Updated Micro-emissions Inventory for the Bulkley Valley – Lakes District of BC

PHASE 4

FINAL REPORT

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Executive Summary

Almost 20 years since the Bulkley Valley – Lakes Airshed Management Society released the first micro-emissions inventory (MEI) of particulate matter (PM), an update of any sort is warranted. This report summarises and details (in the Appendices) the development of such an updated inventory for 2015 and 2016, including the use of a Geographic Information System (GIS) for enhanced spatial resolution and accuracy.

The sources of PM inventoried include: large stationary point sources (N=11 facilities), mobile and linear emissions, residential and commercial heating, and resource management seasonal debris burning. Of these source classes it was found that unpaved road dust of the mobile and linear emissions class had the highest PM emissions of any single source type (3875.4 t/y of total particulate matter [TPM] in 2015; and 3886.0 t/y in 2016), followed by the open burning of debris for resource management (TPM = 3878.0 t/y- in 2015; 3886.0 t/y in 2016). Despite the prominence of road dust in this inventory (unpaved) and the previous inventory (paved) methods of estimating road dust are not considered to be accurate and may grossly overestimate PM.

Total emissions of TPM in the roughly 35,000 km^2 study area were estimated as 9804.0 t/y in 2015 and 10,129.6 in 2016 for those sources inventoried. Totals for both years are lower than in the previous inventory (Weinstein 2005), which is thought to be due to the removal of beehive burners, road dust, changes in industry, and variations in study area boundaries.

The monitoring of PM by the Ministry of Environment and Climate Change Strategy (ENV) revealed frequent exceedances of the 24-hour provincial air quality objectives (AQO) for PM10 at Smithers, Houston and Burns Lake. In addition the 98^{th} percentile 24-hour AQO for PM2.5 was exceeded in Houston both 2015 and 2016, but Houston also boosted the lowest number of PM10 exceedances in both years (90 and 41 in 2015 and 2016, respectively, compared with 367 and 212, respectively, in burns lake.

The combination of a living and constantly evolving MEI with PM monitoring data are believed to be the best tools available for management PM pollution in the BVLD. Storing and utilising data as maps in a GIS allows for spatial resolution to be refined, and potential hotspots or emissions overlap to be identified; however, the resource intensity of these programs should be acknowledged.

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1 Introduction

1.1 Background

All life on Earth depends on the composition of the atmosphere where it resides. However, the local atmosphere's composition is influenced by a myriad of chemical, biological and even physical parameters. One activity that has a substantial effect on atmospheric composition (aka 'air quality') is the release of toxic substances to air; releases that originate from a variety of anthropogenic (industrial, commercial, residential) activities or processes.

Detailed information regarding the releases of these 'atmospheric emissions' is required for activities such as: airshed planning, human health protection, risk assessment, conservation efforts, project permitting and/approval approval, community education, local interest and atmospheric modelling (e.g. climate, pollution and even weather, forecasts). This information is typically prepared in the form of an emissions inventory (EI) that can vary significantly in terms of scale, detail, source and/or contaminant inclusion, accuracy, and format, depending on the purpose for which the inventory is developed, and the information available at the time of EI preparation. Such EI are most often developed for regulatory purposes

The air pollutants most commonly inventoried, emitted and regulated are known collectively as 'criteria air contaminants' (CAC) and include carbon monoxide (CO), nitrogen dioxide (NO_x), sulphur dioxide (SO_2), (tropospheric) ozone (O_3), volatile organic compounds (VOC), and particulate matter (PM). These are considered 'criteria' pollutants due to their direct impacts on human and environmental health, that they have 'criteria' developed for them, and for their contributions to acid deposition ('acid rain') and photochemical 'smog' formation. Both VOC and PM represent a class or grouping of pollutants rather than a single chemically distinct substance (see: Glossary, Appendix A.1). Ammonia (NH_3)— a chemically distinct substance—is also a commonly inventoried and monitored pollutant, particularly in agricultural areas. Specific jurisdictions or organisations may decide to inventory and/or regulate additional pollutants that have become ubiquitous, but are not considered a CAC; [e.g. Canadian Association of Petroleum Producers (CAPP) Inventory of Greenhouse Gases and Criteria Air Contaminants (Clearstone Engineering 2005); Province of Alberta's extensive list of Ambient Air Quality Objectives (Alberta 2019)].

In the Bulkley Valley – Lakes District (BVLD), local and regional concern regarding PM emissions and ambient PM levels (exposure) led to the formation of a micro-emissions inventory (MEI) of PM prepared for the Bulkley Valley – Lakes Airshed Management Society (AMS) in 2005, based on year 2001 and 2002 data (Weinstein, 2005). The inventory was considered 'micro' for two main reasons: first, it represented emissions at the regional/community level—the BVLD (≈35,000 km^2); and second, it accounted for emissions of PM only.

Over fifteen years later, this MEI is out-of-date. The use of 2001 and 2002 emissions means that two decades have passed since the data presented in the inventory were timely. This document represents an updated MEI for the BVLD using data for the years 2015 and 2016. Although these years are also in the past, they represent years for which data were most readily available (see Section 2.1).

The purpose of the MEI on which this report is based is four-fold:

- help understand how PM emissions vary with space and time in the BVLD,

- identify primary sources / source-types that contribute to reduced regional air quality,
- inform decision-makers of options for improved airshed management and planning, and
- provide an answer to: "what is the 'best' emissions inventory that can be developed for the BVLD based on the available information and resources".

This document represents the final inventory report, including background, methods used, results and related data. The report aims to achieve all four individual purposes in a combined and holistic manner.

1.2 Regional Geography and Meteorology

Geographic elements such as topography local and meteorology both strongly influence the way pollutants disperse and eventually settle in any given area. Spatial aspects also influence biological outcomes and/or impacts—i.e. whether or not a biological receptor will be affected by a specific source, combination of sources, etc. The MEI includes sources located within what is roughly the BVLD, an area of 41,497 km^2 as shown in Figure 1.

The BVLD represents an area of complex terrain, with elevations ranging from 600 – 1650 masl, and is marked by the Bulkley River valley and its tributaries, as well as numerous other water features. Mountains and valleys have specific meteorology that affects pollutant 'behaviour', particularly how pollutants are distributed and transported, as well as how they may interact with one another and with biological receptors such as humans. For instance, PM may accumulate in low-lying areas such as valleys; particularly during the winter, nighttime, or any time when a persistent stable boundary layer supresses vertical mixing. The numerous valleys of the BVLD, therefore make it prone to the accumulation of atmospheric pollutants. Examples of PM behaviour in the study area are photographed in the morning and evening in Figures 2a and 2b, respectively.

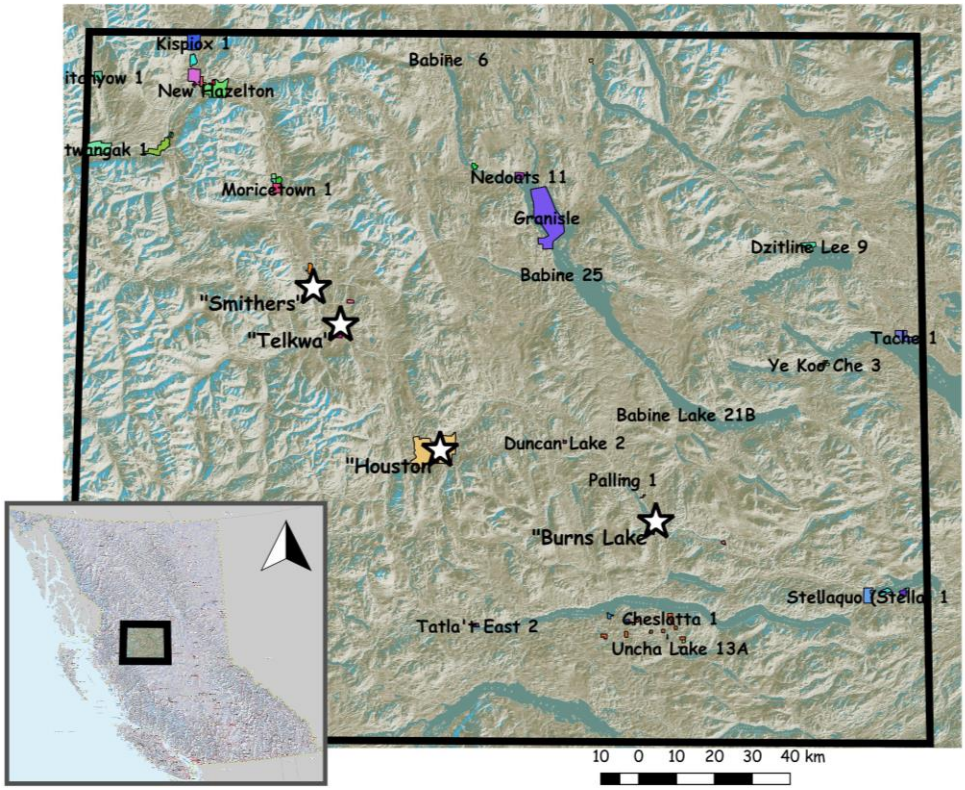
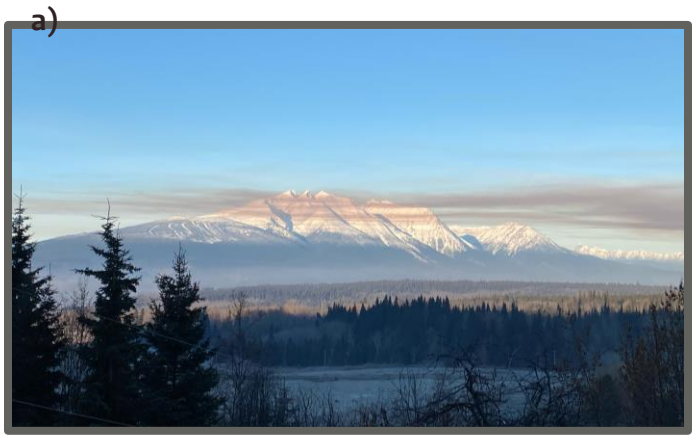
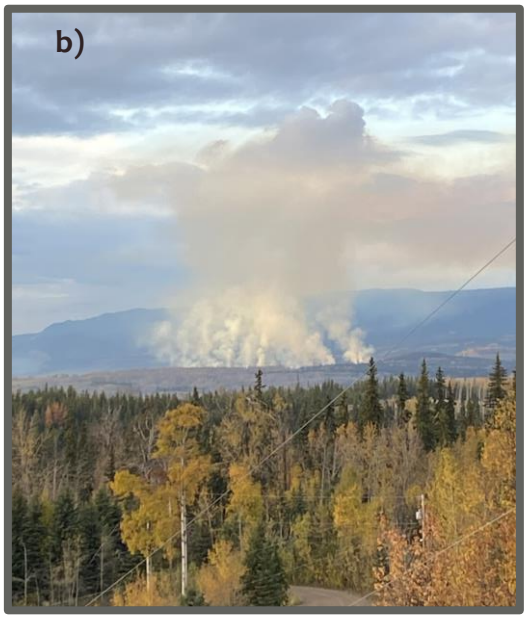


Figure 1. Map of study area in relation to province of BC (inset) showing locations of ENV's meteorological/PM monitoring stations (white stars) and some of the local communities (coloured polygons). Other maps presented in this report use the same scale.

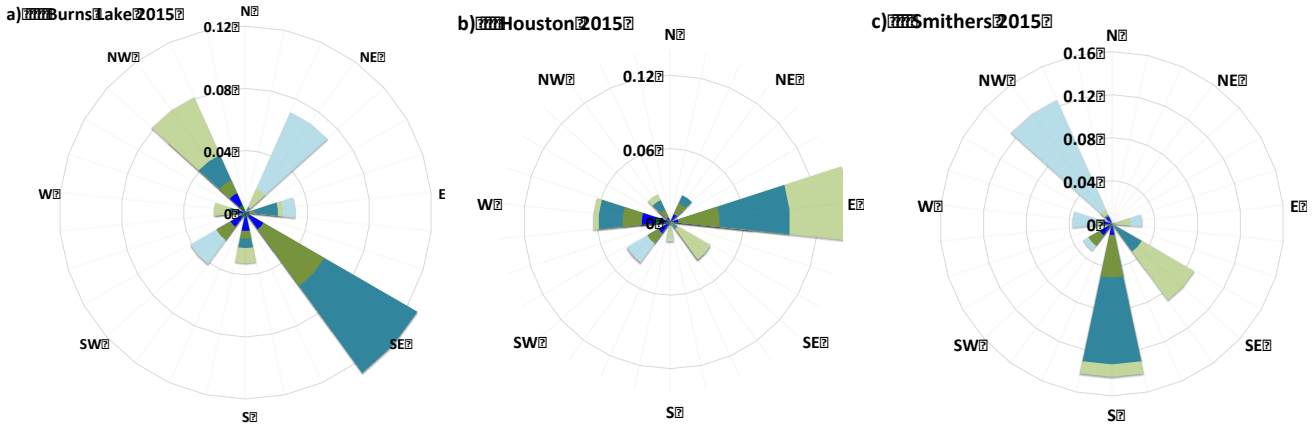
The government of BC's Ministry of Environment and Climate Change Strategy (ENV) has four meteorological monitoring stations in the BVLD that also measure PM (Figure 1). In 2015 and 2016 wind speed and direction were measured in each Smithers, Houston, and Burns Lake, while precipitation data were collected in Telkwa. While these sites also monitored ambient $PM_{2.5}$ and PM_{10}, the Telkwa site measured only $PM_{2.5}$ and stopped doing so in May 2015.

Figure 2. Photos of the study area taken from Hillcrest Road just off Babine Lake Road, at: a) around 9:00 am in the winter showing accumulated PM; b) autumn evening showing plume rise from a resource management open burn. Images courtesy of Servaas Mes, all rights reserved.





All monitoring sites are located within the (Bulkley River) valley where residents are most likely to be exposed to PM. Winds in the area tend to be calm (Figure 3a-f) and their prevalence varies greatly from site to site; thus illustrating the influences of local topography local air flow, and therefore, pollutant transport.



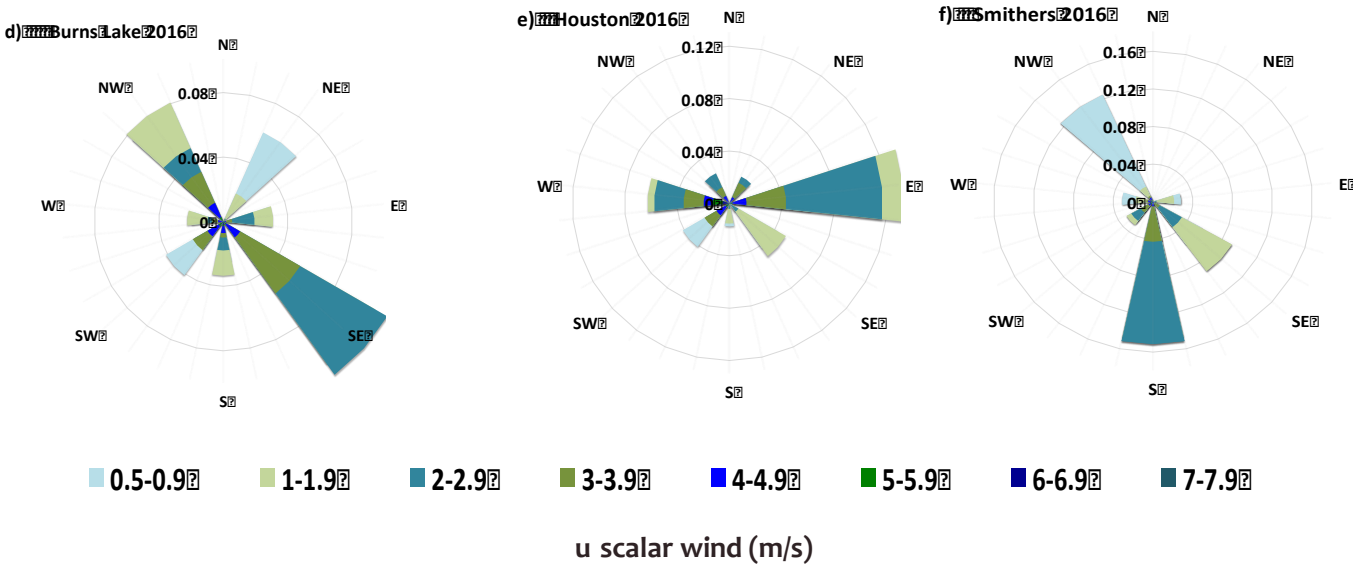


Figure 3. Wind roses for the year 2015 (a-c) and 2016 (d-f), developed using MOE wind speed and direction data from sites in Burns Lake, Houston, and Smithers—that also monitor PM; Telkwa did not have wind measurements for either year. Data were processed and displayed using Microsoft Excel for Mac™ as ratios of wind events rather than a count or percentage. Note that darker colours, representing stronger winds are less frequent and therefore confined to wind rose centers.

1.3 Particulate Matter

Particulate matter (PM) is a non-gaseous class of atmospheric pollutants that exist in either solid or liquid (aerosolised droplets or condensed vapour) forms. Unlike gaseous pollutants, such as the other CAC, particles are not measured by concentration (%) or volume ratio of air (e.g. parts per billion [ppb]). Rather, PM is measured in micrograms per cubic metre ($\mu g/m^3$). Gases can also be measured in $\mu g/m^3$, but the unit is sensitive to changes in temperature and pressure.

Conventionally PM is separated into particles of less than or equal to 10 microns (µm) in diameter–PM_{10}, pand articles of equal to or less than 2.5 microns (µm) in diameter–$PM_{2.5}$. As such $PM_{2.5}$ is also included in PM_{10}, but there may be coarser particles (> 10 µm) that are still able to stay suspended in the atmosphere. Total Particulate Matter (TPM) is the term used to include all atmospheric particles classified as PM10, PM2.5 or greater. Very fine particles or those of less than 1 µm in diameter are becoming more recognised for their importance on human and environmental health; similarly denoted PM_1.

When PM is emitted directly from a source, it is referred to as 'primary' PM. 'Secondary' forms of PM are produced by chemical reactions between primary pollutants and other substances. Emissions inventories (by definition) only look at primary pollutants emitted directly from sources. That being said, even primary PM is chemically and physically complex, the specificities of which can have a significant influence on the biological outcomes of PM exposure (Bølling et al. 2009).

For instance, particles may be composed of, or contain, heavy metals, which according to the Canada's National Pollutant Release Inventory (NPRI), are released by sources within the airshed

(ECCC 2018a). The importance of PM in the BVLD is due to the relatively high number of PM sources in the region, including:

1) Forest sector sources
   a. On-road and off-road dust
   b. Vehicular diesel exhaust
   c. Mill/plant effluent
   d. Smoke from open burning / solid waste management

2) Metal ore extraction/mining
   a. Facility, road and off-road dust
   b. Vehicular diesel exhaust
   c. Process Fuel (crushers, grinders, etc.)
   d. Mineral and ore dust
   e. Condensable process vapours
   f. Smoke from open burning / solid waste management
3) Natural Resource Open Burning (Land Clearing)
   a. Smoke
   b. Ash
   c. Polycyclic aromatic hydrocarbons (PAH)
4) Transportation
   a. Vehicular fuel exhaust (gasoline, diesel)
   b. Unpaved road dust (weathered minerals, wetting agents)
   c. Paved road dust (asphalt, salt/de-icing agents)
   d. Brake pads and tires
   e. Motor oil
   f. Train/rail exhaust
   g. Water vessel fuel (diesel) exhaust
   h. Recreational vehicles and pleasure craft (off-road dust, exhaust)
   i. Aeroplanes and helicopters
5) Residential and commercial heating
   a. Oil (diesel) and gas (propane or methane)[1] (forced air) furnaces or boilers
   b. Wood and pellet furnaces/boilers
   c. Wood and pellet stoves/fireplaces
6) Residential burning[2]
   a. Combustion of (usually moist) yard and garden waste
   b. Combustion of household refuse
7) Agriculture
   a. Exhaust from combustion engines (diesel and gas)
   b. Soil dust from tilling and erosion
   c. Solid and liquid particles or aerosols from crop dusting or spraying of pesticides

[1] PM emissions from natural gases such methane and propane are extremely low; however, if the furnace isn't properly maintained, serviced or functioning, incomplete combustion can cause the formation of soot and ultrafine particles in the furnace. Further, volatized compounds such as complex organics that can be formed from incomplete combustion may adhere to the surface of soot and other PM from the furnace or elsewhere.

[2] Also includes smoke and other emissions from open burning of waste branches, bark, stumpage, etc in the forest sector, already listed as 1 f)

   d. Dust or aerosols from broadcast fertilizer application
   e. Smoke from open burning of crop waste
8) Natural \sources
   a. Wildfires
   b. Vegetation (pollen, condensable VOC)
   c. Land-slides
   d. Commercial and residential food preparation
   e. Use of aerosol sprays
   f. Smoke from the use of tobacco, smudge, incense, etc.
   g. Outdoor camp-/cook-fires
9) Transboundary sources[3]
   a. Sea salt
   b. Long-range transport of foreign dust (e.g. from Asia)
   c. Volcanism

Each of these PM sources has a unique particle size ratio, in addition to a unique chemical signature, and temporal emissions' profile). Although Appendix A.3 includes some sources not inventoried as part of the MEI, it summarises some of the physical, chemical and temporal characteristics of different PM source types. For instance, natural sources are not traditionally included in emissions' inventories, but do usually play a role in the atmospheric modelling that ensues—particularly chemical transport models.

1.4 Health and Environmental Impacts of PM

In terms of ambient exposure, PM is the most likely of all inventoried CAC[4] to exist at levels known to directly impact human health. Both primary and secondary PM exposures are associated with negative human health outcomes. PM has been linked to cause asthma, chronic obstructive pulmonary disease (COPD), decreased lung function and additional pulmonary diseases / respiratory injuries (Pope and Kappos et al, 2004; Hrebenyk et al, 2005; Dockery, 2006). Some less obvious PM-triggered health conditions include arterial hypertension, coronary artery disease, obesity, adverse pregnancy outcomes, cancers, and diabetes (e.g. Kappos et al, 2004; Lelieveld and Pöschl, 2017). Airborne fine particles can make their way to, and impact, every system in the human body. Fine particles ≤2.5 µm in diameter ($PM_{2.5}$) are considered particularly damaging because they can travel more deeply into lung tissue, from where they can then make their way through capillaries and blood vessels to reach other organs (Kampa and Castanas 2008). Particles of ≤1.0 µm (referred to as 'sub-micron', 'nano', or 'ultra-fine') are especially biologically mobile/damaging—the smaller the particle, the larger its capacity to impact human health.

Damage to these diverse aspects of the human system creates comorbidities that influence our responses to other stressors. For example, exposure to fine particles can increase the risk and

[3] Transboundary sources do not exist within the BVLD, but are sources that may affect atmospheric PM levels within the BVLD via long-range transport. Although there are active volcanoes within the study area, none have erupted in over 200 years.

[4] Inventoried CAC are primary pollutants emitted directly from a source and included in emissions inventories, as opposed to measured CAC which also include secondary pollutants that are reaction products between CAC and/or additional atmospheric components (see CAC in Glossary)

severity of other ailments, particularly those with respiratory and/or cardiovascular components. For instance an increase in mortality rates from COVID-19 (severe acute respiratory syndrome coronavirus (SARS-CoV-2)) has been attributed to air pollution—or more specifically $PM_{2.5}$ (Pozzer et al. 2020); as was true of its predecessor SARS (SARS-CoV-1) (Cui et al. 2003). Between 2008 and 2015, Weichenthal et al. (2017) found a relationship between the rates of myocardial infraction (heart attack) and measured $PM_{2.5}$ in Kamloops, Courtenay and Prince George BC—a relationship that strengthened during winter months when temperatures were <6.44°C, and thought the be related to residential biomass burning (wood smoke exposure).

Ambient Air Quality Standards (AAQS) and Air Quality Objectives (AQO) form the basis of air quality management in Canada and supersede all previous Canada-wide Standards. They set a maximum threshold for the concentration of a specific pollutant in the ambient atmosphere. In 2012, the Canadian Council of Ministers of the Environment (CCME) derived the two 24-hour AAQS for $PM_{2.5}$ as: 28 µg/m³ (to obtain by 2015) and 27 µg/m³ (to obtain by 2020). Both values were developed for comparison with 3-year averages of the annual 98th percentile of daily (24-hour average) concentrations (CCME 2012).

In 2009 BC adopted a 24-hour Provincial Ambient Air Quality Criteria (AAQC) for $PM_{2.5}$ of 25 µg/m³ (in addition to the existing 24-hour PM_{10} AQO of 50 µg/m³) representing the annual 98thpercentile of 24-hour concentrations. In contrast to CCME's standard, BC's threshold percentile is based on a single year of data (BC ENV 2020), rather than three. The World Health Organization (WHO) has a 24-hour $PM_{2.5}$ standard of 25 µg/m³; however, it uses the 99th percentile of measurements on which to base compliance (WHO 2005), rather than the 98th. While a 24-hour threshold is useful in protecting human health from daily episodes of poor air quality, PM's potential health effects can be chronic and occur over a much longer time period. As such, annual standards for PM have also been developed. In BC these are an annual: 'objective' of 8 µg/m³, and 'planning goal' of 6 µg/m³ (BC ENV 2020) both representing the annual average of hourly averaged PM measurements. There are no hourly ambient standards for PM, because ill effects generally occur with repeated and/or chronic exposure (with the exception of acute respiratory failure from smoke inhalation, which usually occurs in indoor and/or poorly ventilated environments.

Table 1. The 24-hour 98th percentile ("%") and annual average PM2.5, the annual average of the 24-hour average PM10, and the number of times in a year the 24-hour average exceeded BC's AQO for the 2015 and 2016 inventory years. The 24-hour averages are rolling averages so as to eliminate diurnal bias. Data are from the ENV monitoring sites shown in Figure 1 and were provided as hourly averages. Exceedances of AQO are underlined and all units are µg/m³ except for the last two columns of count data.

Site	$PM_{2.5}$				PM_{10}			
	24-hour 98th %		Annual		Avg 24-hr Avg.		# of 24 hr > 50	
	2015	2016	2015	2016	2015	2016	2015	2016
Burns Lake	18.7	18.3	7.1	6.8	15.7	14.8	367	212
Houston	28.3	26.4	9.7	8.8	14.6	13.8	90	41
Smithers	25.0	22.4	8.3	7.3	14.6	13.1	172	124
Telkwa	31.2	–	9.1	–	–	–	–	–
All Sites	27.4	23.1	8.5	7.6	15.0	13.9	629	377

The 2015 and 2016 24-hour 98th percentile $PM_{2.5}$ and 24-hour average PM_{10} concentrations from the ENV sites in the BVLD are shown in Table 1 for comparison withe BC's AQO for PM. Exceedances of the 24-hour AQO for both $PM_{2.5}$ and PM_{10} were more frequent in 2015. Houston and Telkwa both had 98th percentiles of 24-hour $PM_{2.5}$ that exceeded the provincial AQO of 25 µg/m^3, while Smithers' was 2015 98th percentile was precisely 25 µg/m^3, the the AQO (Table 1). While Burns Lake did not exceed $PM_{2.5}$ standards in either 2015 or 2016, it had the highest measured levels of PM_{10} and the most frequent exceedances of the 50 µg/m^3 24-h AQO.

Figure 4 shows a bar plot of annual average $PM_{2.5}$ emissions measured at the ENV sites over the 2015 and 2016 inventory years, compared with the provincial annual AQO and federal annual CAAQS for $PM_{2.5}$. All sites exceeded BC's planning goal in both 2015 and 2016, and Houston exceeded federal and provincial criteria in both 2015, and 2016.

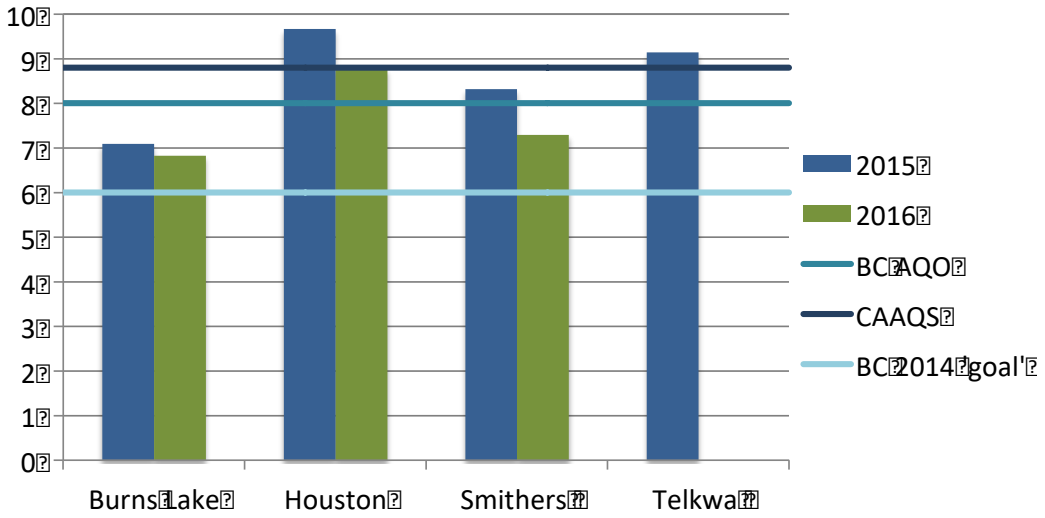


Figure 4. Chart of annual average concentrations (bars) of $PM_{2.5}$ (g/m^3) compared with relevant ambient air quality criteria (lines).

The seasonal trends in PM concentration are shown in Figure 5 by plotting measured PM_{10} and $PM_{2.5}$ as monthly averages for both 2015 and 2016. Earlier fall peak concentrations in 2016 indicate a differences in natural resource debris burning—a regular seasonal (fall) activity that produces large quantities of PM emissions, and is included in the MEI. Also apparent from Figure 5 is a seasonal pattern in regional PM that is more pronounced for PM_{10} than $PM_{2.5}$ (c and d, versus a and b, respectively). The variation in PM between sites is also reduced in $PM_{2.5}$. Although these more homogenized temporal and spatial patterns in $PM_{2.5}$ may be due to the nature and emissions of local sources, it is more likely that the smaller particles in $PM_{2.5}$ reduce their aerodynamic resistance and allow them to stay buoyant for longer periods of time and travel greater distances—reducing variation in time and space.

Additionally, Burns Lake seems to receive more PM_{10} than the other locales, possibly from the Edako mine just to the south (see section 3). We also see a more gradual transition to summertime low PM in 2015, in particular the months of May and June have PM_{10} concentrations up in the teens (µg/m^3), while the following year they were near zero at this same time. This likely implies one of two things:

there was an earlier start to the 2015 wildfire season, or a delayed start to the summer such that residents needed to use more heat than usual. Well, it turns out to be a little of both, but in reverse— 2015 was the norm, not the exception. According to some of the local news outlets, 2016 had an early, but wet, start to the summer (which would dampen the use of heating, and forest fire starts), and by mid-August had only seen one-third of the wildfires as 2015 by same time in the season[5].

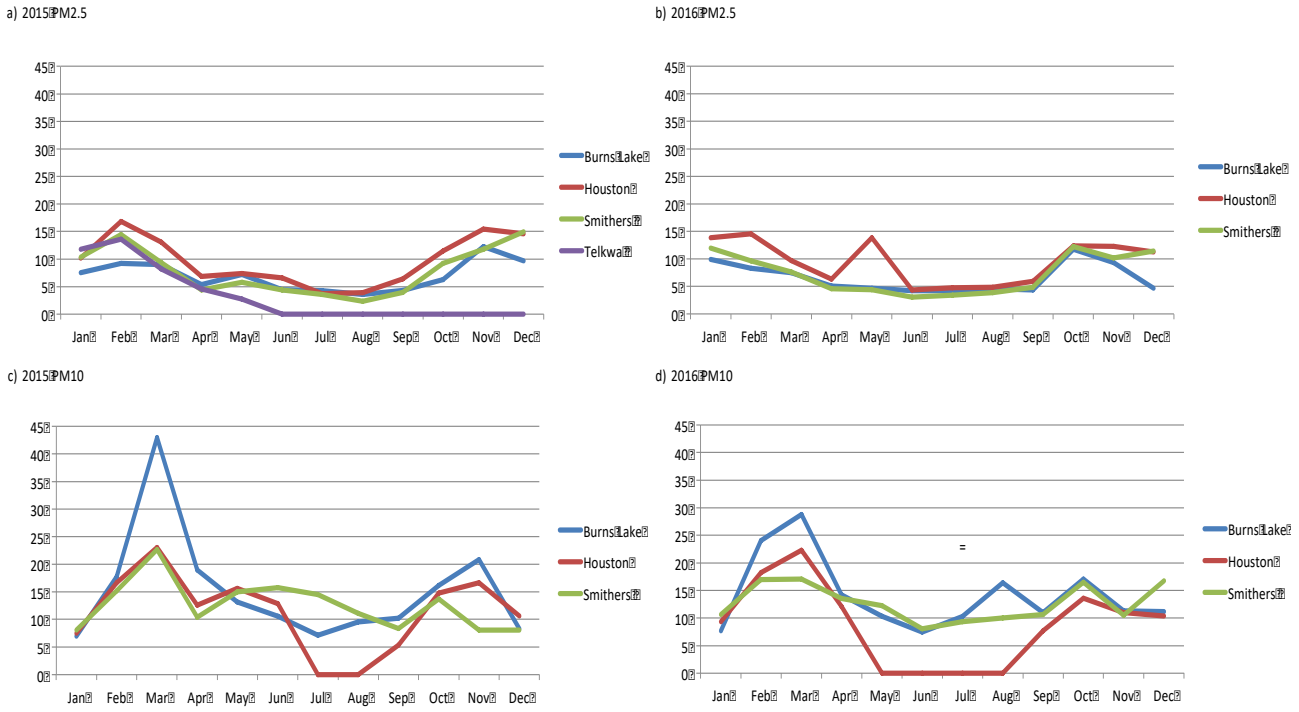


Figure 6. Line charts of monthly average measurements of PM taken at ENV operated monitoring stations within the AMPA: a) and b) show $PM_{2.5}$ for 2015 and 2016, respectively; and similarly c) and d) show monthly PM_{10} for 2015 and 2016, respectively.

[5] Smithers Interior News, August 16, 2016. "Slow B.C. forest fire season may heat up".

2 Methods

2.1 Scale Determination and Source Inclusion

One of the many challenges in EI development at any scale is limiting what could be limitless. An inventory could endlessly grow and evolve over time; to spatially represent the entire globe, at a finer and finer resolution, with the introduction of evermore sources and pollutants, and the perpetual introduction or removal of sources. Although this may sound 'ideal' from a management perspective, it isn't practical, and a position must be found between the imaginable and the achievable.

For the BVLD's MEI, inventory scale and detail were determined by two main factors:

- the purpose/end-use of the information
- the resources available (time, funding, computing, data)

The updated MEI attempts to capture as many local PM emission sources as possible, including area sources such as residential heating and road dust; and large stationary point sources[6] such as those that report to Canada's National Pollutant Release Inventory (NPRI). The largest caveats in the development of this inventory were:

- a lack of data, which limited the sources that could be included, and the data/accuracy of derived emissions' estimates
- large datasets and resource intensive processes involved in use of the Geographic Information System (GIS) [7]

During the scoping phase (Phase 1) of this MEI project, seven different scoping measures (or elements) were chosen: spatial, temporal, chemical, causal, utilizational, financial and practical. Spatially, the study area chosen is similar to that used by Weinstein (2005) and roughly represents the BVLD and/or the AMS' airshed management plan area (AMS 2012). The actual boundaries of study (black outline in Figure 1) were chosen somewhat arbitrarily and adjusted a number of times.

There were some significant PM sources at or beyond the boundary. Initially, important or large out-of-bounds sources, particularly those upwind of the BVLD, were going to be included. However, then yet another boundary would need to be determined—a buffer so to speak—that would contain additional sources and essentially expand the study area. But the reason we set boundaries is to contain what would otherwise be limitless. A decision was therefore made to include those sources that fit within the study area's bounding box when their latitude or longitude was rounded to the nearest 10th of a degree The limits of the bounding box being 53.7 – 55.4°N and -124.7 – -128.1°W. This placed some sources right atop the boundary, and others outside of the study area (for

[6] In this report the terms "point" and "stationary" are used to discern from "area" and "mobile" sources, respectively. See glossary for more information.

[7] There are an extraordinary number of files and datasets associated with the MEI; requiring roughly 30 GB (Gigabytes) of storage.Only the essential and final versions of data and maps are provided digitally for Appendices B and C.

instance if they had a latitude of 55.447°N). Had the bounding box been decided, or even rounded, slightly differently, the number of sources and emission totals would vary significantly from this MEI.

Besides the size of the study area, which was set to be both practical and relevant (from a management perspective), the spatial element of the MEI us essential in terms of source location, the respective contribution to atmospheric PM load at any given location, source receptor interactions, transport models, etc. For instance point sources, with distinct locations and spatial coordinates (latitude and longitude) allow for the direct measurement of distance between them and sensitive receptors—such as vulnerable members of the population. Whereas an area source, such as residential heating, may only be estimated as an average for the area in question, due to a lack of information regarding who burns what, when and where (information that would also lead to a much larger dataset). However, one can reasonably assume that emissions from residential heating occur more in areas of human residence, during the colder months, when people are home.

The temporal aspects of source inclusion and MEI resolution are equally important to the spatial aspects. The more spatial and temporal information, or the higher the spatial and temporal resolution of the MEI, the more usability an inventory has—the downside being larger and less manageable datasets. You may also introduce more and compounded error with each additional variable. For instance if emissions were inventoried in grams per second per square meter ($g/s/m^2$) rather than in tonnes per year for the study area (km^2) ($t/y/35{,}000\ km^2$), then each g, s and m^2 would offer the opportunity for error. Here emissions are presented and summarised in $t/y/35{,}000\ km^2$ because most emissions are reported as annual totals, and it makes it easier to compare and contrast source magnitudes when working with the same spatial and temporal scales (units). However, operational schedules and variances of them are provided in Appendix A.3 by source.

There were some changes made in source inclusion between this updated MEI and the previous MEI (Weinstein 2005). These were:

1) residential Backyard burning was not included;
2) road dust emissions were calculated for unpaved, rather than paved, roads;
3) residential heating with gas and oil were added; and
4) commercial heating was included as a potential source of PM

Backyard burning was omitted from this MEI for a few reasons:

1. No improved methods over those of Weinstein (2005) were identified, and the method's results are easily updated by revising population numbers. The emissions estimates from backyard burning can also be spatially refined by estimating emissions at the municipal, census, or community level. Alternatively, a more detailed population dataset (such as that provided digitally in Appendix C) can be used to allocate emission factors (EF) spatially by household should this level of spatial resolution be warranted.

2. The validity of United States – Environmental Protection Agency (US-EPA) defaults and EF for backyard burning is questionable. First, the routine open-burning of 28% of waste seems extraordinarily high, particularly for household waste, even in rural areas. Assuming there is residential garbage collection, it seems that the open burning of household wastes would be more of an exception than a rule.

3. The EF used for backyard burning appear unfitting. For instance the EF (in kg/t) for household waste burning, are lower than the EF for leaves, brush or grass. It isn't clear what the assumed moisture content of the yard waste would be, despite that moisture inhibits the complete combustion of any fuel, and incomplete combustion favours the formation of suspended fine particles. Although the two waste streams are comparable in terms of carbon and hydrogen content, household waste contains higher amounts of nitrogen and sulphur, which easily oxidise more than once to produce nitrate and sulphate aerosols. A more heterogeneous waste mixture (such as household waste) would also promote incomplete combustion (different substances require different combustion temperatures) and the production of a more chemically diverse set of PM including: halides (in particular fluorides and chlorides), metals, and ultrafine hydrocarbons and soot (elemental carbon). Shemwell and Levendis (2000) found laboratory-produced combustion-derived emission rates for polymers (aka plastics) to be higher than the EPA's EF used for PM_2 (particles < 2 μm in diameter) alone; and household waste is known to be high in such polymers.

4. Lastly the contribution of PM emissions from backyard burning was relatively small (0.1% of TPM) in Weinstein (2005).

Reasons for the other MEI inclusions/exclusions are discussed more in the following sections for specific source types. A detailed list of assumptions made in MEI development is provided in Appendix A.2. The methods used to estimate and quantify PM emissions within the study area are described in the following sections for each PM source type with detailed data and methods provided for each source/section in Appendix B (digital).

2.2 Large Stationary ('Point') Source Emissions

Large stationary emission sources are associated with spatially discrete commercial/industrial facilities that are required to report to Canada's NPRI. Estimated emissions from large stationary point sources were also verified and compared against emissions permits received from BC's Ministry of Environment and Climate Change. Only one PM source was found to overlap between the permits and the NPRI, Pinnacle Renewable Energy Inc.'s wood pellet manufacturing plant just east of Burns Lake. The permit entry was kept, rather than the NPRI entry because emissions' estimates in the permit represent the "maximum" quantity of a pollutant that a source is permitted to emit. The permitted maximum is often equal to the equipment or process maximum—the highest level of a pollutant (in this case PM) that could possibly be emitted by the given activity—but does not represent the emissions that could occur during facility upset or failure. Although these maxima are rarely if ever reached, and therefore likely result in emissions' over-estimates, their use is in line with the commonly used "worst case scenario" of EI development, and within the general precautionary principle of environmental science—if you don't know for sure, it is best to proceed with caution.

The NPRI records for these sources are representative of an entire facility, so although presented as 'point sources' (i.e. a discrete location with a defined latitude and longitude) some of these sources may represent emissions from a variety of processes over a larger, facility-sized, area. For example, PM may be emitted on-site from a fuel-fired boiler stack, vehicles, routine on-site open burning, and

on-site or off-road dust. The author does not think that emissions for these sources could be better estimated than those of the source owners themselves (i.e. the data provided would be used with the same emission factors and yield the same results). That being said, the NPRI has its faults. These faults may be in terms of the way emissions are estimated; or 'who'/ 'what' is included in the inventory (i.e. required to report).

There are substantial NPRI reporting exemptions; be they based on employee hours, energy usage, pollutant thresholds, production thresholds, or another activity limit—as published (now) every three years in Canada's *Gazette*. These threshold-based exemptions can (and do) ultimately mean that many smaller industrial and commercial sources go unreported. However, being small does not necessarily make an emission source insignificant, particularly when all of the sources too small to report are considered on together cumulatively, as found in previous assessments (Krzyzanowski 2009). The reader is encouraged to familiarise themselves with NPRI reporting requirements should they be interested in knowing more[8]. Data used for NPRI sources in the BVLD are available in Appendix B.1.

Sources of PM emissions permitted by ENV under requirements of the Environmental *Management Act* (*EMA*) and not included elsewhere[9], were added using maximum discharge amounts, maximum discharge rates and permitted operating schedule as defined in the respective permit (see: Appendix B.2). The 'all_ams_discharges_v10_-_nov3-_2020_1.xls', or 'alldischarges', file (Government of BC, 2020a) was used to extract permitted sources operating within the BVLD in 2015 and 2016. Because this file contained only maximum discharge rates in units of mg/m^3 or kg/day, each source's permit was accessed using an Authorization Number search in order to obtain maximum discharge rates and operating schedules (Government of BC, 2020b). This allowed the emissions of TPM (E_{TPM}) for each individual source to be calculated in tonnes/year (t/y) as follows:

$$E_{TPM} = Q_{MAX} \times PM_{MAX} \times t \times \frac{1}{m} \qquad 2.2\text{-}1$$

where, Q_{MAX} is the maximum allowable rate of discharge for an individual source (m^3/min, m^3/y, kg/d, m^3/month, t/y, etc.);
PM_{MAX} is the maximum mass of PM allowed in a specified volume or mass of discharge[10] (mg/m^3, kg/t, etc.) [field contained in the alldischarges file];
t is a function of annual operating time that depends on the units of Q_{MAX};
m is factor converting mass into tonnes (t);
x (below) is any positive real number

For example: if Q_{MAX} = *x* m^3/min, then *t* = 120,000 min/year (of operating time) assuming the source in question only operates 40 hours a week, 50 weeks a year. If Q_{MAX} = *x* kg/d then *m* = 1000 kg/t; and if Q_{MAX} = *x* m^3/y then *m* = *x* m^3/t — or the density of the material being disposed of. Q_{MAX} is assumed to be a realistic interpretation of what 'could' be emitted.

[8] A synopsis of NPRI reporting requirements can be found at: https://www.canada.ca/en/environment-climate-change/services/national-pollutant-release-inventory/report/requirements-fact-sheet.html
[9] Or 'removed from elsewhere', such as the Pinnacle Renewable Energy Inc. plant discussed earlier in this section
[10] Also known as "Discharge Max" in the BC_alldischarges file [see Appendix B.2]; and as "Characteristics of the Discharge" in Permits issued under the *Environmental Management Act*, and in Weinstein [2005])

In the case of the routine open burning of wood included in the provincial facility permits, without any Burn Registration Number (BRN) overlap (N = 10), values of wood density were the same as those used in the estimates of seasonal resource sector debris burning (see section 2.5).

For open burning by *EMA* permit (rather than BRN), the emissions equation became:

$$E_{TPM} = \frac{\rho_{wood} \times Q_{MAX} \times PM_{MAX}}{1000} \qquad 2.2\text{-}2$$

where, ρ_{wood} is average estimate wood density for the area in t/m^3;
Q_{MAX} is in m^3/y (wood)
PM_{MAX} is in kg/t (weight of PM per tonne of wood); and
$\frac{1}{1000}$ converts kg to t

This is different from the methods used in subsection 2.5 because the information provided for the two types of sources differs, and Equation 2.2-2 follows the format of Equation 2.2-1 keeping estimation methods consistent for large point source facilities and their multiple sources.

Because permits are active indefinitely, it was assumed that 2015 and 2016 emissions from permitted sources would be the same, with the exception of 2016 being a leap year and therefore representing 366 days. Also, the Huckleberry Mine closure on September 1, 2016 affected that source, but there were otherwise no permit amendments or closures of facilities in the study area from 2015-2016. Basic source information garnered from the alldischarges file and individual source permits, as well as calculated annual emissions, are provided in Appendix B2-2 and B2-3. The value of TPM in the permits was divided into $PM_{2.5}$ and PM_{10} size fractions using the PM size distribution ratios given in Table 2 below.

Table 2. Normalised size distribution ratios for TPM used to estimate the annual emissions of PM_{10} and $PM_{2.5}$ for large stationary sources holding an emissions permit under the *EMA*. All values represent a mass ratio compared with TPM (which is unity as it includes both PM_{10} and $PM_{2.5}$). Because $PM_{2.5}$ is included as PM_{10} the, ratios of the two do not add up to one.

PM RATIOS	TPM	PM_{10}	PM_{25}	Source
Drying Kilns	1.0	0.5800	0.1900	Weinstein (2005)
Hog Fuel Fired Boilers	1.0	0.9000	0.7597	Weinstein (2005)
Cyclones (most)	1.0	0.4000	0.2000	Weinstein (2005)
Molybdenum dryers, crushers, etc.	1.0	0.5100	0.1500	Weinstein (2005)
Vent Fans	1.0	0.5800	0.1900	Weinstein (2005)
Gold Crushers	1.0	0.4035	0.1053	AMEC (2013)
Open Burning	1.0	0.7078	0.6164	CONSUME (US Forest service, 2015)

2.3 Mobile and Linear Source Emissions

Data for mobile emissions estimates from road transport were limited. For instance, the Europe-based COPERT Street-level model v. 5.1.1 (Emisia 2018) could not be used with information from the Government of BC's Traffic Data Program (Government of BC 2018a) due to insufficient data. With the exception of a permanent count site at Pipers Glen on route 16 west of the Nachako River bridge and east of Fort Fraser (P-45-1EW) traffic monitoring sites in the BVLD have 'short count' data (48 hours to one week) reported every three years—the most recent available being from July 2014 for all other sites when this project was initiated. Additionally, traffic volume counts exist only for paved primary roads, which although they likely get the most traffic, they only represent 3.2% of the study area's total road length. However more recently, some BVLD sites and data (such as speed) have been added to the Ministry of Transportation's traffic data site providing a list of the traffic count sites throughout the province and the dates/roads for which these data are available, with some going back to August 2017[11].

Spatial road attribute data were retrieved from BC's Digital Road Atlas (GeoBC 2018) as shapefiles and loaded into a Quantum Geographic Information System (QGIS). BC's Digital road atlas classifies road surfaces as described in Table 3. Because gravel and dirt roads make up the majority (96.8%) of the area's road length, they are considered to be an important source of regional PM emissions. The three road surface types assumed to contribute to road dust emissions (i.e. dust from the road material itself) were loose, recreational, rough and unknown—collectively termed 'unpaved roads'. Unpaved roads are assumed to produce the most dust during the warmer fair weather (snowless) months, approximately July 1 – September 30 (Government of BC, 2018b).

In this respect 'road dust' is considered to be airborne particles that were once part of the road itself—as gravel, sand, or 'dirt'. This would be mostly silicate minerals or 'silicates', a group of minerals composed of mostly silica and oxygen, including clay, quartz, mica, garnet and tourmaline. This dust would become airborne from weathering and subsequent mechanical disturbance, such as a logging truck or passenger vehicle. Considering the large proportion of unpaved roads in the study area, unpaved roads were considered a potentially important source of PM.

Conversely, Weinstein (2005) estimated PM emissions from paved roads in the region. These particles are not composed of road material itself, but rather remnants of 'winter traction promotion' (salt, gravel, de-icing agents) that get kicked up in the spring (mostly March) after snowmelt and drying, that leads to what is termed the 'road dust season'. Although these remnants of winter traction materials represent a significant source of PM, they only occur at a discrete time of year and across a relatively small portion of the study area. Also, in terms of National totals, unpaved roads are estimated to emit more than two-and-a-half (2.5) times the amount of TPM, PM10 and PM2.5 than paved roads (Environment and Climate Change Canada [ECCC] 2018b). Unpaved roads and their associated dust were therefore considered a more important emission source, despite that all road-based PM emissions (paved, unpaved, engines, brakes, re-suspension, etc.) are considered relevant and important as part of a complete and 'ideal' MEI.

[11] see: https://prdoas6.pub-apps.th.gov.bc.ca/tsg/

Table 3. Road classifications used in BC's Digital Road Atlas including the corresponding cumulative length (2-dimensional) of each class represented within the study area shown in Figure 1 (GeoBC 2018).

Surface	Description	Length (km)
boat	Some form of boat or water travel required (i.e. ferries)	11.9
loose	A maintained gravel road	14 708.3
overgrown	A road that has become overgrown with vegetation	282.3
paved	A permanently hard surface such as asphalt or concrete	1341.9
rough	An unmaintained gravel or dirt road	21 698.6
seasonal	A seasonal road including winter ice roads and summer recreational roads	313.4
unknown	An unknown surface type	3755.9
	TOTAL:	**42 112.3**

Because traffic data were both temporally and spatially sparse, as were road- and behavioural-characteristics (of humans), mobile exhaust emissions from passenger and/or transport vehicles were not included in the MEI, but were certainly on the list of inclusions and would be a first choice addition to the MEI. There were no sufficient data at the onset of this project to estimate these emissions.

2.3.1 Road Dust Emissions

Environment Canada's Road Dust Calculator (Environment Canada 2008) was used to estimate road dust from unpaved roads throughout the study area. Although the calculator is designed for facility-based NPRI reporting, each census block was treated as a single facility for these purposes; and the following assumptions were made.

The majority of passenger vehicles are pick-up trucks (45%) and SUVs (34%) with 4-wheel drive and on average a 6 cylinder gasoline-based engine. The remainder are primarily 4 cylinder cars with front wheel drive (20%) with the remaining 1% of passenger vehicles being comprised of vans. These estimates are based on the ratios of new and used vehicles at local car lots (e.g. Coast Mountain GM[12] and Hoskins Ford Sales[13], both in Smithers) and are assumed to be representative of the region's passenger vehicle make-up.

All roads with a road surface classified by the Road Atlas (GeoBC 2018) as either 'loose' (a maintained gravel road) or 'rough' (an unmaintained gravel road) were classified as unpaved. Roads with a road surface of 'unknown' were also included in the calculated total length of unpaved road because after visual inspection, most if not all 'unknown' surfaces were connected to a loose or

[12] See: http://www.coastmountaingm.com/
[13] See: http://www.hoskinsford.com

rough road. In terms of road class, these unpaved roads were either 'local', 'resource', or for the most part 'unclassified' and usually leading to resource roads. Road surfaces classified as 'seasonal' were not included due to their relatively small number, and uncertainty as to whether they are winter or summer roads. Being seasonal and having much lower usage, they could also not be grouped with the other roads for the purpose of emissions estimation. In addition, 'overgrown' roads were not included, as they are considered to be used rarely; and when they are, will not produce much surface dust, except perhaps under extremely dry conditions.

In QGIS, SQL queries were used to calculate two-dimensional length of unpaved road classes, and are provided in Appendix A.4. The total length of unpaved roads within the study area was calculated as 40,162.81 km. For comparison, 'seasonal' and 'overgrown' roads totalled 313.384 meters in length. Road length was summed across each census division for use in the road dust calculator, which was then run for each census division independently.

Because nearly 97% of study area road kilometers are unpaved (total road length is 42,112.300 km), it was assumed that most residents outside of municipal limits will drive on an unpaved road twice daily (if they make one trip a day). It is also assumed that, due to the rural nature of the region, most people over 16 will drive, although households may frequently share a vehicle. Since as the area's rural population makes up

So, *average daily traffic* (*ADT*) for each vehicle class driving on the entire unpaved road network can be estimated as:

$$ADT\ (\#/day/vehicle\ type) = (population_{(16-79)})*0.8)*vehicle\ ratio \qquad 2.3\text{-}1$$

Where passenger vehicle classes include pick-up trucks, SUVs and cars with a vehicle ratio based on that of the local car lots (see previous section). The ADT of transport trucks was based on the number of people employed in agriculture and resources or trades (which includes transport) and also multiplied by 0.8 assuming that people in these occupations drive large trucks 80% of the time. Although this may represent an overestimate (i.e. some drive smaller trucks, or equipment that doesn't move as much), the census data relied upon are from 2011 (the most recent available) and it was expected that the population had increased in all census divisions since then.

Calculations of road dust, using the NPRI's Road Dust Calculator (Environment Canada 2008) were preformed separately for each census unit (N=26) in the BVLD, using data for the population between the ages of 16 and 79. It was assumed that there is no anthropogenic dust control, but that natural factors (such as precipitation) mitigate some of the emissions of road dust—by 62%—based on the number of days with frozen or snow covered roads, or >0.2 mm of precipitation.

Because the Road Dust Calculator is designed for facility-based emission estimates, rather than estimates for a municipality, village or census division, it automatically assumes that all drivers drive on each road in the spatial unit every day. Such a blatant overestimate of ADT would significantly over estimate the emissions calculated using it. So, It was assumed only those unpaved road segments classified as 'local' or 'collector' would be used regularly, as opposed to unpaved roads that were classified as alleyway, driveway, lane, recreation, service, resource or trail. The results for census divisions classified as a District Municipality, Town or Village was roughly 1.25% of unpaved roads were of the local or collector class, and for all other communities these road classes averaged

roughly 12.5% of unpaved roads. These values were used as correction factors in the Road Dust Calculator to yield more seemingly reasonable seeming results.

Documentation for ECCC's road dust calculator can be found on the NPRI webpage[14] and more detailed input and output data can be found in Appendix B.3-1.

2.3.2 *Ferry and Barge Exhaust Emissions*

Roads classified as 'boat' represent ferry or barge routes. In the 2015 – 2016 inventory years there were four existing 'boat' or ferry/barge courses within the study area. Two of the routes, both on Babine Lake, have boats named 'Babine Charger' and 'Babine Itinerant', respectively; and transport primarily logging trucks and forestry workers. One of the other 'boat' routes on Tahtsa Reach was used to transport equipment, vehicles, and staff to and from the Huckleberry mine, but made her last voyage on August 31, 2016 when mining operations at Huckleberry were suspended indefinitely. The 'Francois Forester', a passenger ferry used by workers and residents, services the fourth route, on Francois Lake, that runs every 25 min each way from 5:30 am to 11:00 pm.

Boat specifications such as boat type length, and no of trips per day, were found either on-line via search engines, or the route operator was contacted by telephone. Because all the water vessels' transport cargo and vehicles (including full lumber trucks) they were assumed to be of the Roll on Roll Off (RORO) type, with a flat deck that lies near flush with the dock surface for easy loading/unloading. A PM emission factor for 'Ferries and RORO ships' in kg/GTkm (kilograms per gross tonne kilometer) was used to calculate ferry and barge emissions as found in Denier van der Gon and Hulskotte (2010) based on a boat's GT class. Values of GT, for all but the unnamed boat on Tahtsa Reach that no longer runs to Huckleberry Mine, were found in The Nauticapedia, (2019). For the unnamed boat, GT was calculated using estimates of barge size from Catherwood Towing—the operator/owner of the tugboat (named "Sea Imp IV") that pulled the barge. It was assumed that the amount of fuel and PM emissions required for pulling the barge by tug, would be similar to the emissions from a free-running vessel of the same size (volume or GT).

Ferry and barge emissions were calculated as:

$$E_{ship} = EF_{GT} \times GT \times \left(d_y \times \frac{r}{d} \times l_{run}\right) \qquad 2.3\text{-}3$$

where, E_{ship} is the annual PM emissions in t/y,
EF_{GT} is the emission factor for vessels of the same volume class (GT) in kg/GTkm,
GT is the gross tonnage, a measure of volume,
d_y is the number of days the ship operated in year y ,
$\frac{r}{d}$ is the number of runs/routes completed each day, and
l_{run} is the length of the ship run in km

[14] https://www.canada.ca/en/environment-climate-change/services/national-pollutant-release-inventory/report/sector-specific-tools-calculate-emissions/road-dust-unpaved-surfaces-guide.html

If GT is not known (as for the Huckleberry barge) it can be calculated using

$$GT = \mathrm{V}_{ship} \times \mathrm{K} \quad 2.3\text{-}4$$

where, V_{ship} is the ship's internal volume

$$\mathrm{V}_{ship} = l_{ship} \times w_{ship} \times d_{ship} \quad 2.3\text{-}5$$

and, K is a coefficient calculated using

$$K = 0.2 + (0.2 \times \log_{10} V_{ship}) \quad 2.3\text{-}6$$

All parameters and calculations for mobile/linear emissions are given in Appendix B.3-1 and B.3-2 for road dust and barges, respectively. It was considered beyond the scope of this study, and an exercise in futility, to estimate PM emissions associated with leisure watercraft in the study area—they are neither registered nor do they have usual routes.

2.4 Residential and Commercial Heating

All estimates of PM emissions from both residential and commercial heating were calculated using reported energy consumption totals (in Gigajoules [GJ]) from the Community Energy and Emissions Inventory (CEEI) (Government of BC, 2020c); conversion of energy to either mass or volume using relationships from BC Ministry of Environment (2014); and the AP 42 US-EPA emission factors for wood, fuel oil, natural gas and propane (US-EPA 2017). Because the CEEI is community based, it has a much finer spatial resolution than other similar data, and despite being developed for greenhouse gas (GHG) inventorying, this and other GHG data sources may prove valuable in future inventory development and updating, particularly if they are reported and updated annually. Although all "unincorporated areas" (small communities) were all classed together for the Bulkley-Nechako, separate energy usage counts were available for Burns Lake, Fraser Lake, Granisle, Houston, Smithers, Talkwa, Hazelton and New Hazelton—depending on the heat/energy source. Differences between calculations for different sources are discussed briefly in the following, and all used 2015 and 2016 population and usage data

2.4.1 *Residential Wood Burning*

The use of wood as residential heat is fairly common in the BVLD. Wood is relatively inexpensive as a fuel source, readily available, safe and easy to use, and few things can go wrong with a conventional wood stove or fireplace. Some people use woodstoves or fireplaces to cook meals (Figure 7), while others keep them for purely aesthetic reasons. The nostalgia many of us have for the fireside can be deeply rooted and hard to give up, despite the PM pollution it produces.

Wood burning appliances come in many forms, all of which have different emissions profiles. Still the most complete survey/report for wood burning appliances in the BVLD (those for Smithers may lack some context of the rest of the BVLD, [e.g. Millar 2012; Jesse Hiemstra &Co. 2016]) is that of

Rensing (2005). Although it is assumed that total numbers have changed since 2005, including the Provincial Woodstove Exchange Programme (WSEP) having some successes from its implementation at the local level (Allen et al. 2012), it was also assumed that with the exception of WSEP, the ratio of different wood burning technologies has not changed significantly. Therefore Table 3 of Rensing (2005) was used to divide the wood usage data from Government of BC (2020c) into GJ per appliance type (Table 4). These numbers were modified using local woodstove exchange information from Smithers, Telkwa and Hazelton (Sue Brookes and Matt Davey *Pers. Comm.* Oct. 2020) and energy intensity in kg/GJ was calculated using factors in BC Ministry of Environment (2014) to act as a multiplier. The US-EPA (2017) emission factors for woodstoves (Chapter 1, s. 10) and fireplaces (Chapter 1, s. 9) were then used to calculate emissions in tonnes per year—after some unit conversions and assumptions, including:

- Wood combusted for residential heat is completely dry and contains 0% moisture;
- Wood burning fireplace inserts have the same emissions factor as either catalytic or non-catalytic wood stoves (whose emissions differ ≈2%);
- Wood burning boilers and furnaces of all types have similar emissions to conventional fireplaces with inserts and conventional air tight woodstoves (Weinstein 2005, Table 20); and
- Exchanges through the woodstove exchange programme were from conventional woodstoves to certified non-catalytic appliance models.

Figure 7. An indoor wood-burning fireplace with an insert that increases efficiency and reduces PM emissions, being used to prepare a sizzling meal.

The final EF in kilograms per Gigajoule of energy consumed (kg/GJ) used to calculate PM releases from various wood burning appliances, and the factors that went into their development are provided in Table 4. In summary, EF were calculates as:

$$EF_{wood}\left[\frac{kg}{GJ}\right] = EF_{EPA} \times 0.4536\left[\frac{kg}{lb}\right] \times 0.0011\left[\frac{ton}{kg}\right] \times 55.55\left[\frac{kg}{GJ}\right] \qquad 2.4\text{-}1$$

where:

EF_{wood} is the EF for residential wood burning appliances used on the MEI;
EF_{EPA} = the EF from US-EPA (2017) Chapters 9 and 10, tables 1.9-1 and 1.10-1;
and the energy intensity multiplier of 55.55 is actually the inverse energy conversion factor for wood given in kg/GJ rather than GJ/kg (BC Ministry of Environment 2014).

Table 4. Emission Factors (EF) used to estimate annual contributions of PM from local wood burning residential appliances.

WOODSTOVES					FIREPLACES	
Conventional	Non-catalytic	Catalytic	Cert. Pellet	Masonry	Masonry	Value and Units
0.29	0.02	0.04	0.26	0.02	0.37	usage ratio (Rensing 2004)
30.6	19.6	20.4	4.2	5.6	34.6	EF in lb/ton (US-EPA 2017)
0.850	**0.540**	**0.567**	**0.117**	**0.155**	**0.961**	**Final EF in kg/GJ**

Beginning in 2006, the BVLD's WSEP had exchanged 680 old and polluting t sources of heat by 2015. That in 2016 increased to 684 (three in Smithers and one in Telkwa), and the programme continues today (Sue Brookes and Matt Davey *Pers. Comm.* Oct. 2020). These numbers, up to and including 2015 and 2016 (Appendix B.4.1), were used to modify the residential wood heat emissions to reflect changes in technology not captured by the old survey (Rensing 2005) (i.e. the use of certified woodstoves with reduced PM emissions).

Resulting emissions' estimates for residential wood burning are presented in subsection 3.4.1 of this report, while more detailed digital data and calculations for wood heating are provided in Appendix B.4.1.

2.4.2 *Hydrocarbon-based Fuels*

Although residential fuel oil, gas and propane are not known for their PM emissions the way wood fireplaces are, under imperfect conditions (i.e. what exist most of the time) there is incomplete combustion of any fuel, which can lead to un-combusted superfine carbon particles, and hydrocarbons, most of which are <1 µm in size.

Fuel usage data from Government of BC (2020c) in GJ per appliance type were used, and energy intensity was calculated using factors in BC Ministry of Environment (2014). The US-EPA (2017) emission factors for fuel oil (Chapter 1, s.3), Natural Gas (Chapter 1, s.4) and propane (Chapter 1, s.5) were used, along with some unit conversions, to estimate PM emissions from the respective heating source in tonnes/y. Because commercial fuel usage data were also available, they were included using the same methodology (Appendix B.4.2 and B.4.3). Spatial distributions were assigned using SQL queries in QGIS (Appendix A.4) such that unincorporated areas—which are smaller and emit < 1000 tonnes of PM per year—and the municipalities, all received the correct PM release values.

2.5 Resource Management Debris Burning

Burn data reported by major and minor forestry licenses in the study area were obtained from the BC ENV (c/o Ben Weinstein). The emissions equation used in the previous MEI (Weinstein 2005) was used to estimate PM emissions (E_{PM}) in tonnes (t) from resource management and debris burning as follows:

$$E_{PM} = BQ\ x\ \frac{PpnMassConsumer}{100} \times \frac{EF_{PM}}{1000}\left(1 - \frac{\%soil}{100}\right) \quad 2.5\text{-}1$$

where BQ (t) is the base quantity of wood burned; PpmMassConsumer is the percentage of fuel (biomass) that fully combusts (assumed to be 90%); EF_{PM} is the emission factor for TPM, PM10 or PM2.5; and $\left(1 - \frac{\%soil}{100}\right)$ is the adjustment for soil content in the pile (assumed to be 5%).

The BQ is further estimated using:

$$BQ = \#Piles\ \times\ M_{W/p}\ \times \frac{1}{1000} \quad 2.5\text{-}2$$

$$M_{W/p} = V_{W/p}\ \times\ \rho$$

$$V_{W/p} = V_{pile}\ \times\ PackingRatio$$

$$V_{pile} = D_{pile}\ \times\ CorrectionforPileShape$$

$$D_{pile} = L_{pile} \times\ W_{pile} \times\ H_{pile}$$

$$\rho = \rho_{AvgOD} + M_{H_2O/m^3}$$

$$M_{H_2O/m^3} = \frac{MCD}{100} \times\ \rho_{AvgOD} \quad 2.5\text{-}8$$

where,
MW/P (kg) is the mass of wood per pile,
VW/P (m^3) is the volume of wood per pile,
Vpile (m^3) is the volume of the pile,
Dpile (m^3) are the combined pile dimensions (volume),
Lpile (m) is the pile length,
Wpile (m) is the pile width,
Hpile (m) is the pile height,
ρ (kg/m^3) is the wet density of the wood pile,
ρ AvgOD (kg/m^3) is the average oven dry density of the wood pile,
M_{H_2O/m^3} (kg/m^3) is the mass of water per m3 of the wood pile,
MCD is the moisture content of the pile on a dry basis, and
$\frac{1}{1000}$ converts from kg to t.

Emission factors (EF_{PM}) from the CONSUME model, produced by the US Forest Service (2015) and now part of the Fuel and Fire Tools package, were used to estimate PM emissions from resource and debris burning within the BVLD (US Forest Service 2015). In particular, EF_{PM} for Woody Fuel Accumulation (such as piles) were used, are very similar to those used by Weinstein (2005), and were clarified using the documentation from Consume 3.0 (Prichard et al. 2006). Parameters used for all burn areas are given in Table 5.

Tree species were determined using Appendix Table H from Weinstein (2005); and despite the changes in forest districts/timber supply areas (TSA) since 2005, it was assumed, in regards to major licenses, that: 1) Morice and Lakes merged into Nadina; 2) Canadian Forest Products (CANFOR) is in Morice; and 3) Babine Forest Products (BFP) is in Lakes (Weinstein Personal Communication, July 2018). Both Pacific Inland Resources (PIR) and Houston Forest Products Company (HFP) operated in both the Bulkley and Morice districts as defined in their respective burn plans. The tree species' mix and resulting wood densities, calculated as averages from western Canada retrieved from Gonzalez (1990) and using ratios of forest composition for each forest district/TSA, are given in Table 6.

Table 5. Parameters used in calculating PM emissions from resource and debris burning for major and small licenses in the 2015 and 2016 calendar years; most after Weinstein (2005).

Parameter	Value
Combustion Efficiency	90%
Soil Content	5%
Pile size	10 m x 10 m x 5 m
PackingRatio	0.2 (20%)
CorrectionforPileShape[15]	π/8 (≈0. 393)
MCD	25%
EF_{TPM}	10.95 kg/t
EF_{PM10}	7.75 kg/t
$EF_{PM2.5}$	6.75 kg/t

Some burn records were missing dates or other information. For instance CANFOR's 2015 burn plan was missing the field 'Number Piles Actually Burned' and instead the field 'Number Piles Planned to Burn' was used in estimating the PM emissions associated with debris burning by this major license in 2015.

Table 6. Forest species composition (%) and wood densities (kg/m^3) used to estimate emissions from major and small license debris burning in the BVLD using the CONSUME model (Equation 2.6), after Gonzalez (1990), and used for estimating emissions from point sources that included open burning in their facility permit.

	District/TSA	Spruce (black)	Pine (lodgepole)	Balsam fir	ρ_{AvgOD} (kg/m^3)
Low Elevation	Morice	50	50	-	453.25
	Lakes	40	60	-	456.2
	Bulkley	50	30	20	437.35
Mid-High Elevation	Morice	40	30	30	428.65
	Lakes	40	40	20	436.6

[15] Assuming a paraboloid shaped pile (i.e. mound with a circular base).

(>800 m)	Bulkley	50	30	20	437.35

For all minor license burns, forest make-up and resulting wood density were estimated based on the district listed in the BRN. For those licenses without a listed district: Lake Babine First Nation's, Baker and Morris', and Wetzin'kwa Community Forest Corporation's, licenses were assumed to be in the Bulkley Forest District; while Driftwood Timber Ltd. was assumed to be located in Lakes.

3 Results

3.1 Emission Totals and Airshed Patterns

Total emissions of PM from sources within the study area and included in the MEI are given by source type in Table 7. From these results, resource-based open burning and unpaved road dust, are the largest contributors to PM emissions within the BVLD. However, open burning (an activity also included as a part of some NPRI reported and *EMA* permitted sources) is a much more important source of fine particulate matter ($PM_{2.5}$)—the PM size class that poses the greatest risk to human health.

Table 6. Summary of PM sources included in the MEI in tonnes/year (t/y). Totals are given at the top of the table and the top of each source type (bold). Sources under "other heating" including fuel oil, gas and propane, have emissions classified as 'filterable' and 'condensable' fractions of TPM, and are all fine particles of <2.5 micron in size. Their totals (TPM) were therefore added to the total of each TPM, PM_{10} and $PM_{2.5}$.

		2015			2016		
		TPM	**PM_{10}**	**$PM_{2.5}$**	**TPM**	**PM_{10}**	**$PM_{2.5}$**
All Sources	**TOTAL**	**9,803.98**	**4,888.03**	**3,003.16**	**10,129.63**	**5,023.45**	**2,991.99**
Point Sources							
NPRI	Total	**1712.00**	**812.80**	**328.14**	**2310.00**	**1146.80**	**489.66**
EMA Permits	Total	**410.36**	**208.20**	**111.83**	**387.52**	**189.72**	**97.80**
	Forestry	*338.71*	*151.22*	*68.97*	*339.63*	*151.62*	*69.16*
	Mining	*71.64*	*56.99*	*42.85*	*47.89*	*38.10*	*28.64*
Linear Sources							
Transportation	Total	**3878.06**	**1096.63**	**112.09**	**3888.56**	**1099.51**	**112.27**
	Unpaved Rd	*3875.36*	*1093.93*	*109.39*	*3885.98*	*1096.93*	*109.69*
	Boat	*2.699*	*2.697*	*2.697*	*2.587*	*2.585*	*2.585*
Area Sources							
Open Burning (BRN)	Total	**3473.37**	**2458.32**	**2141.12**	**3208.78**	**2271.05**	**1978.02**
	Major	*3389.88*	*2399.23*	*2089.65*	*3187.19*	*2255.77*	*1964.70*
	Minor	*83.49*	*59.09*	*51.47*	*21.60*	*15.28*	*13.31*
Residential Heating	*Wood*	**331.71**	**307.70**	**305.62**	**336.69**	**312.50**	**310.38**
Other Heating	Total	**4.3752**	**3.2780**	**1.3158**	**3.8604**	**3.2149**	**1.2974**
Other Residential	*Fuel Oil**	*1.060*	*0.6424*	*0.4175*	*1.076*	*0.6520*	*0.4238*
	Gas	*1.4579*	*1.0935*	*0.3645*	*1.4290*	*1.0717*	*0.3572*
	Propane	*0.2041*	*0.1458*	*0.0583*	*0.1841*	*0.1315*	*0.0526*
Commercial Heating	*Gas*	*1.6621*	*1.2466*	*0.4155*	*1.6014*	*1.2011*	*0.4004*
	Propane	*0.2097*	*0.1498*	*0.0599*	*0.2221*	*0.1586*	*0.0635*
		TPM	**Filterable**	**Condensable**	**TPM**	**Filterable**	**Condensable**

*of the condensable fraction of PM from fuel oil, a ratio of 0.65 is inorganic and 0.35 is made up of condensable organics.

3.2 Large Stationary Sources

All individual NPRI sources within the study area (Figure 1) are listed in Appendix B.1 along with accompanying source information, GPS coordinates, operating schedules and annual emission totals Basic source data and the results of annual emission concentration estimates (in t/y) for

sources permitted provincially under the Environmental Management Act are found in Appendix B.2.

Sources with emission permits under the *EMA* are presented alongside NPRI sources as they too can be classified as 'large' and 'stationary'; and a comparison between these two types of sources/datasets is particularly useful in future MEI updating and planning. More overlap was anticipated to occur between the two data sources than was found. Only one source, Pinnacle Renewable Energy Inc.'s pellet plant near Burns Lake reported to the NPRI and had a provincial permit. In this case, the value used in the inventory reflects the permitted PM release, rather than that reported to the NPRI. This value will be higher as it represents the maximum emissions a facility may emit; however, this is in line with the worst case scenario, and precautionary approaches that help ensure public, and environmental, health protection.

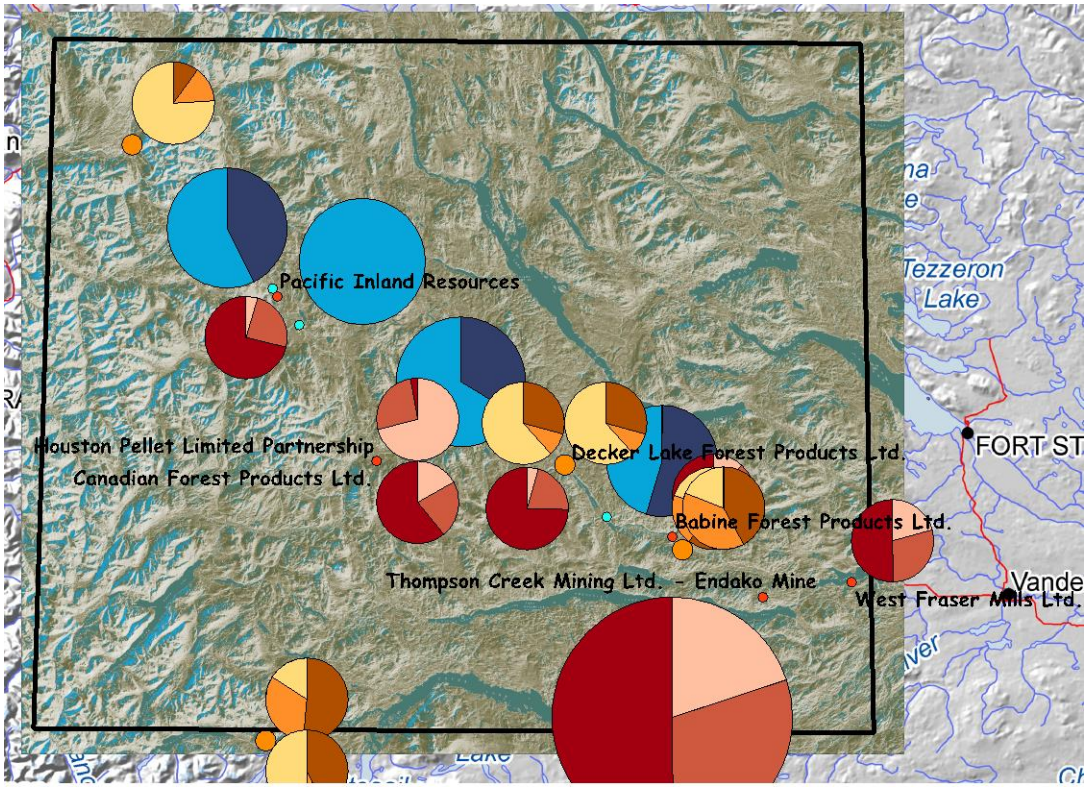


Figure 8. Map of PM monitoring locations run by ENV (blue), and large stationary ('point') sources—those reporting to the NPRI (red) and those with provincial permits (brown) for 2015. Marker size is relative to PM ambient concentrations or stack emissions, and dark colours represent coarse particles (PM_{10}) whereas light colours represent the fines ($PM_{2.5}$). Smaller circles without pie divisions represent the actual physical location of the emission source or monitoring stations, but act as labels because diagrams were offset for visualization purposes.

Figure 8 above, displays both NPRI (red) and permitted (brown) sources and emission intensities in relation to one another for the 2015 inventory year, and the MOE monitoring stations (blue), each showing the relative contributions of PM_{10} (dark) and $PM_{2.5}$ (light). There are a few point sources that dominate the PM emissions landscape, these are: Thompson Mining Inc.'s Endako Mine, Pinnacle Renewable Energy Inc.'s pellet plant in Burns Lake, the CFP Mill in Houston, and the West

Fraser Mills Sawmill in Fraser Lake, each emitting more than 110 t/y in 2015. The amount of PM emitted by the large stationary sources differs only slightly between the two years.

3.3 Mobile Sources

Dust from unpaved roads was estimated as the greatest contributor to TPM in the BVLD in 2015 and 2016. However, road dust emissions are notoriously over-estimated and although 'fugitive dust' is always a primary source in emissions inventories, it only represents a small portion (≈ 10%) of PM measured in ambient air (US-EPA 2011).

Because the ECCC road dust calculator was developed for the assessment of facility-based unpaved road emissions, it assumes that 100% of the unpaved road network within facility boundaries (census district) is travelled by 100% of drivers. But in the BVLD it is very unlikely that 100% of people in the more urban areas drive on unpaved roads at all, or that rural dwellers drive all roads in their respective districts daily.

Therefore, drivers were assumed to be those between the ages of 16 and 79m 80% of whom were expected to make a trip each day, and these trips would have occurred on 'local' or 'collector' classes of unpaved roads, representing roughly 1.25% and 12.5% of unpaved roads in urban and rural areas, respectively; thereby reducing PM emission estimates and making them more 'realistic' (see Appendix B.3). While the accuracy of such factors is questionable, they are considered more representative than the default models assumptions in this case.

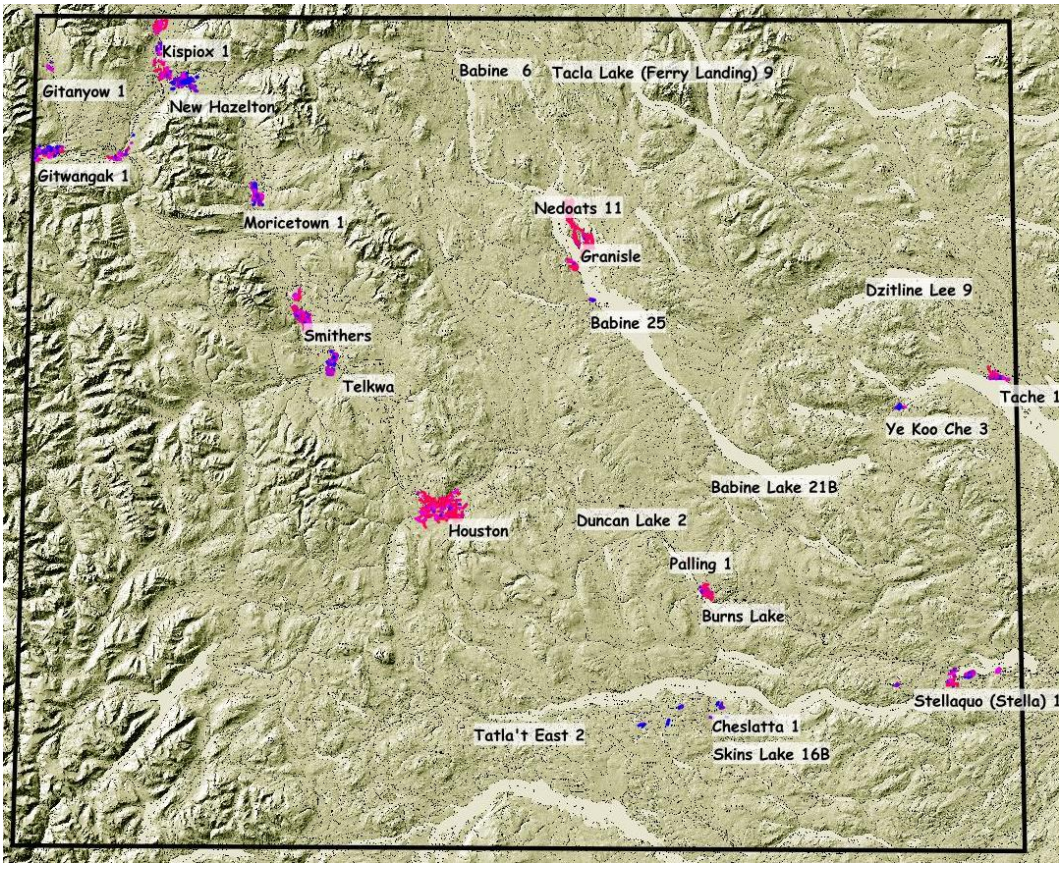
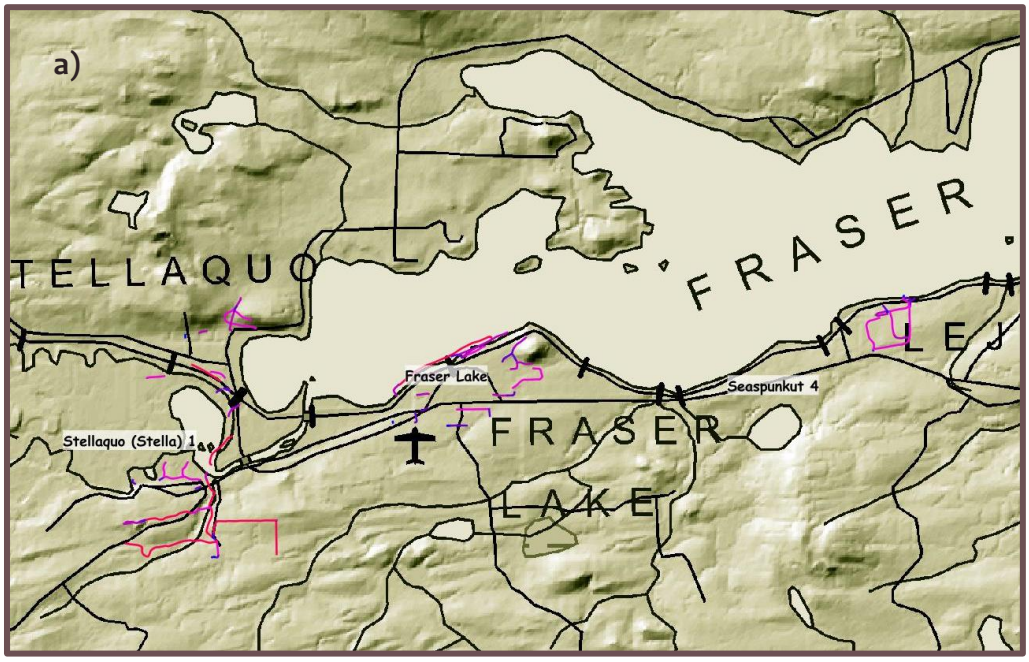


Figure 9. Results from the road dust calculator for each census division (incorporated community). Red represents the highest PM emissions per road segment with transitions to pink, purple and blue–blue being the lowest emissions rate. Scale = 1:50,000.

Despite these modifications road dust emissions are still concentrated in and around population centres, where there are also a higher density of roads in general. The roads are coloured in a gradient from red to blue with red representing the highest TPM emissions. Some roads in smaller communities have lower emissions per road segment (more purples and blues), which is likely an indication of fewer drivers / a smaller population size (e.g. Tatla'ţ East 2, and Cheslatta 1) Figure 9). However, other communities such as Stellaquo, Fraser Lake and Seaspunkut (Figure 10 a) appear to have more drivers per dirt road (or fewer dirt roads per driver), and PM emission rates per segment that are more similar to larger communities, but where dirt roads are used.



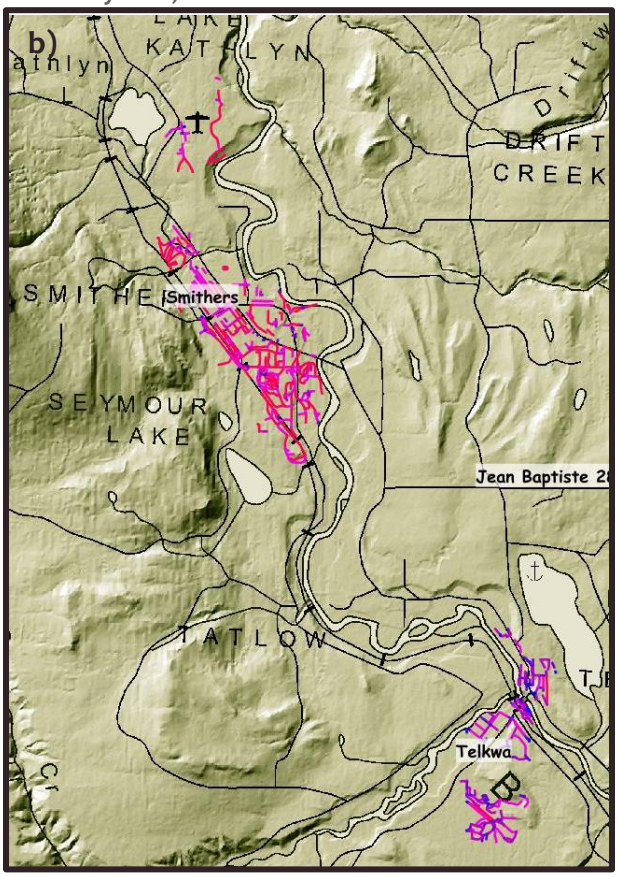


Figure 10. Larger scale results from the road dust calculator, for communities a) in the southeast region of the study area, on the southwest shores of Fraser Lake (Stellaquo, Fraser Lake and Seaspunkut), scale = 1:25,000; and b) towards the northwest end of the Bulkley Valley and corner of the study area (Telkwa and Smithers), scale = 1: 50,000.

Although road dust represents a "mobile" or moving emissions source, the speed of travel and rates of dust plume dispersion, etc. are not included here. This rate feature will need to be included should these emissions be used for modelling as a mobile source. Temporally speaking, unpaved roads are likely more travelled in the fair weather months; the time of year when dust would be more likely to become suspended from ice-free and dry unpaved surfaces.

Here road dust emissions are represented as linear sources (road segments), each allocated a portion of its census district's emissions based on a ratio of road segment length, to total length of roads in the district polygon (Figure 9). Although far from perfect, being calculated and allocated by census district and road segment provides a much higher

spatial resolution of road dust emissions than an average for the entire study area.

Figures 10 a) and b) illustrate this spatial resolution well. In 10 a) there is a close-up (1:25,000) of the Fraser Lake area showing estimates of unpaved road dust emissions for the communities of Stellaquo, Fraser Lake and Seaspsunkut. From the model results it appears that each of these communities has well-travelled dirt roads. The mapped road locations and layout also seem to make them a logical fit for the most travelled unpaved roads. Roads on the north shore of the lake look as though they too would be well travelled, with the exception of a few collector road segments (e.g. Stella Rd.) those pictured are largely paved and therefore, were not included in this emissions modelling exercise. These black coloured roads in Figure 10 a) are also in a region without population data. So, although the magnitude of road dust is thought to be over-estimated throughout the study area, the spatial extent and distribution of unpaved road dust is likely underestimated. Those roads with estimated emissions are considered the most important unpaved roads in terms of dust production, but some of those emissions should likely be spread throughout less populated parts of the region.

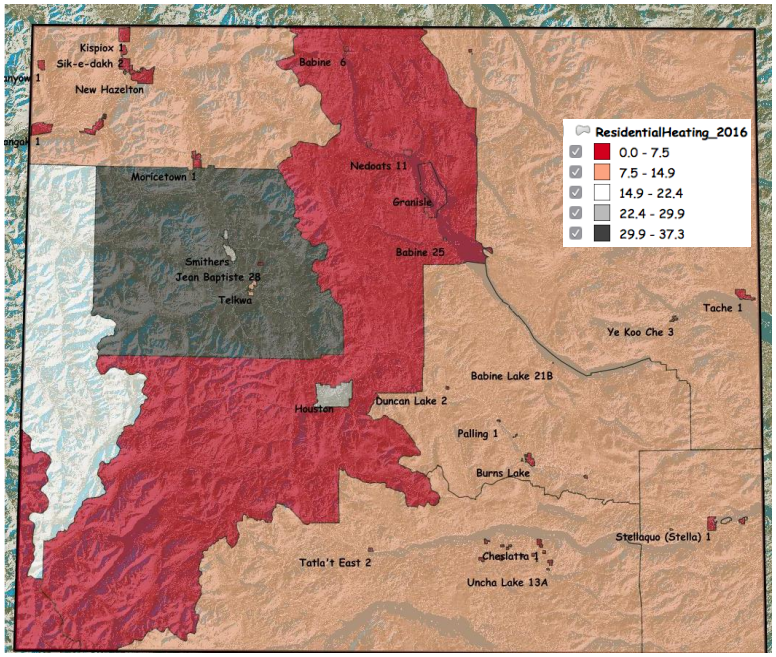
The lakeshore location of the communities in Figure 10 a) and the deep valley location of those in Figure 10 b) mean that road dust (and other PM emissions) will be heavily influenced by local-scale atmospheric processes: lake effects and mountain/valley breezes, respectively (Section 1.2). Rain-out on the downwind side of Fraser Lake for instance, would reduce the amount of road dust emissions and help scrub the air of other locally emitted or transported PM, which is anticipated to be on the western side where the communities are located.

3.4 Residential Heating

The burning of wood in a woodstove or fireplace, as either a primary or supplemental heat source, receives a lot of attention in terms of PM emissions. Maybe it's the signature odour of wood-smoke, a characteristic owed to it by polycyclic aromatic hydrocarbons (PAH); or maybe it's because the some of the smoke particles are fairly large, making the emissions and their source easier to see.

Residential wood burning emissions are highly variable and a function of various wood characteristics and operating practices. Most of the emissions from wood fires are the result of incomplete combustion. In general, conditions that promote a fast burn rate and high flame intensity enhance combustion and thereby lower emissions. Conversely, higher emissions will result from a slower burn rate and lower flame intensity. But, such generalisations apply more correctly to the earlier stages of the burning cycle, when significant quantities of combustible volatile matter are being forced from the wood (a combination of increased temperature, pressure and the chemistry beginning to occur). Later in the burning cycle, when all volatile matter has been driven out, the charcoal that remains burns with relatively few emissions (US-EPA, 1996b). This is why PM emission spikes, or streams of smoke from chimneys, are generally observed at times of wood loading.

Looking at residential heating emissions by census grouping (Figure 11) there appear to be some discrepancies between different areas and the amount of energy used and/or emissions produced from residential heating. For instance, the Municipality of Houston and the district of Bulkley-Nechako A—home to Smithers and Telkwa —are the most PM intensive jurisdictions in the region. However, if we display the same information on a per capita basis (i.e. divide TPM emissions by the population residing in each area), the emission distribution is much more even, and all categories, based on the quantile of PM, have 6.9 kg/person for at least one (if not both) ends of their range (Figure 12). This could be used as a regional residential heating emissions factor for PM in kg/person. However, Smithers, Houston and Telkwa have per capita heat EF of 25.3, 22.7, and 11.9 kg/person years, respectively.



Although emissions of PM from propane, NG and oil are lower than most wood burning appliances, it is important to remember that they are not free from PM emissions and need to be properly installed and maintained to achieve the highest combustion efficiencies. The emissions of PM from fossil-fuel based heating are of a similar magnitude to those of backyard burning as estimated by Weinstein (2005).

Figure 11. Emissions of TPM from residential heating by census division (2015 and 2016).

It should also be noted, that rather than being divided into PM_{10}, and $PM_{2.5}$ size classes, liquid fuel emissions are classified as either 'filterable' or 'condensable' due to their inherently small particle size, with filterable particles being those that can be directly measured/captured by PM filters (usually >0.3 µm) and 'condensable' being finer particles or vapours/gases that condense into aerosols shortly after atmospheric release (see Glossary Appendix A.1).

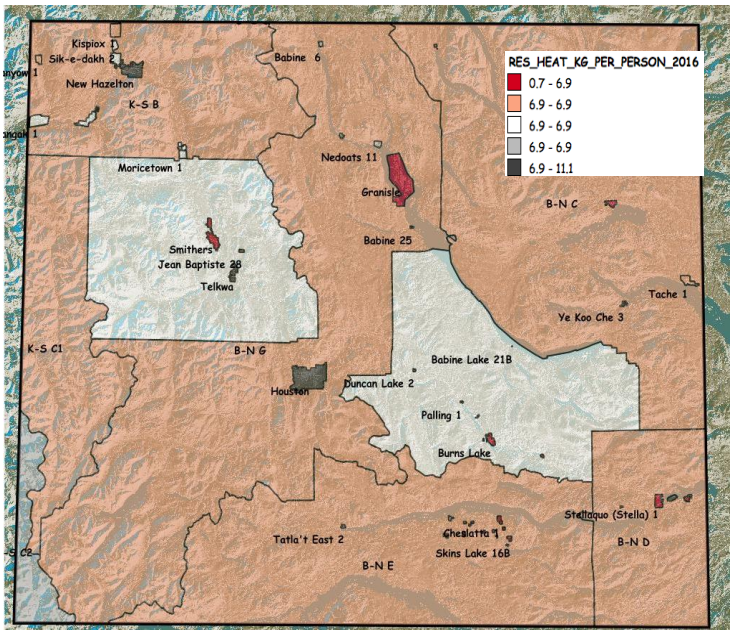


Figure 12. Per capita residential heating emissions by census division for 2016.

3.5 Forestry, Natural Resource and Agricultural Burning

Open burning was a common activity included in Permits given under the *Environmental Management Act,* representing five of the fifteen individually permitted study area sources (See Appendix B.2-3). Open burning sources in permits, unlike those presented here, make up part of a facility permit that also includes other resource extraction activities (forestry and mining) use of fuel, etc., and burning is a regular enough activity to warrant a permit. Therefore, in effort not to separate sources within a facility permit, these open burning events are included in the previous section (3.2) on (permitted) large stationary sources rather than under open burning. That being said, open burning represents the largest source of both PM_{10} and $PM_{2.5}$ in the study area, and if the burning from permits were included, TPM emissions from open burning would have an increased 2015 and 2016 atmospheric loading of approximately 57 and 39 tonnes, respectively.

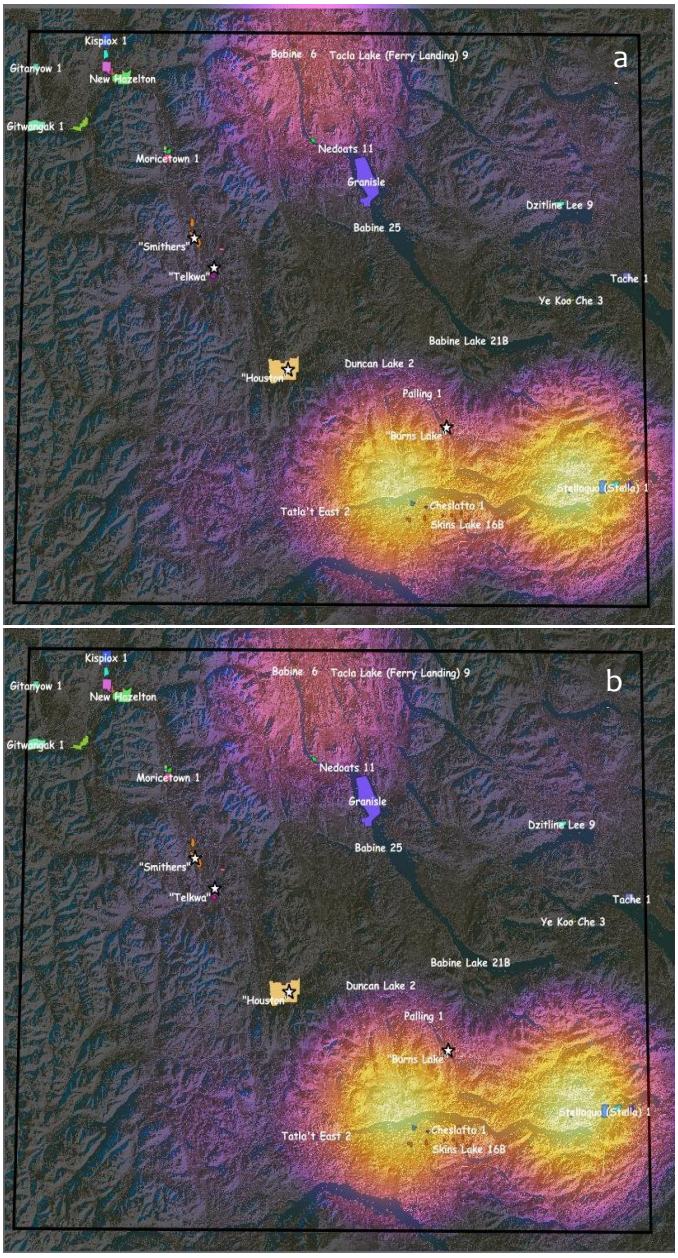


Figure 13 shows 'heat maps' of autumnal (Sept. – Nov.) open burning within the study area in a) 2015 and b) 2016, with no obvious difference between years. Heat maps are used to display spatial densities, where the heat surface is drawn based on an intensity field (in this case TPM) and a distance of influence. In this case (Figure 13) a 50 km (100 mm at a 1:500,000 scale) radius was chosen to reflect how far the smoke from any given burn might travel, and both years were given identical parameters. However, the smoke plume(s) could likely travel much further and Figure 14 shows a heat map produced for 2015 with a 100 km radius of influence for comparison. There were also a number of burns just outside of the study area that would too be capable of influencing local air quality.

Figure 17. 'Heat maps" showing the intensity of TPM emissions from resource sector burning in 2015 a) and 2016 b) with a 50 km influence radius

The region in the southeast of the study area seems to be the most affected by open burning and related wood smoke. An area that encompasses Fraser lake, Skins Lake, Tatla't East 2, Cheslatta and Burns Lake, this seasonal burning may explain why Burns Lake has the highest levels of PM_{10} in the fall, when compared to the other monitoring sites (see Figure 8). However, the prevailing winds trace the Bulkley River valley from the southeast to the northwest such that open burning emissions of PM make their way to Houston and Smithers, but to a lesser degree. In the central north, communities such as Babine 6, may be influenced by local smoke (PM) during burn season. But, the radius of influence from resource burning is likely larger than 100 km (Figure 14); and the associated PM emissions are expected to spread far beyond the confines of the valley, or of Figure 14. It is also improbable for PM plumes to spread laterally in a perfect circle, unless there is no wind whatsoever, and a smooth homogeneous surface. So, although heat maps are a visually appealing way to present emissions from numerous densely packed sources of similar size, it does not capture the intricate and limitless dance of an air parcel, carrying wood smoke particles across the BVLD's complex terrain.

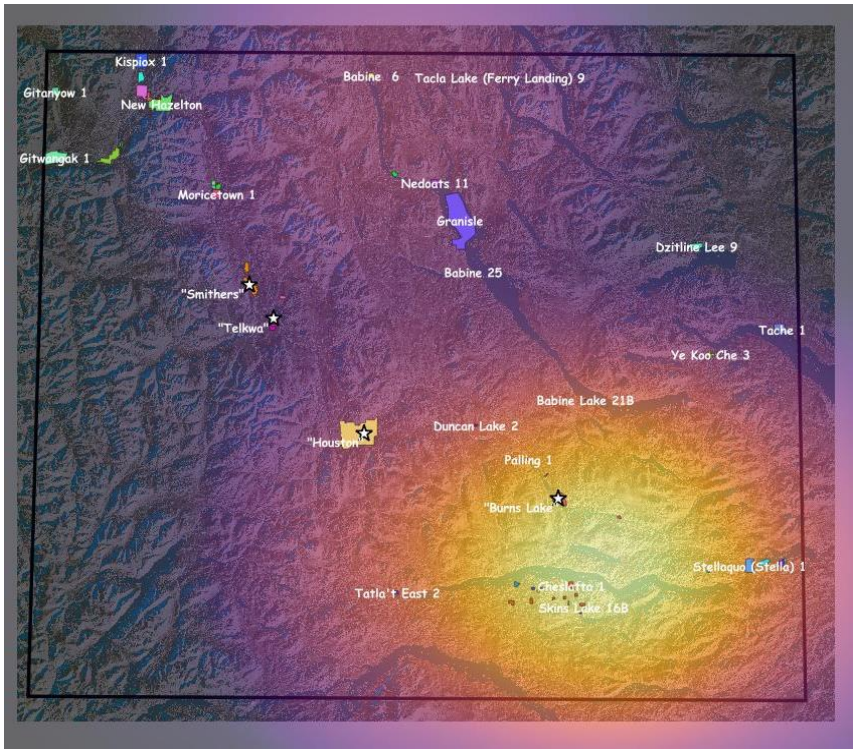


Figure 13. 'Heat map" of resource burning emissions in 2015, with a 100 km radius of influence.

4 Discussion

4.1 Trends and changes over time

A few changes have occurred in the BVLD since the 2000/2001 MEI was developed by Weinstein (2005) that affect local air quality. One major step, and perhaps the most influential, was the discontinuation of beehive burners in the forest sector throughout BC—a source type that contributed nearly 10% of the area's emission in 2000 and 2002: 1654 t/y and 1769 t/y of TPM to the airshed, respectively. So important were beehive burners as emission sources that they were deservedly given their own emissions category in Weinstein (2005). They were also an icon of BC's sawmill culture, but saw their last days in December 2007 as per the *EMA's Wood Residue Burner and Incinerator Regulation* (B.C. Reg. 22/2006) designed to improve air quality in forest–dependent communities across the province. Even without the inclusion of beehive burners, PM emissions from permitted sources are much lower than in the previous MEI (less then one fifth). However, the specific emission sources have all changed and there are no similarities between permitted sources whatsoever. However, when the NPRI emissions are included with the permitted data, emission totals from the large stationary sources included here, are much closer to the permitted emissions of Weinstein (2005). Particularly interesting is that there are similarities between the previous MEI permitted sources, and this MEI NPRI reported sources, including: CANFOR, PIR, Houston Forest Products, Decker Lake Forest Products and Babine forest products. Some differences in point source emissions may be due to artefacts of study area boundaries. Although this and the previous MEI used very similar study areas, a difference of a km or the rounding of boundary coordinates can have a significant impact on source inclusion, and therefore, emission totals. For instance when considering a buffer area around the study area from which to include emissions, there were an additional four NPRI sources, totalling nearly 500 t of TPM in 2015 that would have been included. For permitted sources, there was a as single facility, a mine, that would have added approximately 1800 t/y of TPM to the 2015 and 2016 MEI.

Another shift that has contributed to an improvement in the airshed's air quality is WSEP, which has been fervently implemented at the local level since 2006. Most active in the communities of Smithers, Telkwa and the Hazeltons, by 2018 the program had swapped out a total of 723 inefficient woodstoves for improved efficiency models. . The cash incentives (rebates) offered by the program, help ease the cost burden on homeowners and makes the swap a practical option for many. Assuming a switch from conventional to catalytic woodstoves, and the annual burning of four and a half tonnes per stove (from Weinstein, 2005) this exchange program has reduced PM_{10} emissions in the BVLD by approximately 17 t/y by 2016. Although this may not sound like much against an airshed total ≈337 t/y for wood heat alone, it is comparable to the PM emissions from all other heating sources and barge/ferry exhaust, combined. If the stoves exchanged were from conventional to masonary woodstoves, PM_{10} levels decreased by 15.81 t/y. Either way, or somewhere in between, it's a step towards clearer air in the BVLD.

However, the woodstove exchange certainly doesn't account for the >40% decrease in TPM from residential wood heating between 2001 and 2016 (572 and 33 1, t/y respectively). So there has either been a general change or reduction in wood burning over 15 years that isn't documented, or it has something to do with a difference in methodology. It seems to have been the latter, a difference in

methods. Weinstein (2005) began with spatially disaggregating wood heating emissions using "rural" and "urban" (incorporated and unincorporated municipalities). Here, estimates were made for each individual incorporated municipality and then for the remaining unincorporated communities using population ratios and emissions for the remainder (or entire) region (Appendix 4, Figures A,4-2, A.4-3, A.4-4).

Different types of wood burning appliances were allocated based on the survey results of Rensing (2005), simply an updated version of the Rensing (2004) relied upon by Weinstein (2005); none of the changes involved the BVLD and both inventories assigned wood usage on a per capita basis. Also, these values were used only to determine the relative contribution of each stove type to each area (census division) to ensure that appropriate emission factors would be applied. Both MEI relied upon AP-42 emission factors for woods stoves, Weinstein's (2005) from 1996, whereas Fifth Edition AP-42 (2017), were used here. That being said, the 1996 EF for fireplaces and wood stoves are still used in AP-42 so this is not where the difference lies.

The only place where this difference could arise is from the usage rates. Weinstein (2005) derived usage data in tonnes/year of wood fuel per appliance type from Rensing (2005), based on volume and density; whereas here usage was based on consumption totals in GJ/year from CEEI. Because the CEEI is reported to at a community level, the determination of factors such as fuel consumption rate may vary between jurisdictions or even between years. The Best Practices Methodology (BC Ministry of Environment 2014) does not provide guidance on collecting usage rates, only on the use of emission factors and calculation of GHG emissions. So, it remains uncertain whether the reduction in residential wood burning emissions is from a true reduction in emissions (updated appliances, drier wood, fewer households using wood for heat, etc.), or whether it is due to unintentional discrepancies in the usage rates of wood, and presumably other heating fuels as well. However, we can not determine which values are correct, or even which are closest to being correct.

Although the annual total TPM for the study area in Weinstein was nearly double that estimated here, the discrepancy comes almost entirely from road dust emissions (3878 t/y for unpaved 2015-2016, and 9726 t/y for paved roads 2000-2001), which are dominated by the coarser particle fractions represented by TPM. But neither set of road dust emissions bears any legitimacy on actual PM contributions, and they are incomparable because they estimate emissions from different road types: unpaved and paved, which release suspended particles through different processes and have particles of differing chemical make-up. The totals for PM_{10} and $PM_{2.5}$ are more comparable, and actually become very similar on removing the influences of beehive burners and road dust (Table 7). Only half of the paved road dust emissions were removed in the top row comparison because it puts them at a magnitude more similar to those (unpaved) road dust emissions estimated here.

Table 7. Comparison of source totals from the MEI described here (2021) and the earlier MEI of Weinstein (2005). An additional study area comparison is given for 2005 emissions minus those of beehive burners (which have since been regulated out of existence) and half of the paved road dust.

	MEI Report/Year	TPM	PM10	PM2.5	TPM	PM10	PM2.5
	2005-(beehive +dust/2)	11,090	5,053	3,095	11,637	5,481	3,477
TOTAL	2021	9,804	4,888	3,003	10,130	5,023	2,992
	2005	17,607	6,890	3,983	18,260	7,376	4,408
Point Sources	NPRI + Permits	2,122	1,021	440	2,698	1,337	587
	Permits	2,971	1,676	703	2,853	1,625	675
Linear Sources	Unpaved + Ferries	3,878	1,097	112	3,889	1,100	112
	Paved	9,726	1,864	446	9,726	1,864	446
Open Burning	Major License 2021	3,390	2,399	2,090	3,187	2,256	1,965
	2005	2,104	1,491	1,270	2,370	1,679	1,430
	Minor License 2021	83	59	51	22	15	13
	2005	564	402	348	964	693	598
Wood Heat	Wood 2021	326	308	306	331	313	310
	Wood 2005	572	540	540	572	540	540
Miscellaneous	Other Heating 2021	4.38	3.28	1.32	3.86	3.21	1.30
	Beehive burners 2005	1,654	905	665	1,760	963	708

In addition, one of the point sources in this inventory that self-reported to the NPRI in 2015 and 2016—the MEI's largest single point source—was apparently placed on early suspension in July 2015, laid off most employees, and has remained that way ever since. Oddly enough, the suspended mine was a still able to produce 1227 tonnes of TPM in 2015, and 1661 tonnes in 2016, the latter for which the mine was not operating. The majority of the Endako mine's PM emissions are coarse and >10 µm in diameter, indicating that the PM is fugitive dust from the physical mine or the mechanical processing of ore, rather than PM from a combustion source such as the mill. But, according to local news both milling and mining have been suspended, and only "care and maintenance and environmental obligations" are being carried out by it handful of remaining employees.[16]

Unlike provincial permits, records in the NRPI do not always list the source type or activity producing a particular pollutant, which can make the NPRI a modelling nightmare. However, according to the 2015 record of emissions reported by Thompson Creek Mining ltd. to the NPRI, the majority of PM emissions are fugitive and there is a small amount of road dust. The comments read "Site non-operating. Increased fugitive dust from tailings facilities (water)." So, contrary to what we think or assume will happen—for instance that even a temporary closure of a mine or other emission source will reduce PM emissions in an amount relative to the length of closure—we see the opposite, an increase in PM emissions with activity suspension.

Thompson Creek Mining made their last NPRI submission in 2017, reporting the release of 432 t of TPM, so it isn't known what the emission situation may be like now. If they went down from eleven

[16] https://www.burnslakelakesdistrictnews.com/news/what-happened-to-endako-mine-employees/

to nine full-time employees they would not need to report to the NPRI due to the 20,000 employee hour per year reporting threshold. In addition to PM, pollutants reported for Endako to the NPRI up to and including 2017 are: ionic cyanides (used as a reactant, likely coming from tailings), manganese and compounds (a by-product released to surface water), and selenium (released to creeks, no information on source). When in operation many additional pollutants, mostly metallic compounds, are released at the facility. A surprising 12 tonnes of mercury was released in 2014, just before closure, as either waste rock or tailings. These are the same tailings that have been producing high levels of fugitive dust since the mine's suspension in early 2015. Despite most of the dust particles being to large to inhale into the respiratory tract, particles that are made of, or laden with, certain forms of metals can be particularly toxic and cause human or environmental health impacts through other routes of exposure (e,g, dermal or oral exposure to settled dust on food and surfaces).

In contrast with the Endako Mine, some sources reduce their PM emissions over time. For instance, even the Huckleberry copper mine that had a temporary suspension beginning in September 2016, reduced emissions from 2015–2016. However, these reductions are a mathematical artifact of mine activity, on which PM emissions are dependent based on traditional EF. Unlike the Endako mine, that reports actual (estimated) emissions to the NPRI, with the Huckleberry we are left with a modified (or non-existent) activity schedule and must decide whether they would emit more, or less, PM during suspended operations.

Still other shifts in activities may continue to increase PM emissions. For instance, there has been an apparent increase in usage of BC Parks and Recreation Campgrounds, combined with the seeming necessity of having a diesel- or gasoline-fired generator in addition to the traditional campfire—activities that have led to an obvious and visible increase in PM emissions. This is just one of many additional PM sources to consider including in any future improved and updated MEI for the region. A local resident identified this 'PM problem' in parks, and there are likely more PM issues at the community-level that may not be thought of until experienced. Community participation may help ease the burden of inventory development, help fill in important gaps in knowledge and data and, depending on the circumstances, help bridge the public and policy.

With the exception of Pinnacle Renewable Energy and Decker Lake Forest Products Inc., all of the NPRI sources in the study area reported an increase in PM emissions between 2015 and 2016 (Figure 15. Although the permit information for Pinnacle was used for the MEI, permits are generally issued once (stave the odd amendment to add or remove an emission source), and therefore the 2015 and 2016 permit-based emissions are identical. Because both NPRI and permit information were available for Pinnacle, and it is one of the major point sources, its NPRI emissions are also shown in Figure 15. Not only do Pinnacle's emissions decrease from 2015 to 2016 (the only emitter that can claim this), but Pinnacle also illustrates a perfect example of the imperfections involved in emissions estimates and choice of datasets/information.

The permit information used to estimate Pinnacle's PM emissions for 2015 and 2016 (which don't capture the emissions decrease, and would not have captured an emissions increase either) include separate emissions estimates for each of the six processes at the facility requiring an emissions permit. Summing these processes gives TPM values that are roughly double the NPRI reported

release, a marginally increased PM_{10}, and a lower $PM_{2.5}$! Assuming the source has a stable size class ratio, the relative distributions of TPM and PM2.5 shouldn't vary much across years or data sets.

But emissions were calculated using permit information, representing the worst-case scenario, or highest permissible emission rate, such that the drastically increased TPM emissions aren't very surprising. However, $PM_{2.5}$ emissions being higher than the worst-case scenario seems more like evidence of error. NPRI emissions are self-reported as annual emissions (actual) so there is no need to calculate anything for, or with, them. On the other hand, permit derived emissions were calculated (Eq-n 2.2-1) and do rely on some standards, including the size class ratios presented in Table 2 in order to calculate $PM_{2.5}$ and PM_{10} from TPM. It is within these size class calculations; the error that lead to the reported value exceeding it's maxima regulatory level. Pinnacle's PM emission estimates used the size class factors for 'drying kilns' and 'cyclones (most)'; and it is within these factors that the contribution of PM2.5 to TPM is being underestimated.

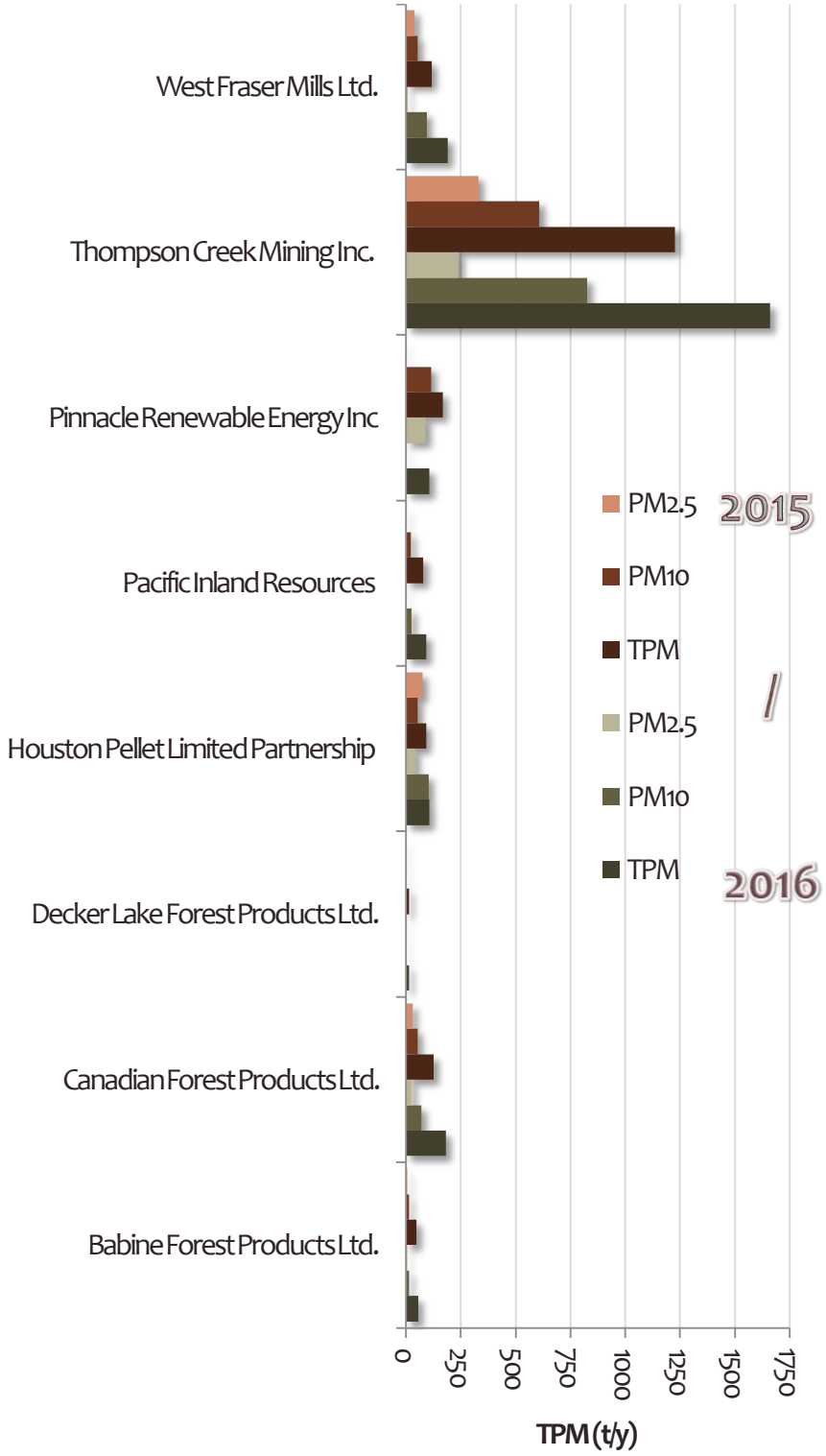


Figure 15. Bar chart of PM emissions from NPRI reporting facilities in 2015 and 2016.

This illustrates some of the error and uncertainty associated with the use of a single year of data in emissions inventories, error associated with using a mix of data sources, uncertainty in applying one-size fits all parameters or methods (e.g. emission factors), and to some extent attests to the lack of industry incentive to reduce PM emissions.

From Figure 15 it also becomes apparent that the Endako Mine emitted more PM [TPM = 1227 (2015); 1661 (2016)] than all of the other NPRI sources combined [TPM = 654 (2015); 758 (2016)], despite having ceased all operations halfway though 2015. However, there is no way to verify these emissions, as they are self-reported and all additional information comes at the discrepancy of the emissions reporter, although it is un likely that they would lie to show greater emissions than what occurred/ .

4.2 Data Quality, Accuracy and Sources of Error

It is crucial for the reader of this report, or user of the MEI to understand that any and all of the numbers presented in the MEI are estimates, not absolute values or measurements. An estimate is largely a numeric educated guess about something we know about, but don't know the definitive answer.

Another source of error is the use of maximum value estimates, or worse case scenarios, particularly in terms of *EMA* permitted sources. The probability of them being precise measures of actual emissions is very low, meaning these source PM emissions are routinely overestimated and in real life actually lower (either that or they are consistently exceeding the permitted discharge amount).

Source types that were completely omitted from the MEI are another source of error and uncertainty. For instance the omission of seasonal and overgrown roads lead to study area road length being reduced by 0.67% and 0.74% causing insignificant error. The emissions associated with these roads would have been so uncertain, that including them likely would have introduced an even higher level of error due to the number of estimated unknowns. Greater overall error is also introduced by the lack of vehicular emissions, paved road dust, and secondary particles from non-fuel vehicular sources.

In retrospect it would be ideal to have somewhat accurate estimates of dust production from both paved and unpaved roads, as they would have differing chemical, temporal and spatial profiles and complement one another in illustrating PM diversity. But the methods used here and elsewhere are known to over-estimate road dust emissions, and therefore also overestimate their contribution to PM in inventories. One reason for this over-estimate is the lack of differentiation between 'suspendable' and 'transportable' PM (Weinstein 2005), the former being able to become aloft in air, the latter being able to travel distances from its source and remain aloft for much longer periods of time. Larger or coarser PM particles, such as PM_{10} and greater, settle from air very quickly, whereas finer particles ($PM_{2.5}$) can remain suspended in air for weeks, providing adequate time to be transported long distances, particularly when emitted from an elevated release point (small stack). Transportable PM is also sometimes referred to as respirable PM because particles of > 10 µm are too large to be inhaled into the human respiratory tract (Johnson 1998).

In addition to their larger size, unlike most other inventoried emissions, road dust particles do not have an elevated release or thermal buoyancy provided by the heat of combustion, and therefore settle very quickly, rather than being transported laterally over large distances. Unpaved road dust contains ten times more PM_{10} than $PM_{2.5}$, and is more than 70% coarse particles > 10 µm in diameter, still part of the TPM size class because they are suspendable. Additionally, originating from the

ground gives road dust a very short distance to travel before depositing on the surface (section 1.5). Over-estimation of atmospheric instability (vertical mixing generated at least partially from buoyancy) has also been found to significantly overestimate road dust emission rates in models such as CALINE (US-EPA 2011).

An analysis completed by the US-EPA (2011) found that local silt loading and specific vehicle speed and weight measures greatly enhanced the accuracy of road dust predictive equations, making their results closer to those of monitored values. However, this site- and vehicle-specific information is not readily available, and trying to estimate these parameters at a finer resolution would not only require more data and computing, but also introduce additional sources of error.

Still neither the unpaved road estimates in this updated MEI, nor the paved road estimates of Weinstein (2005) take into account what may be the largest source of vehicular PM—fine particles from brake and tire wear (Winkler et al. 2018). Furthermore, the actual paving of roads (with asphalt) produces considerable PM in the form of secondary condensable hydrocarbons (or secondary organic aerosols). It was found that when exposed to moderate sunlight, emissions of secondary aerosols increased 300% and emissions doubled (a 100% increase) when subjected to typical summertime temperatures.

Because the application of NPRI's Road Dust Calculator (Environment Canada 2008) relied on both census data (to determine the number of drivers) and unpaved road length, urban centres were predicted to have the highest level of PM emissions. Many of the remote census divisions have no population data, meaning that PM emissions were not estimated for these areas using the method at hand, which is likely incorrect due to the number of unpaved roads, and general lack of paved ones in these areas. Many are resource roads, also anticipated to be used by the communities at large.

There is not enough information available to estimate the level and speed of traffic on the 1000s of roads in the BVLD. Therefore, it is assumed that these resource and agricultural emissions were captured in the MEI total through the addition of transport trucks to the estimates, based on the number of residents working in the trades (which includes trucking) or agriculture and resources. There are also many unpaved roads within the more populated areas, and these are likely travelled on a daily basis. However, road dust emissions in populated areas were likely over-estimated and were modified such that only 1% of the urban population drove on a dirt road daily (but the model still assumes they travelled 100% of the dirt road length).

However, a new robust and realistic tool should be developed that can adequately classify and generate emissions datasets for roads in the region (and in general), as the methods used here, in Weinstein (2005), and elsewhere, are painfully inadequate. Something like the NPRI's unpaved road dust model perhaps, but with all assumptions and defaults designed for the BVLD or similar locales.

This inventory is not unique in high levels of error and uncertainty; rather, these are the traits emissions inventories are best known for. Errors of 100% or more for some variables are not uncommon, particularly once you begin to run models with an already erroneous EI. Much of this error stems from generalisations, for example: all households do this, or all trucks weigh this much. There may be less error through the addition of truck detail for instance, but each detail has its own

variables that too have their own error. As such, the simplest EF may at times be more accurate than the most complex of models.

4.3 Inventory Usage and Updating

As mentioned in the beginning, inventory development and improvement are potentially limitless. The addition of new sources, finer spatial and temporal detail, and more accurate methods of quantifying the tonnes released, could be infinite. Nonetheless, more detail doesn't necessarily mean more accuracy; and no emissions inventory will ever be 'perfect'.

The request to update this inventory was simple, 20 years had passed and things change. But, if communities and their respective decision-makers remain informed and up-to-date on emissions in the region, policy and incentives can be developed to instigate the positive developments that improve air quality, while suppress those actions that lead to further air quality decline. All the while these policies will balance economic and social requirements, with the health and environmental impacts of particle pollution.

The appendices of this inventory have been provided in digital format, as have the mapping files. The hope is that they are easy enough to understand that the MEI can become somewhat of a living entity. If an industry picks up and leaves town, they can be removed to see the impact, or their legacy may keep on giving for years to come. If a new development that contributes PM is being proposed or approved in the region, they can be added to the MEI, and their potential airshed impacts explored—even prior to approval. Adjustments and refinements can me made to temporal profiles and spatial allocation as new information arises; and if so inclined, the data should be fit for regional dispersion and/or forecast modeling—with some tweaks to the formats dependent on which model is used.

The most important determinator in MEI update focus is the end use. If used for modelling, spatial and temporal allocation are important; if policy-making, then a break-down of sources into further subtypes may help point out where efforts are the most needed. Decisions regarding where to place new air monitors, could maybe use some spatial refinement and even a model run to fully support placement decisions.

One aspect of inventory refinement that doesn't seem to receive due attention is the chemical speciation of PM—something we will likely see more of in years to come. As well the keeping track of ultrafine particles, condensables and secondary PM that can be physically and chemically complex. From a human and environmental health perspective there is a big difference between the mechanical damage a fine particle of pure carbon may have on the lungs, versus the effects a promethium particle or mercury containing aerosol may have on the whole system.

As mentioned in the previous section, error is an integral part of emission inventories, and unless there are real measured data with which to compare your estimates, then that error is not really quantifiable. For instance we saw some example of how emissions change from one year to the next, casting doubt upon the usage of a single year of data. Maybe every source / source type

should have the most up-to-date information available used for the MEi, even if that puts the data for some sources years apart.

There are a number of source types that were not included in the MEI, largely because 1) there were insufficient data available to produce, justify or defend an emissions estimate; 2) there simply was not enough time to complete an assessment of each source within reasonable accuracy; or (3) a combination of the above. Most of these missing sources are listed in section 1.3, and even though some are of natural origin and therefore cannot be controlled or regulated, they do have an impact on local PM levels that should not be completely excluded when assessing local monitoring data or the state of air in the BVLD as a whole. For instance a spike in measured PM could indicate that one of the region's many volcanoes is awakening.

Releases of PM not covered by the MEI include natural and anthropogenic sources such as:

- Wildfires
- Sea salt
- Volcanism
- Long-range transport of foreign dust (e.g. from Asia)
- Vegetation (pollen, condensable VOC)
- Land-slides (dust)

Natural sources are generally not included in EI, which tend to be reserved for anthropogenic sources that we can regulate and control—should we chose to do so. However, we cannot ignore the physical, chemical and biological interactions between human-produced emissions and those produced by natural processes. If not be accounted for it is possible that poor air quality may persist without remedy.

From a human-induced PM stand-point:

- Automobile exhaust
- Automobile brake and tire wear
- Household / commercial cooking and cleaning (bakeries, meat smoking, etc.)
- Backyard burning
- Camp fires and generators in parks or backyards
- Airplane exhaust
- Recreational vehicles and pleasure-craft
- Fuel-based tools and equipment (chainsaws, snow-blowers, etc.)
- Agricultural emissions (tillage dust, burn smoke, sprays, equipment exhaust, aerosol promoting ammonia, etc.)
- Use of aerosol sprays
- Smoke from the use of tobacco, smudge, incense, etc.

Ideally these sources and more should be included. However, it is important not to add a source just for the sake of adding it. Add a source when the information available warrants its addition. The goal should be not to add further error unnecessarily, but to assemble a more complete and robust representation of the BVLD airshed..

4.4 Future Research, Emerging Technologies and Moving Towards a Well-Managed Airshed

It seems ridiculous to burn your garden matter so that your soil nutrients are transported away, and then apply chemical fertilizer to your lawn and garden because system is now in deficit. However, Smithers (BVLD-AMS, 2006) is the only community to date that has outright banned the open burning of yard waste or household refuse within municipal limits. Although backyard burning was not included here, and represented only a small portion of BVLD airshed emissions in Weinstein 2005, this source is considered important at the community/neighbourhood scale.

Local green bin pick-up and or composting facilities are promoted as an alternative to the backyard burning of yard waste, but an outright ban on household refuse burning seems in order, and not too difficult to draft/instate. The smoke from burning garbage may seem to be no different from grass- or leaf-fire smoke, but it is. There are 1000s of compounds already present in household waste, add to that heat and flame oxidation and 1000s more compounds will be formed, some of which we are likely still to identify.

Some of the products of household waste combustion include dioxin and furans, a class of compounds (chlorinated hydrocarbons) that have a specific chemical structure. These compounds are toxic; known carcinogens, neurotoxins and endocrine disruptors; and are extremely persistent in the environment, largely owed to being fat-, rather than water-soluble. There are many different substances in this class, that vary in their degree of toxicity, and are formed when organic matter is combusted with other matter containing chlorine (e.g. many of the plastics we use, bleached paper, salty wood, etc.). They exist in the atmosphere as solids and gases or adhered to the surfaces of mineral particles. They will still be around long after the fire is over—just one example from the slew of things formed from household waste combustion. That being said, there is a BC Government webpage that claims the burning of garbage or construction debris "is illegal unless specifically authorised"[17]; maybe this law just needs to be advertised or enforced.

Road dust can be somewhat controlled with frequent wetting, and the near 225 days a year that the region has precipitation or below freezing temperatures certainly assists with the natural mitigation of road dust.[18] However, part of the issue is dust left from salt and other de-icers that remains in the spring after snow melt. As shown by this inventory unpaved roads are another source. Thankfully road dust is composed of coarser particles and doesn't travel too far; and despite being the largest contributor to TPM in this and the year 2005 inventory (Weinstein 2005), it likely isn't the largest culprit in terms of human health risk. Not only are the particles too large to transport, they are also too large to travel very far into human airways. Overall road dust of all kinds \ seems to be overestimated by the various tools available to assist with PM estimation, and it may be time to move beyond road dust to the particles we don't see, but can wreak havoc.

One such group of particles contains hazardous air pollutants (HAP) known as POM (polycyclic organic matter). This group includes potential carcinogens such as benzo(a)pyrene (BaP), the

[17] https://www2.gov.bc.ca/gov/content/environment/air-land-water/air/air-pollution/smoke-burning/burning-outdoors
[18] see section 3.3 for results from the model used for unpaved road dust that—based on 224.6 days of >0.2 mm of precipitation, snow cover, or frozen conditions—predicted a nautral mitigation rate of 62%

formation of which is due to free radical formation in the flame region during incomplete combustion. Combustion efficiency improves as burn rate and flame intensity increase. Non-catalytic fireplace inserts reduce emissions by directing unburned hydrocarbons and CO into an insulated secondary chamber, where they mix with fresh preheated air to enhance, and complete, the combustion process (US-EPA 2017).

Incomplete combustion is what makes PM from wood-smoke (and other heating fuels such as NG and propane) the most harmful. Combustion efficiency improves as burn rate and flame intensity increase; and factors such as moisture content and oxygen levels can at least partially control combustion efficiency. But despite having larger particles than other heat sources (e.g. natural gas; see Appendices A.4and A.4,), the US-EPA (2017) recount that Rau and Huntzicker (1984) found 95 percent of all particles emitted from a wood stove to be less than 0.4 micrometers in size. This places PM from woodsmoke in the "ultrafine" category, also increasing the likelihood and diversity of potential health impacts linked to exposure.

However, inefficient wood burning appliances, industrial boilers, and open burning can all create a myriad of harmful particles including tiny sub-micron carbon spheres, and POM or other hazardous hydrocarbons that adhere to the surface of otherwise chemically benign carbonaceous smoke particles or become part of the particle physical matrix. Upgrading and replacing inefficient wood stoves is a step in the right direction, hopefully the provincial program will continue and expand in coming years. Other practices such as using dry, properly aged firewood; and educating the public to do so, are also useful tools in reducing PM emissions in the BVLD (Figure 16).

Figure 16. An example of public education to help reduce PM emissions from woodsmoke in an add of the BVLD AMS (from : Pinna Sustainability Inc. 2015).

Problems with PM emissions in the BVLD led to some local and provincial initiatives to both educate the public and reduce emissions in the airshed and have been quite successful. One of the simplest and most effective ways to reduce harmful emissions from externally fired combustion engines may be education of the operator. This could come in a variety of forms, depending on the specific target audience. Inefficient combustion not only produces more toxic emissions, it also burns more fuel—meaning it costs more money. Everyone wants to save money, no matter how they feel

about the health of their neighbours or the environment. Making people aware of cost savings could lead to the added benefit of cleaner air. Fuel producers also play a role, for instance businesses that deliver firewood by the cord. There isn't really any positive reinforcement mechanism for them to have customers need less of their product; but there are potential incentives, or penalties, that may reduce practices such as the common "of course this wood is seasoned for over a year" claim. Similarly, boilers using heavier (higher grade) fuel oils combust more efficiently at lower loads (such as 50% of their rating) (US-EPA, 2007); however, the precise load at which enhanced efficiency makes up for the load reduction is unclear.

The science of air pollution monitoring is constantly changing. Electrochemical sensors such as the Purple Air Monitors, that are being used by AMS throughout the BVLD in collaboration with the University of Northern British Columbia (UNBC), are showing real promise as an affordable and reliable alternative to traditional air quality monitors that cost tens of thousands of dollars. In the past decade or so, electrochemical sensor technology has improved greatly, particularly in terms of measurement sensitivity. Such sensors used to only find purpose in occupational and emergency settings as they could only register concentrations (of gases) in ppm, despite AQO being in ppb (i.e. the levels at which human and environmental health effects are known to occur). Increased sensitivity can also have its challenges. For example electrochemical PM sensors have a habit of registering those tiny glittery ice crystals that form under the right conditions of temperature and relative humidity leading to false peaks and episodes. These sorts of measurement artefacts may need to accounted for and/or teased from data sets with a standard algorithm that accounts for the conditions under which such conditions occur en masse.

We are also always learning more about pollutants themselves—their chemical make-up, the size of particles, etc. For instance a decade ago no one really spoke much of PM1, or superfine sub-micron nanoparticles—our filters that were once the sole way of measuring PM were too coarse to catch them (let alone 'condensables'). But as our technology and understanding of air quality both improve, air quality itself should also improve. This relies on dedication and the knowledge afforded by inventoried emissions in concert with ambient monitoring.

5 Conclusion

To go back to the fourth goal of the updated MEI: to determine "the best" inventory that can be developed for the BVLD based on existing data and what we know; and that wasn't necessarily accomplished here. The best MEI is likely one that keeps evolving as new information and data sources arise; one that can be modified or updated to be used for different purposes. But one must be weary of the infinite nature of something that could become so detailed you get stuck drowning in those details. At the same time, there are so many variables from source characteristics, to size fractions, to weather conditions, things that affect the completeness of combustion, the chemical species of the PM, whether it's organic or inorganic, is it chemically benign or coated/impregnated with a compound that can lead to cancer or inheritable genetic mutations? Ideally the best inventory could deal with all those variables, answer all those questions, and more. But, this perfect MEI doesn't exit for PM in the BVLD, or for any pollutant type, anywhere. Instead the best inventory finds a balance between the details (e.g. household) and the big picture (e.g. airshed) and is used to make decisions that protect air quality from degradation. How the MEI is used may

be the most important part; and ensuring that respective communities and residents realise, particulates matter.

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