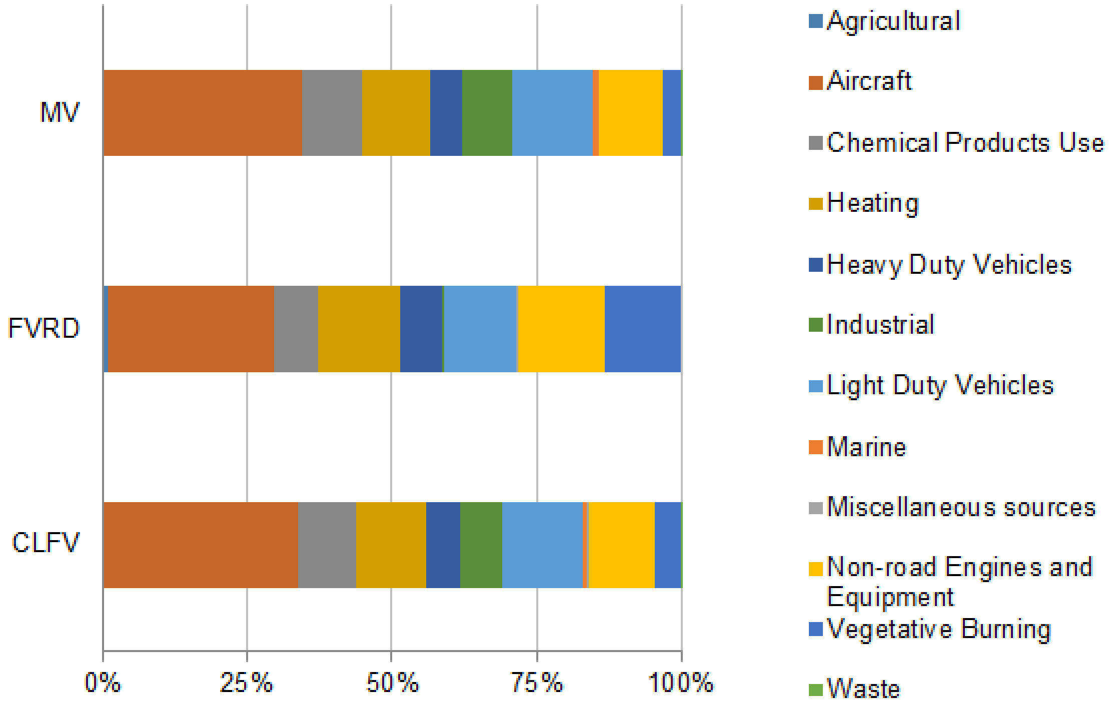

Figure 3-3. 2010 acrolein emissions contributions from each source category for MV, FVRD, and CLFV.

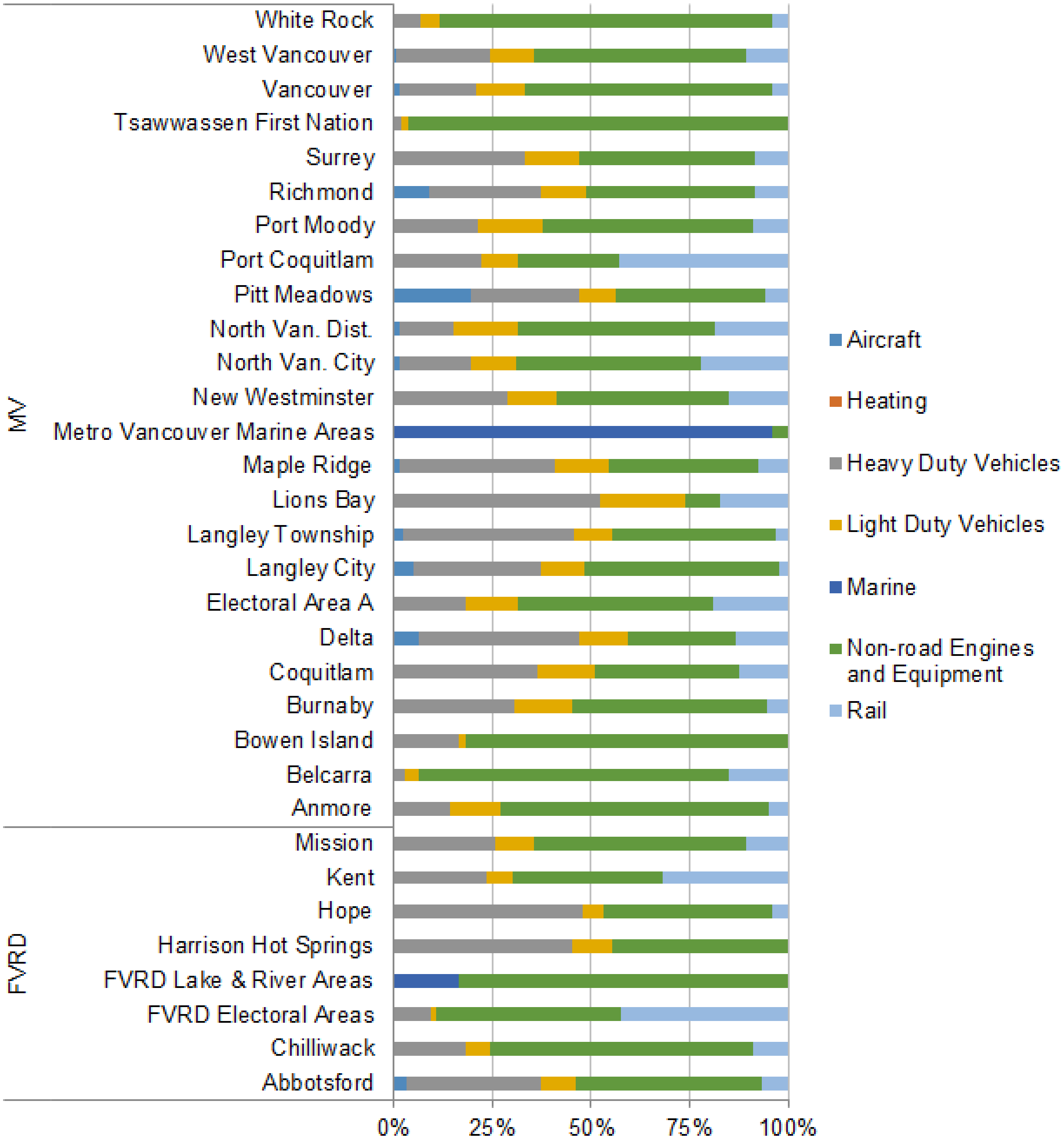

Figure 3-4. 2010 diesel PM emissions contributions from each source category by jurisdiction.

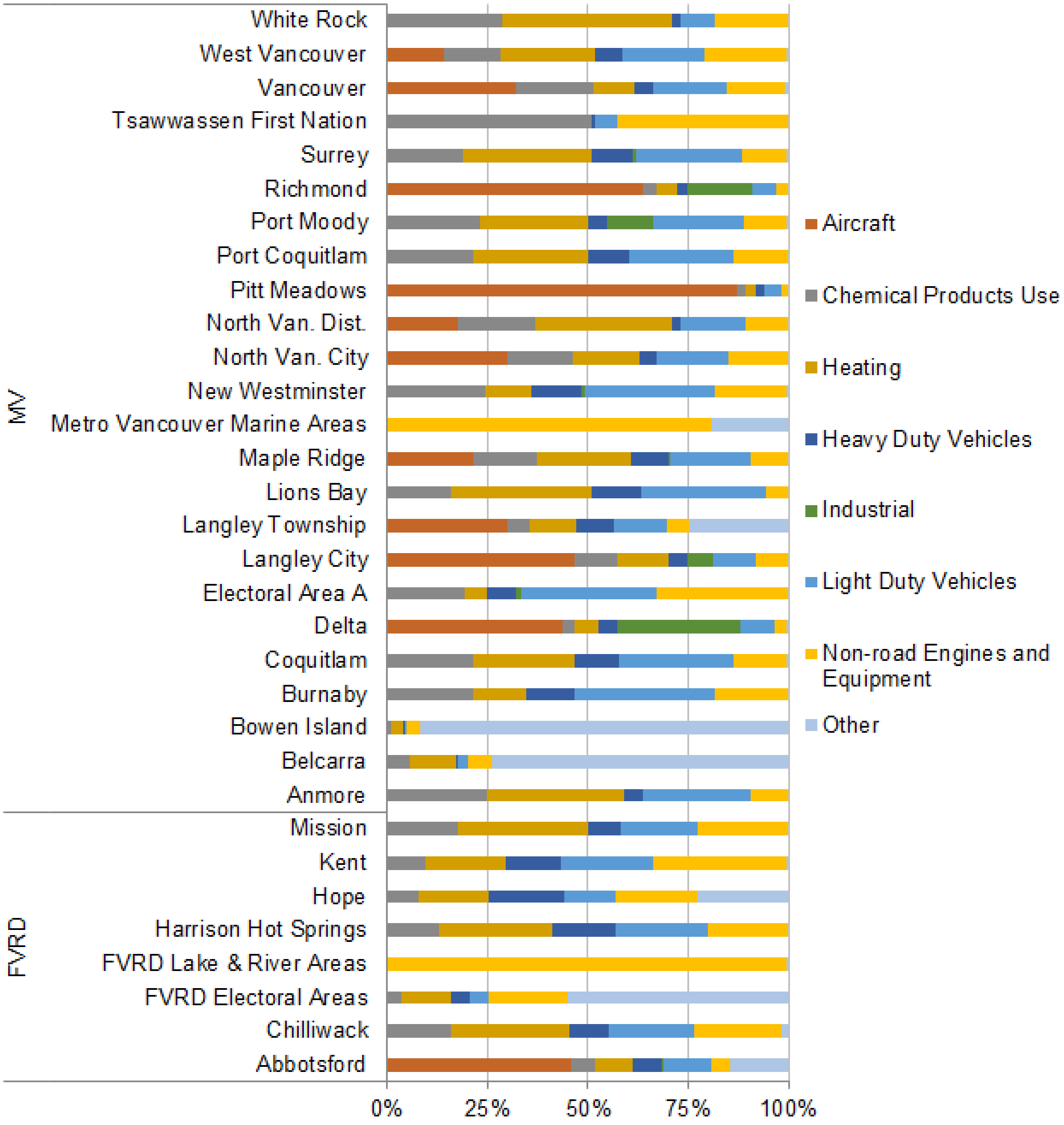

Figure 3-5. 2010 acrolein emissions contributions from each source category by jurisdiction[21].

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[21] The "Other" source category here includes the following source categories: Agricultural, Marine, Miscellaneous sources, Vegetative Burning, and Waste.

The spatial distribution of TAP emissions varies by pollutant. Sample GIS maps (**Figures 3-6 and 3-7**) illustrate the spatial distributions of diesel PM and acrolein emissions across various jurisdictions. Both diesel PM and acrolein are primarily emitted by non-road and on-road mobile sources. However, marine vessels are a significant source of diesel PM emissions, while aircraft are a significant source of acrolein emissions. Therefore, in addition to being in the most highly populated municipalities (e.g., Vancouver, North Vancouver), diesel PM emissions are concentrated in marine areas, while acrolein emissions are elevated in municipalities with airports (e.g., Richmond, Delta, and Langley City).

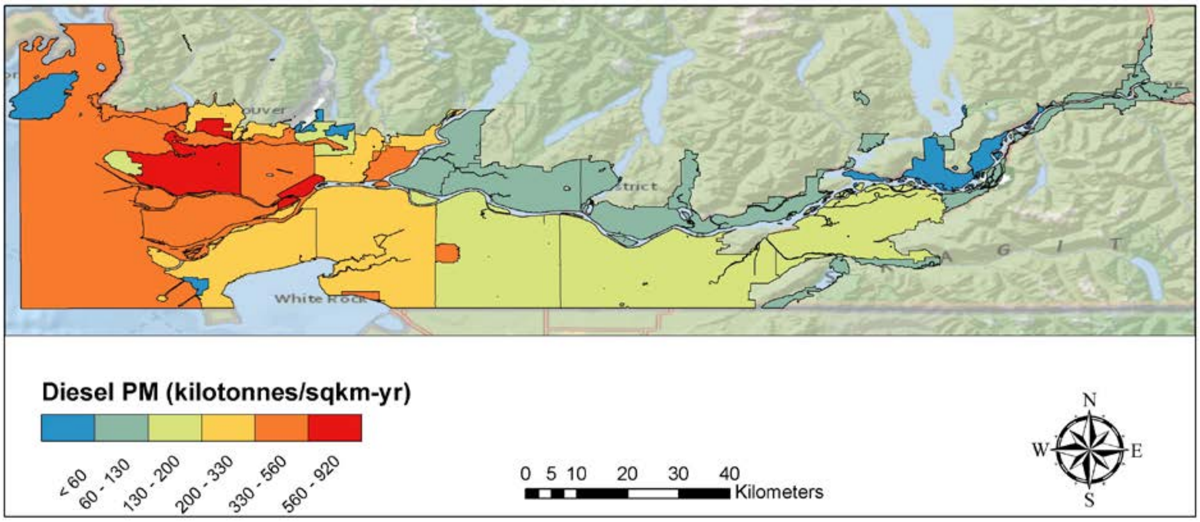

Figure 3-6. 2010 diesel $PM_{2.5}$ emissions density (risk-weighted) by jurisdiction.

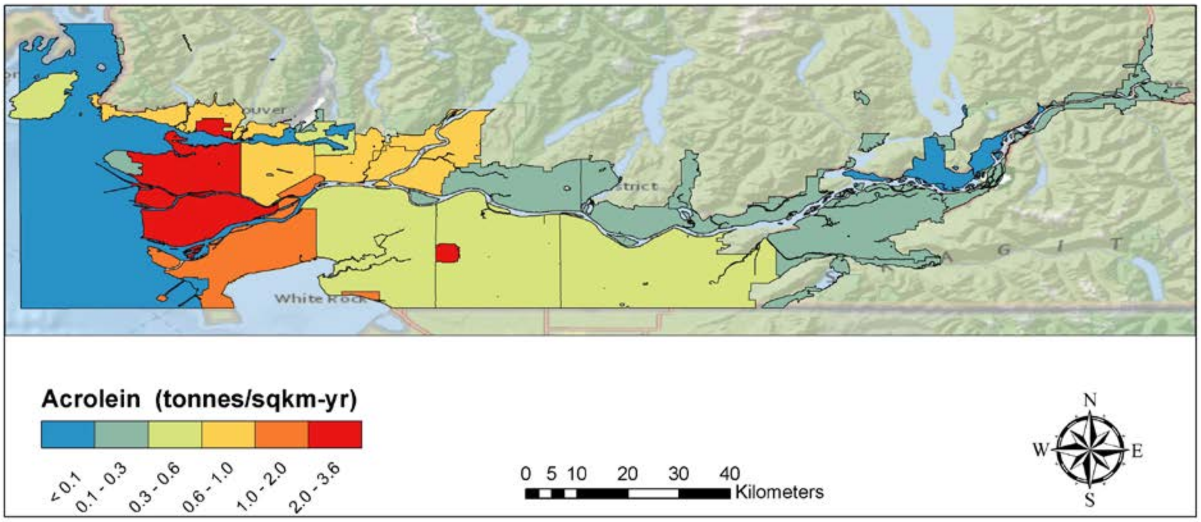

Figure 3-7. 2010 acrolein emissions density (hazard-weighted) by jurisdiction.

In summary, the 2010 TAP emissions inventory indicates that diesel PM is the predominant source of cancer risk-weighted TAP emissions in both MV and the FVRD. Marine vessels are the key source of diesel PM in MV, contributing 41% to the annual total emissions, while non-road equipment is the key source of diesel PM in the FVRD, contributing 53% to the annual total. Agricultural equipment, in particular, accounts for 69% of non-road equipment emissions in FVRD. On-road mobile sources are another important source of diesel PM emissions in both MV and the FVRD, contributing 24% and 32%, respectively, to the annual total.

Similarly, acrolein is the predominant source of noncancer hazard-weighted TAP emissions for in both MV and the FVRD. Aircraft are the most important sources of acrolein emissions across the CLFV, with non-road equipment and on-road vehicles also contributing significantly to total acrolein emissions in the region.

4. Conclusions and Recommendations

A selection of high-priority TAPs were characterized through a health risk assessment and an emissions inventory while considering both cancer risk and noncancer hazard. Of the TAPs studied, those likeliest to exert adverse health impacts in the CLFV were diesel PM (which primarily poses a cancer risk but is also a noncancer health hazard) and acrolein (a noncancer health hazard). Of the 40 TAPs studied, diesel PM was estimated to contribute around two-thirds of the cumulative total excess cancer risk, or 224 per million (of 322 per million). Acrolein contributed about 93% of the cumulative total non-cancer hazard, or 15.2 (of 16.4). The spatial distributions of estimated cancer risks and noncancer hazards in the CLFV are generally consistent with the spatial distribution of emissions sources. Pollutants not included in the health risk assessment are not expected to contribute significantly to either cancer risk or noncancer hazard.

Of the carcinogenic TAPs emitted in the CLFV, diesel PM is emitted in the greatest risk-weighted quantities by a wide margin. Marine vessels, on-road vehicles, and non-road diesel equipment are the most significant sources of diesel PM emissions in the study area, together emitting more than 80% to 90% of the total. After diesel PM, the next most significant cancer risk driver is a group of oxygenated hydrocarbons: formaldehyde, acetaldehyde, and ethylene oxide. Formaldehyde, acetaldehyde, and ethylene oxide are directly emitted into the atmosphere; however, formaldehyde and acetaldehyde are also formed through photochemical reactions in the atmosphere. Thus, a photochemical modeling study would be necessary to fully characterize the sources of these pollutants as observed in the atmosphere. Key hydrocarbons important to cancer risk include benzene, naphthalene, 1,3-butadiene, and ethylbenzene. Each of these compounds is emitted by fossil fuel combustion sources. Key chlorocarbons include carbon tetrachloride (primarily a global background pollutant), 1,2-dichloroethane, chloroform, and PCDD/PCDFs. Wood smoke PM was also estimated to be an important cancer risk driver, but this conclusion is highly uncertain because wood smoke has not yet been formally identified as a human carcinogen and no benchmark risk factor has been generally accepted. Key metals important to risk include arsenic, cadmium, chromium, and hexavalent chromium.

Of the TAPs posing noncancer hazards in the CLFV, acrolein is emitted in the greatest amounts on a hazard-weighted basis. Acrolein is released by a relatively diverse array of sources and is also formed from the photochemical oxidation of 1,3-butadiene in the atmosphere. Aircraft, which produce more than 30% of the total acrolein emissions in the CLFV, represent the most significant source. After acrolein, the most significant noncancer hazards TAPs were found to be diesel PM, formaldehyde, and acetaldehyde.

Further investments in TAP monitoring, analyses, and modeling could improve upon the certainty of these findings, deepen the general understanding of potential health impacts, and/or clarify the most effective strategies for impact mitigation. Recommendations for further steps in the areas of monitoring, modeling, and data analyses are as follows.

Monitoring. Limited ambient data were available to support the analyses of certain key TAPs. Additional monitoring data would significantly improve the certainty of findings in future studies of these key TAPs, and would also facilitate better understanding of the temporal trends and spatial patterns in the ambient concentrations of these TAPs.

- Permanent monitoring of acrolein, formaldehyde, acetaldehyde, and ethylene oxide at multiple sites is recommended. The Burnaby South and Abbotsford sites are recommended as a first priority, primarily because these are the sites where the widest arrays of TAPs are currently being measured.
- Further monitoring to improve characterization of wood smoke and diesel PM is recommended. The relative contributions of wood smoke and fossil fuel combustion to ambient $PM_{2.5}$ can be estimated by following the techniques of Wang et al. (2011), Sandradewi et al. (2008b; 2008a), and Favez, et al. (2010). A relationship between these absorbances and relative contributions of wood smoke and fossil fuel combustion is then applied.
- Regional ambient monitoring of PCDDs, PCDFs, and PAHs is recommended. Measurements of these pollutants on an intermittent schedule (e.g., monitor for a year every five years) may be sufficient to assess these pollutants' risks and trends.

Modeling. Exposure modeling studies would be necessary to quantitatively estimate real-world exposures to environmental TAPs and resultant health impacts. Analyses may be extended to investigate key issues; for example, (1) identify specific sources or source regions exerting relatively high health impacts, (2) estimate the health and economic costs associated with those health impacts, or (3) compare and analyze the relative effectiveness of mitigation strategies under consideration. The Multiple Air Toxics Exposure Study (South Coast Air Quality Management District, 2014) is one such example. In general, the following inputs would need to be developed to model air pollutant concentrations and support an exposure modeling study.

- The emissions inventory would need to be resolved at an appropriate degree of spatial and temporal resolution (e.g., to the Census tract, point, or grid cell level) for one or more TAPs of concern. This resolution can be achieved by applying appropriate spatial surrogate data, such as population, land use, or traffic volumes, to the risk-weighted emissions inventory. If analyses of future conditions and/or comparisons of mitigation options are desired, future projections of emissions patterns will also be necessary. These future projections are based on expected growths in population, urbanization, and development, as well as expected future reductions in emissions due to pre-existing control strategies and policies.
- Meteorological inputs would need to be developed at a suitable time scale and spatial resolution for the area of study and time period of interest. Depending on the type and complexity of exposure modeling to be performed, meteorological inputs may be developed from local meteorological observations or by using a vertically resolved, gridded model such as the Weather Research and Forecasting (WRF) model.
- Employing a land-use regression model may be a feasible alternative for estimating air pollutant concentrations (rather than using emissions inventory and meteorological inputs to drive an air quality model). Assumptions may be applied to use estimated elemental carbon concentrations as surrogates for diesel PM. Examples of such studies

have been published by Henderson et al. (2007), Abernethy et al. (2013), and Urman et al. (2014).

- Receptors would need to be defined as model inputs. These may be represented as spatially resolved but diffuse populations (e.g., Census tract or grid cell level population densities) or point receptors (e.g., locations of sensitive populations, such as schools, nursing homes, or hospitals). Future projections should be consistent with the projections used to develop the emissions inventories.

Data Analyses. Rather than a full exposure modeling study, extended data analyses of spatially resolved emissions and receptor data could be undertaken to qualitatively investigate key issues. For example, spatially resolved population and emissions data may be reviewed in consideration of prevailing winds to identify areas where exposure risks are likely to be largest. Such an analysis may focus on one or two TAPs of greatest concern, such as diesel PM.

Comparing the CLFV to other major metropolitan areas in Canada (Edmonton, Calgary, Toronto, Ottawa, and Montreal) would provide additional context to the risk assessment. Comparisons of major pollutants would likely be possible for most major TAPs, although the availability of proxies for diesel PM and wood smoke PM may be limited.

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Appendix A: Pollutant Selection Criteria

This appendix contains a table documenting the rationale for the selection of toxic air pollutants of concern in the Hazard Identification step of the health risk assessment (see **Table A-1**). In an iterative process that included input from both Sonoma Technology and study stakeholders, pollutants were ranked on a scale of 0 to 5 for a number of categories relevant for performing a health risk assessment, with 5 indicating the highest rating in a category, 1 indicating the lowest rating, and zero indicating the pollutant does not belong in that category (e.g., not a carcinogen in the cancer dose-response category).

Table A-1. Candidate pollutants and their five-star ranking criteria across key selection categories. Greater star numbers indicate higher rankings and more importance for inclusion in a health risk assessment.

Candidate Pollutant	Chemical Abstracts Service Number (CASN)	Regional Emissions Screening	Ambient Data Availability	Cancer Dose Response	Non-Cancer Dose Response	Ranking in Other Studies	Included in Study	Comments
1,1,2,2-tetrachloroethane	79-34-5	*	***		*	*	X	
1,2-dibromoethane	106-93-4	*	**	****	**	**	X	
1,2-dichloroethane	107-06-2	*	***	**		**	X	
1,3-butadiene	106-99-0	***	***	****	***	***	X	
1,4-dichlorobenzene	106-46-7	**	***	***	*	**	X	
Acetaldehyde	75-07-0	**	***	***	***	***	X	
Acrolein	107-02-8	*****	**		*****	****	X	
Acrylonitrile	107-13-1	*	**	***	*	*	X	
Ammonia	7440-62-2	****	***		*	**	X	
Antimony	7440-36-0	*	**	*	*	*	X	
Arsenic	7440-38-2	**	**	***	***	**	X	
Benzene	71-43-2	****	***	****	***	***	X	
Benzyl chloride	100-44-7	*	**	***		**	X	
Cadmium	7440-43-9	**	**	**	***	**	X	
Carbon tetrachloride	56-23-5	*	***	****	**	***	X	
Chloroform	67-66-3	*	***	*	*	*	X	
Chromium	7440-47-3	**	**	*	**	*	X	
Chromium (VI)	18540-29-9	***	**	***	*	***	X	
Dichloromethane	75-09-2	**	***	**	*	**	X	

Candidate Pollutant	Chemical Abstracts Service Number (CASN)	Regional Emissions Screening	Ambient Data Availability	Cancer Dose Response	Non-Cancer Dose Response	Ranking in Other Studies	Included in Study	Comments
Diesel PM	N/A	*****	*	*****	***	*****	X	
Dioxins & Furans (PCDD/PCDF)	N/A	*	X	**	*	*	X	Regional ambient data not available for 2009-2012; NAPS data from other Canadian cities was used to provide estimate.
Ethylbenzene	100-41-4	**	***	**	*	**	X	
Ethylene oxide	75-21-8	*	***	***	**	**	X	
Formaldehyde	50-00-0	***	*	****	****	***	X	
Lead	7439-92-1	*	**	*	*	*	X	
Manganese	7439-96-5	*****	**		****	***	X	
Mercury	7439-97-6	**	X	*	**	*	X	Regional ambient data not available for 2009-2012; Data from Saturna Island CAPMon site was used to provide estimate.
Methanol	67-56-1	**	**	*	**	*	X	
Naphthalene	91-20-3	**	***	***	**	**	X	
Nickel	7440-02-0	**	**	**	***	**	X	
Polycyclic aromatic hydrocarbons (PAH)	N/A	*	X	*		**	X	Regional ambient data not available for 2009-2012; NAPS data from other Canadian cities was used to provide estimate.
Styrene	100-42-5	**	***	*	**		X	
Tetrachloroethylene	127-18-4	**	***	**	*	**	X	
Toluene	108-88-3	***	***	*	***		X	
Trichloroethylene	79-01-6	**	***	*	**		X	
Vanadium	7440-62-2	**	**		X		X	Acute hazard only.

Candidate Pollutant	Chemical Abstracts Service Number (CASN)	Regional Emissions Screening	Ambient Data Availability	Cancer Dose Response	Non-Cancer Dose Response	Ranking in Other Studies	Included in Study	Comments
Vinyl Chloride	75-01-4	*	***	**	*	**	X	
Woodsmoke Particulate Matter	N/A	*****	*	X		*	X	Large regional emissions, regional ambient data available by integrating multiple measurements, including two-wavelength aethalometer black carbon, levoglucosan, and elemental carbon.
Xylenes	1330-20-7	***	***	*	***		X	
1,2,4-Trimethylbenzene	95-63-6		***					
1,3-dichloropropene	542-75-6	**		**				
1-Butene/Isobutene	106-98-9		***					
2,4,6-Trichlorophenol	88-06-2							
2-Methyl-1-butene	563-46-2		***					
2-Methyl-1-Pentene	763-29-1		***					
2-Methyl-2-butene	513-35-9		***					
α-Pinene	80-56-8		***					
Asbestos	12001-29-5			***				
Beryllium	7440-41-7	*	**		*			
Butane	106-97-8		***					
Chlorine	7782-50-5	**			***			
Chlorine dioxide	10049-04-4	****						
Chloropicrin	76-06-2	****						
cis-2-Butene	590-18-1		***					
Coal Dust	N/A							

Candidate Pollutant	Chemical Abstracts Service Number (CASN)	Regional Emissions Screening	Ambient Data Availability	Cancer Dose Response	Non-Cancer Dose Response	Ranking in Other Studies	Included in Study	Comments
Cobalt	7440-48-4	**	**		*			
Copper	7440-50-8		**					
Diethylene glycol monobutyl ether	112-34-5	**			*			
d-Limonene	5989-27-5		***					
Ethylene	74-85-1		***					
Ethylene glycol	107-21-1	**			*			
Glutaraldehyde	111-30-8	***						
Hexachlorobenzene	118-74-1	*		*				
Hydrogen Chloride (hydrochloric acid)	7647-01-0	***			**			
Hydrogen Fluoride (hydrofluoric acid)	7664-39-3	**			*			
Isopentane	78-78-4		***					
Isoprene	78-79-5		***					
Isopropanol	67-63-0	**						
n-Hexane	110-54-3	**	***		*			
PCBs	N/A	*		*				
Pentachlorophenol	608-93-5	*		*				
Pentane	109-66-0		***					
Propionaldehyde	123-38-6	**	**		*			
Propylene	115-07-1	*	***		*			

Candidate Pollutant	Chemical Abstracts Service Number (CASN)	Regional Emissions Screening	Ambient Data Availability	Cancer Dose Response	Non-Cancer Dose Response	Ranking in Other Studies	Included in Study	Comments
Radon	10043-92-2							
Sulfuric acid	7664-93-9	***						
Tellurium	13494-80-9							
Thallium	7440-28-0							
Toluene diisocyanates	N/A	**			*			
trans-2-Butene	624-64-6		***					
trans-2-Pentene	646-04-8		***					
White phosphorus	7723-14-0	***						
Zinc	7440-66-6		**					

Appendix B: Health Risk Assessment Method Details

This appendix documents additional details regarding development of the risk assessment for the CLFV.

B.1 Dose-Response Assessment

- All risk and hazard estimates displayed in Figures 2-2, 2-3, 2-4, 2-5, and 2-6 of the body of the report are based on the unweighted average of dose-response factors across all available agencies. If only one dose-response factor is available, it was used; if four were available, then an average of those four dose-response factors was applied for risk and hazard characterization in the summary figures. Most health risk assessments use a single agency set of dose-response factors (e.g., OEHHA only).We note that individual agency-specific estimates of risk are shown in Figures 2-8 and 2-9; these estimates were provided in a spreadsheet to document the sensitivity of risk estimates to the choice of dose-response factor.
- The wood smoke PM dose-response factor was taken from Lewtas (1988). This value has not been adopted by any U.S. or Canadian agencies for dose-response screening and should be considered significantly less reliable than other dose-response values. It has been applied in toxic air pollutant risk assessments performed for the Puget Sound Clean Air Agency (Keill and Maykur, 2003; Puget Sound Clean Air Agency and the University of Washington, 2010).
- U.S. EPA’s IRIS provides a range for the dose-response factor for benzene. OAQPS selected one of the endpoints of the range. For calculation purposes, IRIS was assigned the other endpoint of the range. This has a small impact (~3%) impact on the average risk values displayed in Figures 2-2 through 2-6.

B.2 Inhalation Exposure

- Ambient TAP measurements were drawn from monitoring sites in the CLFV that measured ambient concentrations at any time between 2009 and 2012.
- Each pollutant at each site was required to have at least one complete year (>75%) of monitoring data to be considered valid.
  - If at least one valid annual mean was available, data were averaged for that site-pollutant combination at the site to generate the final average concentration estimate. Data below detection limits were used as reported; no MDL/2 or other substitutions were applied. Multiple parameters at sites with 100% of data below the average detection limit had a few reported concentrations that generated average concentrations above a value of zero.
  - Many site-pollutants have different periods of operation during the study period. In other words, one site will operate from 2009-2010, another will monitor the same pollutant for 2009-2012, and another will monitor for 2012 only. Different temporal

averaging periods will potentially affect average concentrations because of differences in climatology and meteorology.

  - This difference in sampling periods introduces a confounding element in comparisons between pollutants. However, we chose to allow this in order to include more data available in the analysis, rather than exclude information by restricting to identical sampling periods across sites and pollutants.

- Three target pollutant types were not routinely measured in the CLFV:
  - Polychlorinated dibenzodioxins (PCDDs, or dioxins) and polychlorinated diobenzofurans (PCDFs, or furans). For exposure estimates for PCDD/PCDFs, Metro Vancouver generated concentration estimated based on a 2009-2011 average from NAPS monitoring sites in four Canadian cities: Edmonton, Winnipeg, Toronto, and Montreal. **Table B-1** details the individual PCDDs/PCDFs and their toxicity weighting.
  - Polycyclic aromatic hydrocarbons (PAHs). For exposure estimates for PAHs, Metro Vancouver generated concentration estimated based on a 2009-2011 average from NAPS monitoring sites in four Canadian cities: Edmonton, Winnipeg, Toronto, and Montreal. **Table B-2** details the individual PAHs and their toxicity weighting.
  - Mercury. For gaseous mercury, data were acquired from Environment Canada's CAPMon site on Saturna Island, west of Vancouver. Because the exposure data for these pollutants were not measured within the Canadian Lower Fraser Valley region, the concentrations for these pollutants are marked as very low certainty in all figures.

Table B-1. List of PCDDs and PCDFs included in the lumped PCDD/PCDF pollutant category and their toxicity weighting factors relative to 2378-TCDD.

Pollutant	Abbreviation	Toxicity Weighting Factor (NATO I-TEF)
2,3,7,8-Tetrachlorodibenzodioxin	2378-TCDD	1
1,2,3,7,8-Pentachlorodibenzo-p-dioxin	12378-PeCDD	0.5
1,2,3,4,7,8-Hexachlorodibenzo-p-dioxin	123478-HxCDD	0.1
1,2,3,6,7,8-Hexachlorodibenzo-p-dioxin	123678-HxCDD	0.1
1,2,3,7,8,9-Hexachlorodibenzo-p-dioxin	123789-HxCDD	0.1
1,2,3,4,6,7,8-Heptachlorodibenzo-p-dioxin	1234678-HpCDD	0.1
Octachlorodibenzodioxin	OCDD	0.001
2,3,7,8-Tetrachlorodibenzodioxin	2378-TCDF	0.1
1,2,3,7,8-Pentachlorodibenzofuran	12378-PeCDF	0.05
2,3,4,7,8-Pentachlorodibenzofuran	23478-PeCDF	0.5
1,2,3,4,7,8-Hexachlorodibenzofuran	123478-HxCDF	0.1
1,2,3,6,7,8-Hexachlorodibenzofuran	123678-HxCDF	0.1
2,3,4,6,7,8-Hexachlorodibenzofuran	234678-HxCDF	0.1
1,2,3,7,8,9-Hexachlorodibenzofuran	123789-HxCDF	0.1

Pollutant	Abbreviation	Toxicity Weighting Factor (NATO I-TEF)
1,2,3,4,6,7,8-Heptachlorodibenzofuran	1234678-HpCDF	0.01
1,2,3,4,7,8,9-Heptachlorodibenzofuran	1234789-HpCDF	0.01
Octachlorodibenzofuran	OCDF	0.001

Table B-2. List of PAHs included in the lumped PAH pollutant category and their toxicity weighting factors relative to benzo[a]pyrene.

Pollutant	Abbreviation	Toxicity Weighting Factor[a]
Acenaphthene	ANA	0.001
Acenaphthylene	ANTL	0.001
Anthracene	ANTH	0.01
Benz(a)Anthracene	B[a]ANTH	0.1
Benzo(a)Pyrene	B[a]P	1
Benzo(b)Fluoranthene	B[b]FLAN	0.1
Benzo(e)Pyrene	B[e]P	0.01
Benzo(g,h,i)Perylene	B[g,h,i]PERY	0.01
Benzo(k)Fluoranthene	B[k]FLAN	0.1
Chrysene	CH	0.01
Dibenz(a,c) & (a,h)Anthracene	D[a,c]AN, D[a,h]AN	1
Fluoranthene	FLAN	0.001
Fluorene	FL	0.001
Indeno(1,2,3-cd)Pyrene	I[1,2,3-cd]PY	0.1
Perylene	PERY	0.001
Phenanthrene	PH	0.001
Pyrene	PY	0.001

[a] Malcolm and Dobson (1994).

- The target TAPs were sampled, analyzed, and measured using methods that are appropriate for each pollutant. Given the different chemical and physical characteristics of the target pollutants, there were multiple methods employed. The main descriptions of pollutant methods are listed by pollutant category:
  - VOCs – includes hydrocarbons and chlorinated hydrocarbons.
  - PVOCs – includes hydrocarbons, chlorinated hydrocarbons, oxygenated hydrocarbons.
  - Metals – speciated metals measured by x-ray fluorescence (XRF) or inductively coupled plasma mass spectrometry (ICP-MS).
  - BC – measured by aethalometers.

  - EC – measured at speciated $PM_{2.5}$ sites (same as metals).

B.3 Woodsmoke and Diesel PM

- Woodsmoke PM and diesel PM cannot be measured directly. Two methods were used to apportion wood smoke PM and diesel PM from BC and EC measurements in the CLFV.
- In the primary approach, aethalometer measurement channels of UV and BC were used to characterize black and "brown" carbon. The UV-BC differential is a good proxy of wood smoke PM. The remaining BC is then attributed to diesel PM. Statistics for these calculations are available in an Excel spreadsheet for reference. Note that Magee AE22 aethalometers have a known optical saturation issue which should be corrected, but their data have not been corrected for this study. Saturation of the filter tape leads to underestimation of true BC (and therefore diesel PM) concentrations.
- In the secondary approach, measurements of levoglucosan, BC, and EC were used as surrogates to estimate the concentrations of wood smoke PM and diesel PM.
  - Both wood smoke PM and diesel PM contain BC, but only wood smoke PM contains levoglucosan. The amount of levoglucosan emitted is highly dependent on the type of wood burned.
  - Survey data were taken from a British Columbia Ministry of Environment-commissioned survey (Mustel Group, 2012) that reported the prevalence of different firewood usage; Douglas Fir, Pine, Alder, Spruce, Birch, Hemlock, and Cedar were the most prevalent woods burned.
  - Literature estimates of levoglucosan:EC ratios from Fine et al. (2004a; 2004b) and Mazzoleni et al. (2007) and a weighted average of the these ratios for different woods burned in British Columbia from the Mustel Group's 2012 survey were applied to generate an average levoglucosan:EC ratio of 1.06 and a levoglucosan:wood smoke PM ratio of 0.119. These ratios are dependent on the assumptions that woods used in BC are comparable to the Vancouver area and that the emissions estimates from U.S. woods are representative of Vancouver woods. The numbers from these calculations are provided in Excel spreadsheets for reference.
  - Levoglucosan measurements were available at the Burnaby and Abbottsford sites. Data from those two sites were used to estimate Metro Vancouver and FVRD wood smoke PM values. Burnaby had a full year of data, while Abbottsford had no summer measurements of levoglucosan. The winter-summer ratio at Burnaby was estimated and a seasonally weighted average was applied with this ratio to generate annual mean levoglucosan concentrations at Abbottsford.
  - Using the levoglucosan:EC ratio, annual wood smoke PM concentration estimates of 0.65 $\mu g/m^3$ and 1.26 $\mu g/m^3$ were derived for the Metro Vancouver and FVRD, respectively.
- After accounting for the wood smoke PM portion of EC based on the levoglucosan:EC ratio, all remaining BC/EC was assumed to be diesel PM.

- A BC:EC ratio of 1.63 was calculated based on empirical comparisons of BC:EC at sites with collocated aethalometer and filter measurements; this is consistent with the range of BC:EC ratios seen at other sites reported in Appendix 1 of the *Report to Congress of Black Carbon* (U.S. Environmental Protection Agency, 2012).
- Other studies have also considered the EC:diesel PM ratio, which ranged between values of 1.06 to 1.95 as discussed in the South Coast Air Quality Management District's *MATES-III: Multiple Air Toxics Exposure Study* (South Coast Air Quality Management District, 2008). In the previous Levelton Consultants (2007) study, a value of 1.06 was chosen for the EC:diesel PM ratio, based on the MATES II EC:diesel PM ratio.
- An updated draft report for MATES-IV: Multiple Air Toxics Exposure Study[1] lowered the EC:DPM ratio to a value of 0.82. This most recent values was used for this study. We note that this updated values does not include possible contributions from secondary organic aerosol formation as discussed in May et al. (2014).
- The two approaches (aethalometer vs. levoglucosan) gave similar wood smoke PM estimates at most sites. Using the UV-BC differential as our primary estimate, estimated diesel PM concentrations at sites range from 0.57 µg/m^3 (Abbotsford YXX and Port Moody) to 1.07 µg/m^3 (North Vancouver – 2nd Narrows) across the CLFV.

B.4 Population Representativeness

The population represented by the nearest monitor was calculated using Thiessen Polygons and the underlying population at the dissemination block layers from Statistics Canada (2011). Different parameters were measured at different sets of sites, so polygons were calculated for the four major pollutant measurement methods: aethalometer BC, VOCs, PVOCs, and metals.

Figures B-1 through B-4 show the population representativeness of each measurement site for the four parameter groups. Polygons indicate the area where each site is the closest. Numbers inside the polygon indicate the 2011 population within that area.

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[1] http://www.aqmd.gov/home/library/air-quality-data-studies/health-studies/mates-iv .

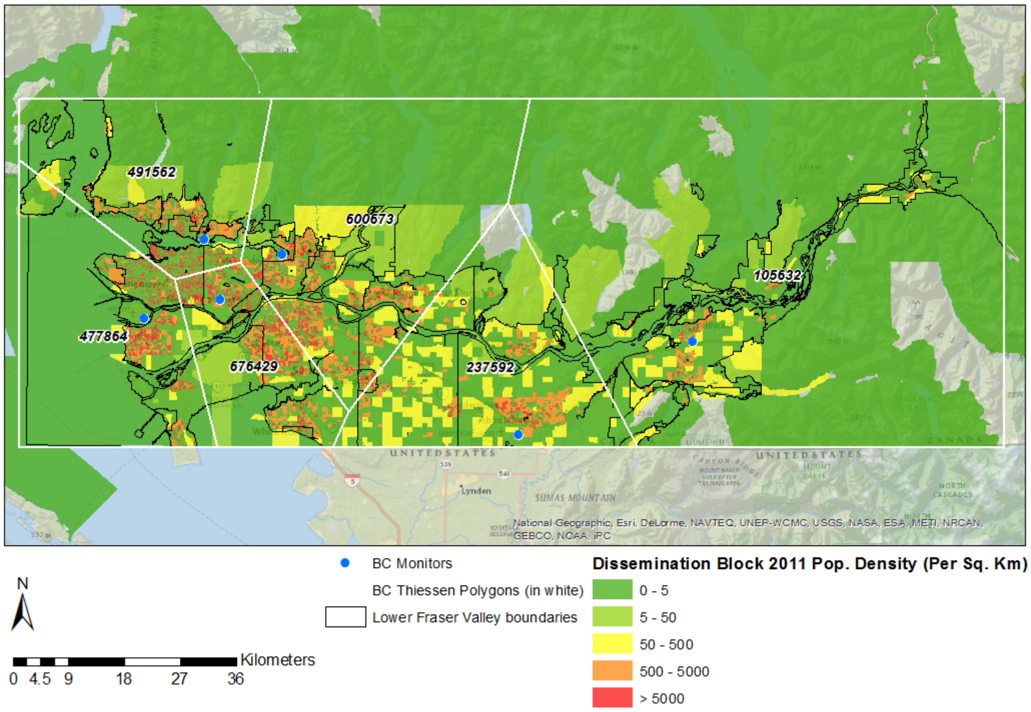

Figure B-1. Population representativeness of aethalometer BC in the CLFV study domain.

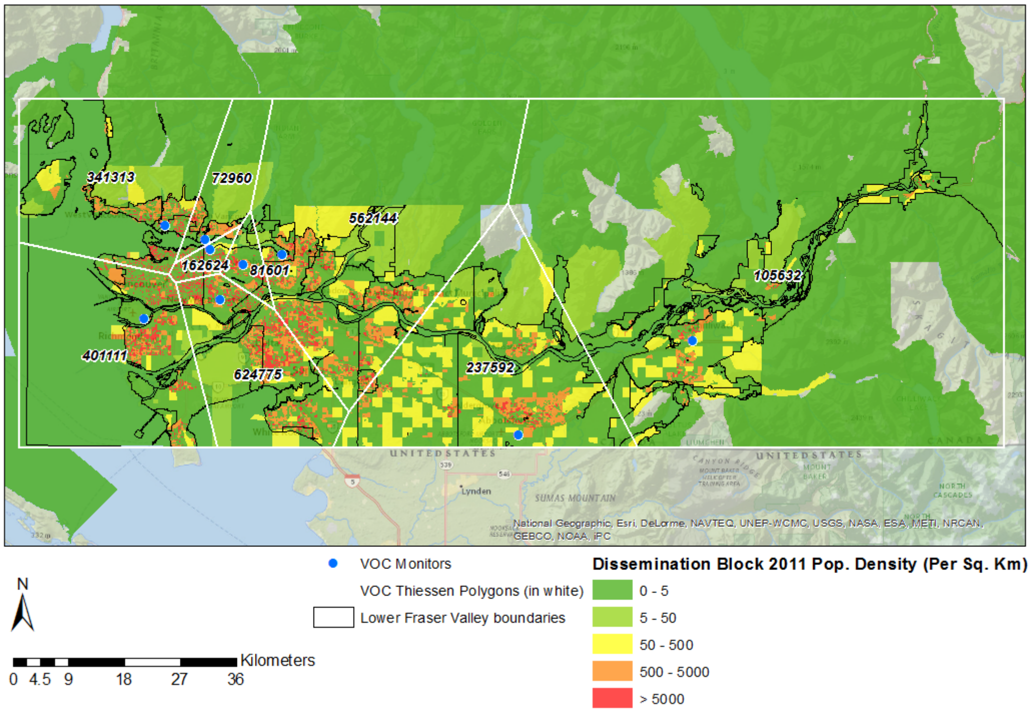

Figure B-2. Population representatives of VOC monitors in the CLFV study domain. Some of the monitors are special study locations that did not operate the entire time.

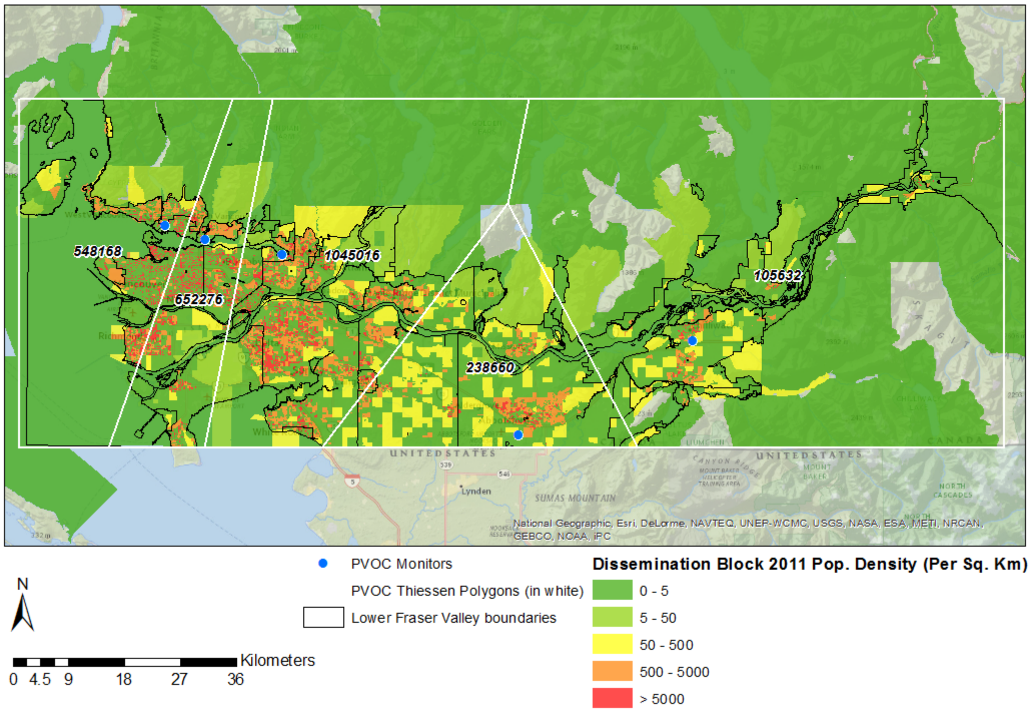

Figure B-3. Population representatives of PVOC monitors in the CLFV study domain.

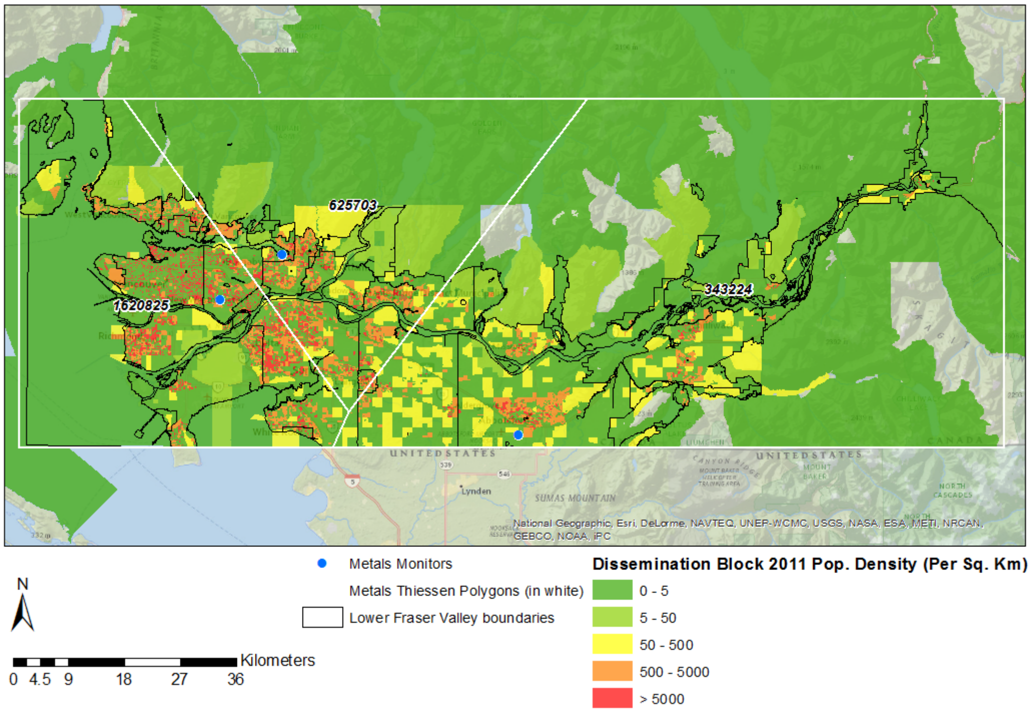

Figure B-4. Population representativeness of metals monitors in the CLFV study domain.

The original intention was to use the Thiessen Polygon approach to apportion risk across each site as a function of population within the CLFV. However, this approach was not implemented because of inconsistencies in parameter monitoring. For example, PVOC monitoring sometimes included key species such as formaldehyde, acetaldehyde, and acrolein, but did not at other times. Acrolein and acetaldehyde were measured at two sites, formaldehyde at only one. Of the ten most important pollutants for cancer risk, only diesel PM, benzene, 1,3-butadiene, and naphthalene are measured at more than three sites and have spatially varying concentrations.

Diesel PM has the exact same average (224-in-a-million) if the sites have an unweighted average or a population-weighted average. Most other pollutants have relatively small amounts of spatial variability such that the final average risk numbers are insensitive to the population weighting. Thus, we did not population-weight the average risk numbers for the general results.

B.5 References

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Appendix C: Exploratory Source Apportionment Analysis

A key objective of a health risk assessment is to identify the underlying emissions sources responsible for impacting human health. Diesel PM has been characterized as the most important contributor to excess cancer risk in the CLFV in both Levelton (2007) and the current health risk assessments. Woodsmoke PM is an important contributor to overall PM mass. Both sources (wood smoke and diesel vehicles) emit black carbon (BC), which can be used as a proxy for their concentrations in the ambient atmosphere. In addition, aethalometer ultraviolet (UV) channel data, and the difference in UV and BC channel, can be useful to identify the relative portion of wood smoke PM.

Source apportionment can be a useful tool for identifying and quantifying underlying sources emitting key toxic air pollutants using ambient data. In this analysis, positive matrix factorization (PMF) was used to analyze Aethalometer data (BC and UV) measured across the CLFV, and determine whether wood smoke and diesel emissions can be identified as sources. PMF relies upon spatio-temporal differences in ambient data that can be used to identify unique emissions sources. Diesel PM is primarily from transportation and port sources in the CLFV; emissions are higher during the daytime, on weekdays, and during seasons with heavy port traffic. Woodsmoke PM is most likely a result of residential biomass burning during the cooler months, especially in the inland valleys; emissions are typically higher during the nighttime, on weekends, and during the cooler seasons.

The goal of this analysis was to use source apportionment of BC and UV at the five sites across the CLFV to support the health risk assessment with an independent and levoglucosan-free evaluation of the diesel PM:wood smoke PM ratio of contributions to BC.

The preliminary analysis encompassed the following steps:

1. Process the hourly data (both BC and UV channels) to format it for use in PMF, and to identify the best time period for which to perform source apportionment. The date range selected maximizes the available data and minimizes substitution of missing or invalid data.
2. Generate uncertainty estimates for each hourly concentration value.
3. Perform exploratory analysis of the data set with the EPA PMF v5.0 GUI to assess site-to-site correlations, signal-to-noise ratios, seasonal patterns, and diurnal patterns; identify potential outlier events that unduly influence results.
4. Perform initial source apportionment with two- and three-factor solutions. Identify potential factors that may represent wood smoke PM and diesel PM.

The 2011-2012 time period was selected for PMF analysis. More than 96 % of the expected data values were available for the selected date range. Only complete samples (i.e., BC and UV data were available at all sites) were included; therefore, no data substitutions were required. The T045 site was excluded due to its limited data availability. **Table C-1** provides basic summary statistics for the selected time period.

Table C-1. Summary statistics by site for the selected analysis time period (February 11, 2011 to December 28, 2012). Concentrations are in µg/m^3.

	UV				
	N. Vancouver-2nd Narrows (T006)	Port Moody (Rocky Point Park) (T009)	Chilliwack (T012)	Burnaby South (T018)	Richmond-YXX (T031)
N	15825	15825	15825	15825	15825
Min	0	0	0	0	0
Max	9.30	12.50	3.50	3.30	7.80
Mean	0.83	0.51	0.70	0.53	0.72
Median	0.60	0.40	0.50	0.40	0.50
St Dev	0.75	0.43	0.74	0.50	0.76
	BC				
	N. Vancouver-2nd Narrows (T006)	Port Moody (Rocky Point Park) (T009)	Chilliwack (T012)	Burnaby South (T018)	Richmond-Airport (T031)
N	15825	15825	15825	15825	15825
Min	0	0	0	0	0
Max	8.60	14.00	3.80	2.70	4.30
Mean	0.81	0.46	0.62	0.49	0.68
Median	0.60	0.40	0.40	0.40	0.50
St Dev	0.75	0.39	0.65	0.47	0.72

For the UV and BC concentrations, we used the following uncertainty calculation:

$$\text{Uncertainty} = 0.1\ \mu g/m^3 + (0.1 * \text{concentration})$$

For the (UV-BC) parameter, we used the following uncertainty calculation:

$$\text{UV-BC uncertainty} = ((\text{UV uncertainty}^2)+(\text{BC uncertainty}^2))^{0.5}$$

Summary of Results

PMF analysis was unable to identify a diesel versus wood smoke signal in either a two- or three-factor solution. As shown in **Figure C-1**, the two-factor PMF solution separated the N. Vancouver and Chilliwack sites into two distinct factors; BC and UV (or the UV-BC difference) were not apportioned into distinct factors. A three-factor solution results in a third factor where most of the Richmond Airport data is apportioned. BC and UV were well predicted (R2 > 0.9) at N. Vancouver, Chilliwack, and the Richmond airport sites. The results were stable and consistent, and did not change greatly for any of the sensitivity runs performed: varying the parameters (including BC and UV, BC and UV and their difference, BC and the UV-BC difference); including only the winter months; modeling 3-hr average concentrations; and modeling 6-hr average concentrations.

The results are likely due to the strong collinearity between BC and UV at each site (**Figure C-2**), as well as the lack of a relationship in BC between sites (**Figure C-3**). There are distinct differences in the diurnal profile between N. Vancouver and Chilliwack (**Figure C-4**), which suggests a different proportion of wood smoke and diesel emissions. While elevated morning hourly BC concentrations were apparent at all sites, the peak at Chilliwack was earlier. Concentrations remained high at N. Vancouver throughout the day, while the other sites experienced lower midday BC concentrations. In addition, elevated hourly BC concentrations were found at Port Moody, Chilliwack, and Burnaby South in the later evening (7-11 p.m.). While the UV-BC difference does vary seasonally (**Figure C-5**), PMF was unable to differentiate. This may be a result of the inadequate data precision or more likely the overwhelming difference in BC and UV patterns across the CLFV.

The null result for the source apportionment motivated an alternate approach to estimate woodsmoke PM and diesel PM using the differential in UV and BC concentrations. Assessing the differential in UV-BC concentrations can provide information on the fraction of BC attributable to woodsmoke PM. This method is described in the main body of the report.

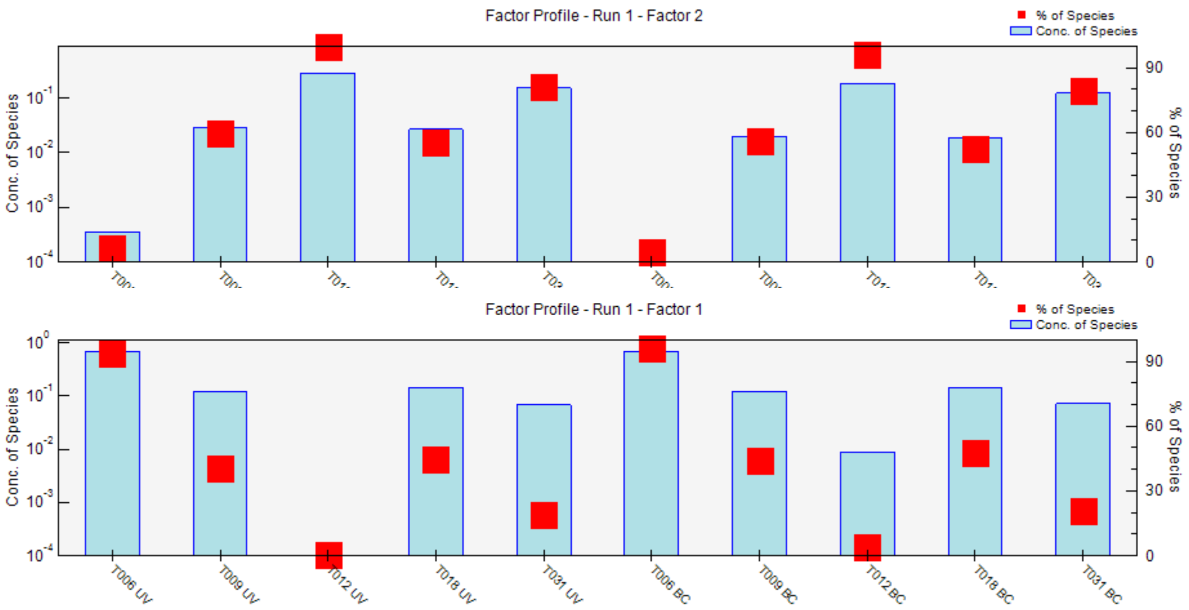


Figure C-1. PMF solution with two factors: Factor 1 is dominated by N. Vancouver and Factor 2 is dominated by Chilliwack. The other three sites are evenly divided between the two factors.

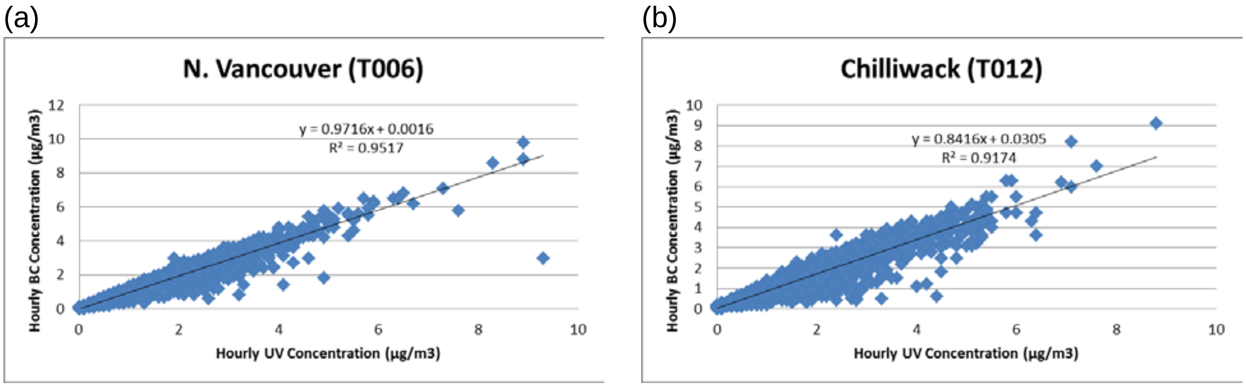


Figure C-2. Strong relationship between hourly concentrations of UV and BC at N. Vancouver (a) and Chilliwack (b).

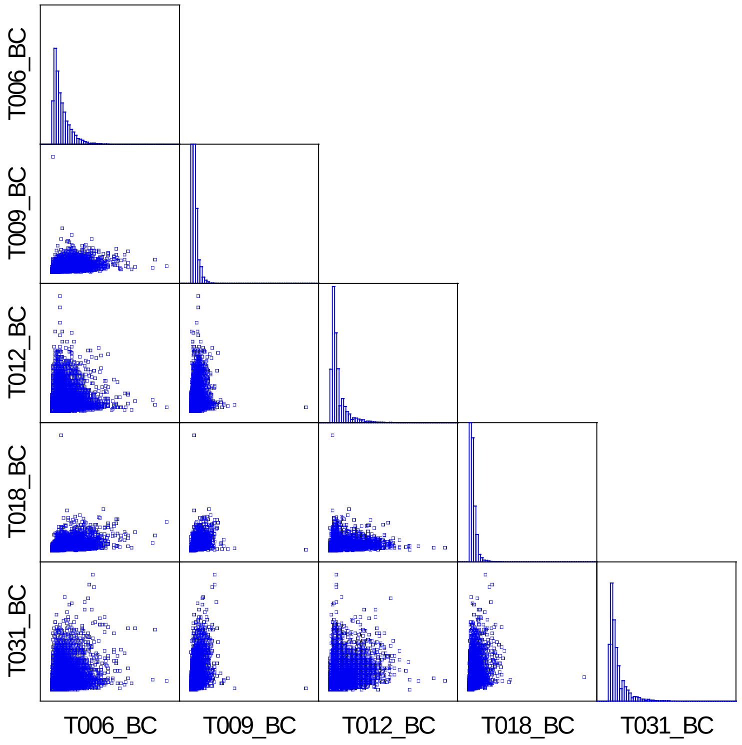


Figure C-3. Scatterplot matrix showing the lack of a relationship between hourly BC concentrations for each pair of sites.

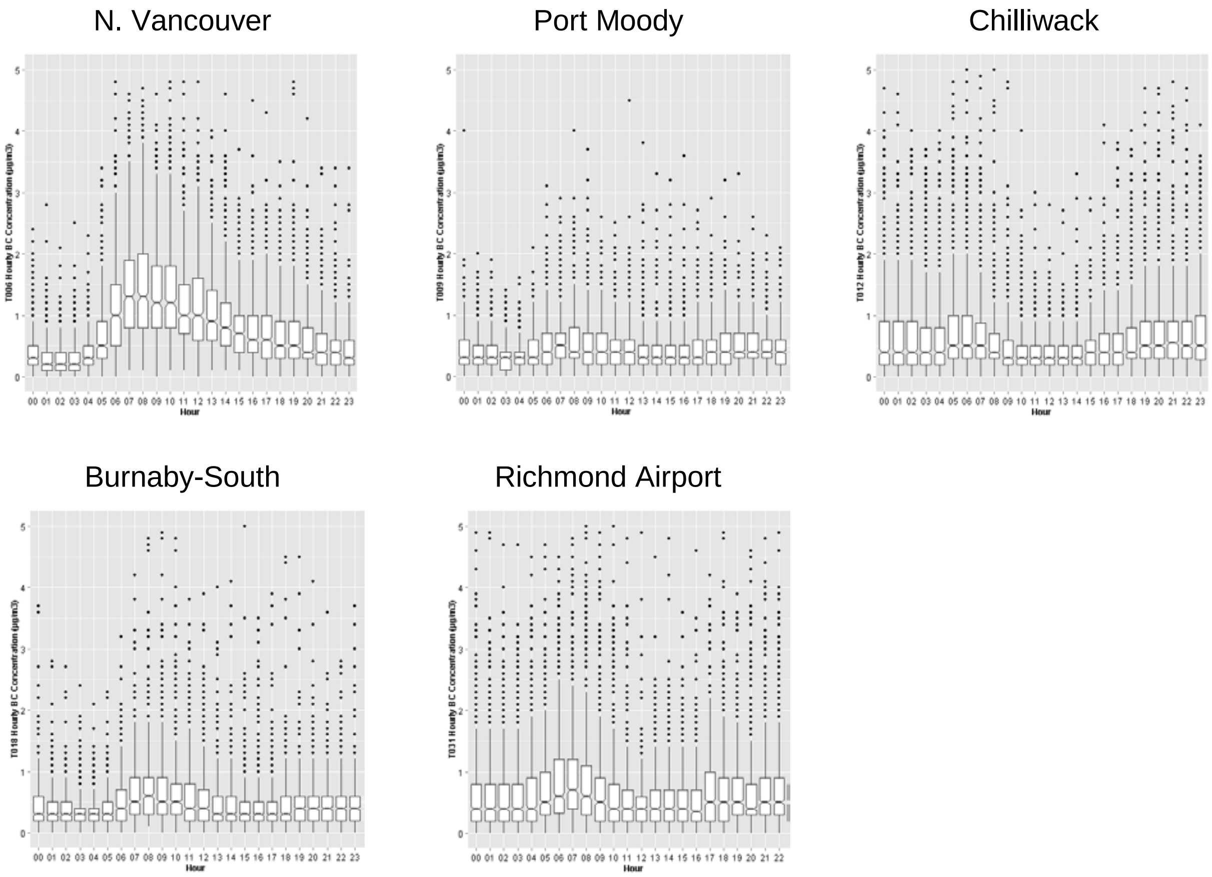
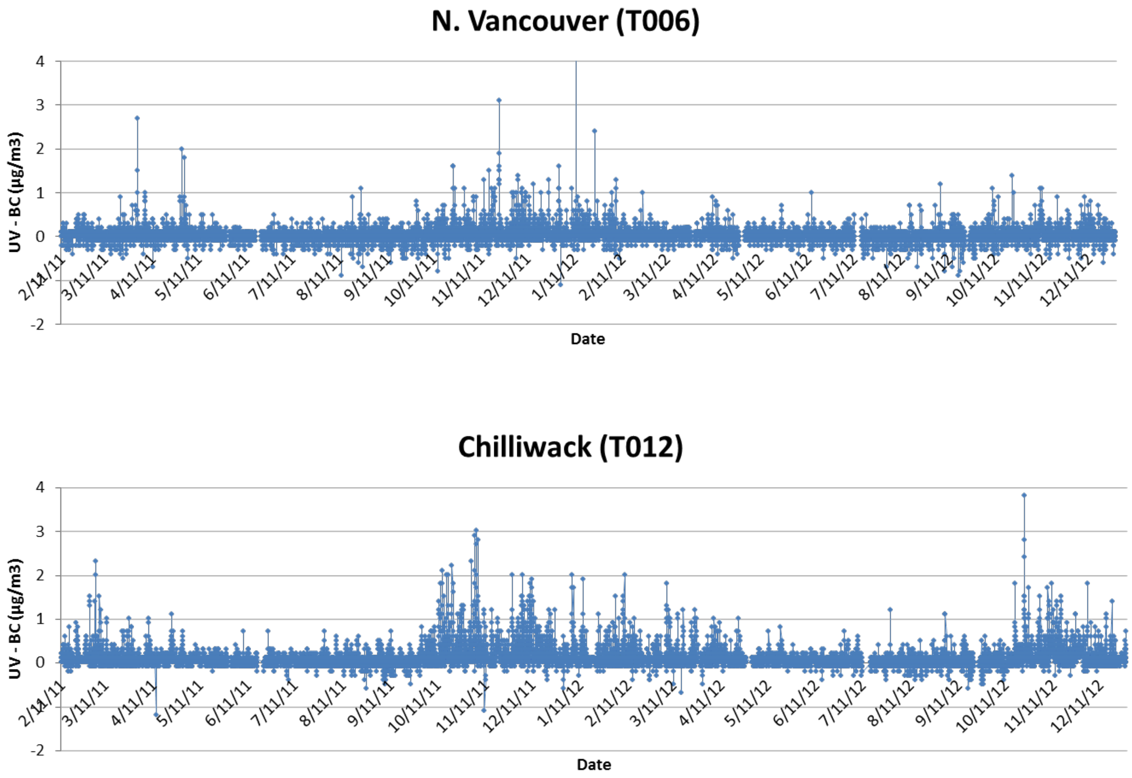


Figure C-4. Hourly BC concentrations at each site.



(a)

(b)

Figure C-5. Time series of the difference in hourly UV and BC concentrations at the N. Vancouver (a) and Chilliwack (b) sites.

Reference

Levelton Consultants, Ltd. (2007) Air toxics emission inventory and health risk assessment. Summary report prepared for the Greater Vancouver Regional District Policy and Planning Department, Burnaby, BC, and Environment Canada, Vancouver, BC, 404-0423 Nov. 30.

Appendix D: TAP Emissions Inventory Development

The appendix documents the general procedures used to develop the refined 2010 toxic air pollutant (TAP) emissions inventory for the CLFV, including the emission estimation approaches applied to each of the 114 source categories (i.e., combination of sector/subsectors) and the data sources relied upon. The general steps are summarized as follows:

1. **Identify applicable emissions calculation approaches and hierarchy** ordered as follows:
    a. Existing TAP emissions estimates (e.g., MOVES outputs, NPRI data)
    b. An emissions factor-based approach, in which emissions were estimated by multiplying available TAP emissions factors from U.S. EPA and other data sources by corresponding activity data provided by Metro Vancouver.
    c. A TAP augmentation approach (U.S. Environmental Protection Agency, 2013), in which TAP emissions were calculated by multiplying the appropriate criteria air pollutant (CAP) emissions (i.e., volatile organic compound [VOC] or fine particulate matter [$PM_{2.5}$]) by established TAP-to-CAP ratios for various pollutants. These ratios vary among sources and were applied based on SCCs.
    d. Chemical speciation of CAP emissions, which disaggregate VOC or $PM_{2.5}$ emissions into individual chemical compounds (including TAPs) using chemical speciation profiles available from U.S. EPA's SPECIATE 4.4 database and a previous VOC study for the CLFV (SNC Lavalin, 2012).
2. **Collect/review raw data**, including activity and data CAP emissions, emissions factors, TAP-to-CAP ratios for TAP augmentation, and speciation profiles and cross-reference table.
3. **Select appropriate data/approaches hierarchy** for each source category (see **Table D-1**) based on data availability.
4. **Apply assigned approaches** to estimate mass-only TAP emissions for each source category. The final results were reconciled based on the selection hierarchy displayed in Table D-1.
5. **Filter emissions for 39 priority TAPs and 11 additional TAPs**[1]**.** The results were filtered to exclude any pollutants other than the 50 selected TAPs of interest.
6. **Compile mass-only TAP EI by source category and municipality.** The filtered emissions from the fifth step are not uniform in terms of spatial resolution because the raw activity/CAP emissions provided by Metro Vancouver vary in spatial extent (e.g., facility, airport, municipality, MV/FVRD, CLFV) among source categories. In this step, all emissions estimates were converted from their native spatial resolution to the municipality level following the methods below:

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[1] Metro Vancouver designated 12 additional TAPs; however, no chlorine dioxide emissions sources were identified in the CLFV.

   a. *Spatial aggregation*. For sources at sub-municipality level (e.g., point sources by facility, aircrafts by airport), the emissions are aggregated to municipality level according to their locations.
   b. *Spatial disaggregation*. For sources at the regional level (e.g., on-road vehicles, marine vessels, and wastewater treatment facilities at CLFV level; non-road equipment, cremation, and meat cooking sources by MV/FVRD), spatial allocation factors derived from the 2010 CLFV CAP emissions inventory were used to allocate regional TAP emissions to the municipality level. For each TAP and source category, the spatial allocation factor for a municipality was calculated as the ratio of municipality-level CAP emissions to the corresponding regional CAP emissions. In this process, the spatial distribution of emissions for a TAP is assumed to be consistent with that of its parent CAP. For example, diesel PM emissions from marine vessels are assumed to have the same spatial distribution as total $PM_{2.5}$ emissions from that source.

7. **Apply risk factors to mass-only TAP EI** from previous step to develop the risk-weighted TAP EI. To be conservative, the risk factor applied to each TAP is the minimum value in the chronic dose-response factors assembled from various sources for the HRA task. For emissions ranking purposes, only the cancer-risk factor was applied if a TAP has both cancer and noncancer risk factors. To calculate risk/hazard weighted emissions, the actual mass emissions of each TAP are divided by the corresponding risk/hazard factor. Taking diesel PM and benzene as examples, the 1-in-a-million cancer risk level for diesel PM is 1.408E-3 [μg/m^3]^1, while the threshold for benzene has a higher value of 0.03448 [μg/m^3]$^{-1}$. The 2010 mass-only emissions for diesel PM and benzene are 1,451 tonnes and 823 tonnes, respectively. After dividing by 1-in-a-million cancer risk levels, the cancer-risk weighted emissions for diesel PM and benzene are 1,030,349 tonnes and 23,880 tonnes, respectively.
8. **Quality assurance/quality control (QA/QC).** The summary tables and graphics were created and reviewed to identify and fix any errors in the raw data and refined TAP EI.

These steps were conducted using an MS Access database, Excel spreadsheets, customized Python scripts, and ArcGIS. The refined TAP EI is being provided to Metro Vancouver in an Access database that contains risk-weighted emissions for each of 32 municipalities in CLFV, 39 priority TAPs and 11 additional TAPs[2], and 106[3] source categories.

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[2] Metro Vancouver designated 12 additional TAPs while no chlorine dioxide emission is found in CLFV.

[3] Though 114 source categories were evaluated and processed, only 106 source categories have non-zero emissions for at least one pollutant in the 39 TAPs of interest.

Table D-1. Matrix of TAP Emissions estimating approach selection and data source for each sector/subsector.

Sector	Subsector	Existing TAP	EF Approach	TAP Augmentation	Speciation
Industrial	Bulk Shipping Terminals	a			f
	Chemical Manufacturing	a			f
	Electric Power Generation	a			f
	Heating / Cogeneration Utilities	a			f
	Metal Foundries and Metal Fabrication	a			f
	Non-metallic Mineral Processing Industries	a			f
	Paper and Allied Products	a			f
	Petroleum Products	a			f
	Primary Metal Industries	a			f
	Wood Products	a			f
	Miscellaneous Industrial Sources	a			f
Agricultural	Wind Erosion				f
	Fugitive Dust from Tilling				f
	Fertilizer and Pesticide Application				f
	Cattle				f
	Pigs				f
	Sheep				f
	Poultry				f
	Horses				f
	Miscellaneous Animals				f
	Agricultural Burning			d	f
	Greenhouses				f
Vegetative Burning	Prescribed Burning		c		f
	Residential Open Burning		c		f
	Land clearing		c		f

Sector	Subsector	Existing TAP	EF Approach	TAP Augmentation	Speciation
Fuel Distribution	Truck Loading				f
	Trucks in Transit				f
	Railcar Loading				h
	Barge and Ship Loading				f
	Marine Vessels in Transit				f
	External Floating Roof Tanks				f
	Internal Floating Roof Tanks				f
	Fixed Roof Tanks				f
	Service Stations & Bulk/Cardlock Refueling				f
	Aircraft and Marine Refueling				f
	Natural Gas Distribution				h
Natural Sources	Trees, Crops & Vegetation				g
	Marine Aerosol				f
	Wildlife				f
	Forest Fires				h
Chemical Products Use (Industrial, Commercial, and Consumer)	Consumer Products			d	f
	Dry Cleaning			d	f
	Glues, Adhesives and Sealants				f
	Metal Degreasing			d	f
	Architectural Surface Coating			d	f
	Automotive Refinishing			d	f
	Industrial Coatings			d	f
	Pesticides				f
	Printing Inks			d	f
	Other Industrial and Commercial				f
	Refrigerants				h
	Asphalt Paving				h

Sector	Subsector	Existing TAP	EF Approach	TAP Augmentation	Speciation
Heating	Residential - Natural Gas				f
	Residential - Fuel Oil				f
	Residential - Wood		c		f
	Light Industrial/Commercial/ Institutional - Natural Gas				f
	Light Industrial/Commercial/ Institutional - Fuel Oil				f
Miscellaneous sources	Meat Cooking			d	f
	Bakeries				f
	Tobacco Smoke				f
	Domestic Pets				h
	Vehicle Fires				f
	Structural Fires				f
	Crematoria		c		f
Fugitive Dust	Construction and Demolition (dust)				f
	Coal Dust from Rail Locomotives				f
	Quarries				h
	Road Dust				f
Waste	Anaerobic digestion				h
	Waste to Energy Facilities				f
	Landfills				f
	Wastewater treatment facilities		c		f
Light Duty Vehicles	Motorcycle	b			f
	Passenger Car	b			f
	Passenger Truck	b			f
	Light Commercial Truck	b			f

Sector	Subsector	Existing TAP	EF Approach	TAP Augmentation	Speciation
Heavy Duty Vehicles	Intercity Bus	b			f
	Transit Bus	b			f
	School Bus	b			f
	Refuse Truck	b			f
	Single Unit Short-Haul Truck	b			f
	Single Unit Long-Haul Truck	b			f
	Motor Home	b			f
	Combination Short-Haul Truck	b			f
	Combination Long-Haul Truck	b			f
Aircraft	Helicopter		c		f
	Turboprop Plane		c		f
	Commercial Piston Plane		c		f
	General Piston Plane		c		f
	Commercial Jet		c		f
	Military Jet		c		f
Rail	Line Haul Locomotive				f
	Switcher Locomotive				f
	Passenger Locomotive				f
Marine	Bulk (Oceangoing)			d	f
	Container (Oceangoing)			d	f
	Cruise Ship (Oceangoing)			d	f
	Ferry			d	f
	Fishing Vessels			d	f
	General Cargo (Oceangoing)			d	f
	Harbour Vessel			d	f
	Miscellaneous (Oceangoing)			d	f
	Motor Vehicle Carriers (Oceangoing)			d	f
	Tanker (Oceangoing)			d	f

Sector	Subsector	Existing TAP	EF Approach	TAP Augmentation	Speciation
Non-road Engines and Equipment	Agricultural Equipment			e	f
	Airport Equipment			e	f
	Commercial Equipment			e	f
	Construction and Mining Equipment			e	f
	Industrial Equipment			e	f
	Lawn and Garden Equipment (Commercial)			e	f
	Lawn and Garden Equipment (Residential)			e	f
	Pleasure Craft			e	f
	Railroad Equipment (non-locomotive)			e	f
	Recreational Equipment			e	f

a - TAP emissions were extracted and reconciled from Canada's three annual NPRI (2009, 2010 and 2011) provided by MV.
b - TAP emissions were extracted from MV's MOVES modeling results for year 2010.
c - Emissions factors were gathered from U.S. EPA's 2011 NEI Technical Support Document (TSD) and supporting datasets. Activity data for year 2010 were provided by MV.
d - TAP-to-CAP ratios were gathered from U.S. EPA's 2011 NEI TSD and supporting datasets. 2010 CAP emissions were provided by MV.
e - TAP-to-CAP ratios were gathered from U.S. EPA's National Mobile Inventory Model (NMIM). 2010 CAP emissions were projected by MV's from NMIM modeling results for year 2005.
f - Speciation profiles were assigned to each sector/subsector based on U.S. EPA's 2011 modeling platform, a previous VOC study for the CLFV (SNC Lavalin, 2012), and the similarity between the descriptions of speciation profiles in the SPECIATE database and the characteristics of source sector/subsector. Speciation profiles were extracted from the SPECIATE database and the previous VOC study (SNC Lavalin, 2012). 2010 CAP emissions were provided by MV including 2010 point emissions in facility permits for each point source, and nonpoint sources emissions by sector/subsector.
g - No speciation applied due to lack of appropriate speciation profiles and this is not an anthropogenic source that can be controlled.
h - No need for speciation due to zero or no VOC/PM emissions according to MV's CAP emissions inventory.

Table D-2. List of dioxins and furans included.

Pollutant	Abbreviation	Toxicity Weighting Factor (NATO I-TEF)
2,3,7,8-Tetrachlorodibenzodioxin	2378-TCDD	1
1,2,3,7,8-Pentachlorodibenzo-p-dioxin	12378-PeCDD	0.5
1,2,3,4,7,8-Hexachlorodibenzo-p-dioxin	123478-HxCDD	0.1
1,2,3,6,7,8-Hexachlorodibenzo-p-dioxin	123678-HxCDD	0.1
1,2,3,7,8,9-Hexachlorodibenzo-p-dioxin	123789-HxCDD	0.1
1,2,3,4,6,7,8-Heptachlorodibenzo-p-dioxin	1234678-HpCDD	0.1
Octachlorodibenzodioxin	OCDD	0.001
2,3,7,8-Tetrachlorodibenzodioxin	2378-TCDF	0.1
1,2,3,7,8-Pentachlorodibenzofuran	12378-PeCDF	0.05
2,3,4,7,8-Pentachlorodibenzofuran	23478-PeCDF	0.5
1,2,3,4,7,8-Hexachlorodibenzofuran	123478-HxCDF	0.1
1,2,3,6,7,8-Hexachlorodibenzofuran	123678-HxCDF	0.1
2,3,4,6,7,8-Hexachlorodibenzofuran	234678-HxCDF	0.1
1,2,3,7,8,9-Hexachlorodibenzofuran	123789-HxCDF	0.1
1,2,3,4,6,7,8-Heptachlorodibenzofuran	1234678-HpCDF	0.01
1,2,3,4,7,8,9-Heptachlorodibenzofuran	1234789-HpCDF	0.01
Octachlorodibenzofuran	OCDF	0.001

🟧 The emissions of dioxins and furans are reported as 2,3,7,8-TCDD equivalent.

Table D-3. List of polycyclic aromatic hydrocarbons (PAHs) included.

Pollutant	Abbreviation	Toxicity Weighting Factor [a]
Acenaphthene	ANA	0.001
Acenaphthylene	ANTL	0.001
Anthracene	ANTH	0.01
Benz(a)Anthracene	B[a]ANTH	0.1
Benzo(a)Pyrene	B[a]P	1
Benzo(b)Fluoranthene	B[b]FLAN	0.1
Benzo(e)Pyrene	B[e]P	0.01
Benzo(g,h,i)Perylene	B[g,h,i]PERY	0.01
Benzo(k)Fluoranthene	B[k]FLAN	0.1
Chrysene	CH	0.01
Dibenz(a,c) & (a,h)Anthracene	D[a,c]AN, D[a,h]AN	1

Pollutant	Abbreviation	Toxicity Weighting Factor [a]
Fluoranthene	FLAN	0.001
Fluorene	FL	0.001
Indeno(1,2,3-cd)Pyrene	I[1,2,3-cd]PY	0.1
Perylene	PERY	0.001
Phenanthrene	PH	0.001
Pyrene	PY	0.001
Notes		

[a] Malcolm and Dobson (1994).

🟧 The emissions of PAH are reported as Benzo(a)Pyrene equivalent.

References

Malcolm H.M. and Dobson S. (1994) *The calculation of and Environmental Assessment Level (EAL) for atmospheric PAHs using relative potencies*, Department of the Environment, London, Great Britain.

SNC Lavalin (2012) Lower Fraser Valley (LFV) volatile organic compounds (VOC) emissions inventory improvement project. Final report, April.

U.S. Environmental Protection Agency (2013) 2011 National Emissions Inventory, version 1: technical support document. Report, November.