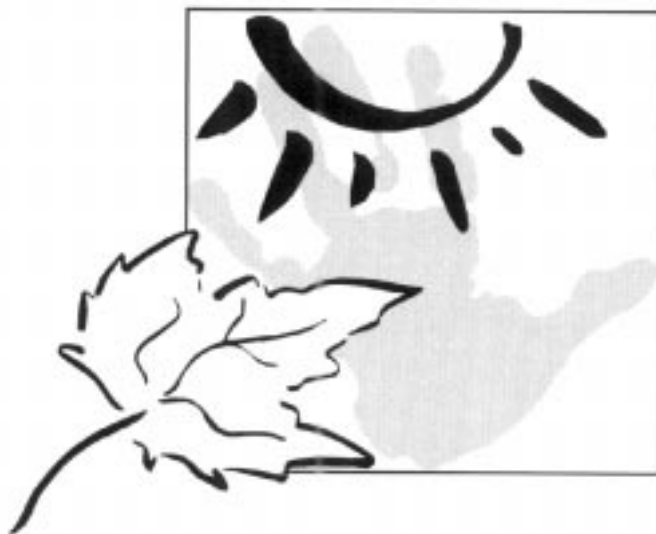


NATIONAL AMBIENT AIR QUALITY OBJECTIVES FOR PARTICULATE MATTER



ADDENDUM TO THE SCIENCE ASSESSMENT DOCUMENT

A report by the
Federal - Provincial Working Group
on Air Quality Objectives
and Guidelines

December 1997, Revised April 1999

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APPENDIX A INCREMENTAL RISK ANALYSIS

Positive associations have been observed between particulate matter and both daily mortality and hospital admissions for cardiorespiratory disease with no evidence of a threshold level, i.e., any increase in ambient particulate matter is associated with an increase in mortality and hospitalization rates. In the PM Science Assessment Document a variety of studies examining associations between hospital admissions and mortality were used as the basis for deriving both the PM_{2.5} and PM₁₀ lowest observed adverse effect levels (LOAELs). These statistically derived LOAELs are levels above which the incidence of health effects in the population attributable to PM can be quantified and the uncertainties estimated.

The first part of the incremental risk analysis estimates daily health impacts at current ambient levels. This is equivalent to assessing the potential change in health impacts associated with a reduction in ambient PM levels from current concentrations to the LOAELs. In the second part, estimates of the potential change in mortality and hospital admissions (the number of avoided impacts) associated with reductions in ambient PM levels from current concentrations to a range of PM concentrations which if achieved would substantially reduce the risk to human health.

A.1 METHODS

Most of the epidemiology studies provide an estimate of the relative risk associated with a specific change in PM concentration, Δx (e.g., a 50 $\mu\text{g}/\text{m}^3$ increase in PM₁₀ or a 25 $\mu\text{g}/\text{m}^3$ increase in PM_{2.5}). Given the concentration-response function (reflected by β , the coefficient derived from the epidemiology studies) and a change in ambient PM levels (Δx), the relative risk (denoted as $RR_{\Delta x}$) associated with that change in ambient PM is equal to $e^{\beta \Delta x}$. If the concentration-response relationships for both PM₁₀ and PM_{2.5} are linear down to the LOAEL, the relative risk is recalculated to reflect the association between a one $\mu\text{g}/\text{m}^3$ change in ambient PM levels and the resulting adverse health impacts ($RR_1 = e^{\beta}$, where $\Delta x = 1$).

Estimates of the number of deaths and hospitalizations, y , for a standard population of one million people, is directly related to the number of one $\mu\text{g}/\text{m}^3$ increments above the LOAEL. Depicting the ambient data as a cumulative concentration of all one $\mu\text{g}/\text{m}^3$ increments above the LOAEL in a year, conceptually a SUMLOAEL ($\mu\text{g}/\text{m}^3 \times \text{days}$), is similar to the notion of the accumulated exposure over a threshold of 40 ppb (AOT40, developed by the Europeans) to protect vegetation from ozone exposure. For example, a 24-hour ambient PM₁₀ concentration of 30 $\mu\text{g}/\text{m}^3$ is equivalent to five 1 $\mu\text{g}/\text{m}^3$ increments above the LOAEL (30 minus 25 = 5).

Current PM related health impacts are equal to the risk per $\mu\text{g}/\text{m}^3$, multiplied by the daily baseline impact rate, β , multiplied by the cumulative concentration (SUM):

$$y = \beta (RR_1 - 1) \text{SUM}$$

The potential number of avoided impacts attributable to reductions in ambient PM concentrations is directly related to the decrease in the number of 1 $\mu\text{g}/\text{m}^3$ above the LOAEL.

Summing all concentrations above the LOAELs does not imply adverse health effects are not associated with lower concentrations. The epidemiological evidence supports the conclusion that adverse health effects occur after exposure to ambient PM concentrations less than 24-hour concentrations of 25 $\mu\text{g}/\text{m}^3$ for PM_{10} and 15 $\mu\text{g}/\text{m}^3$ for $\text{PM}_{2.5}$. The data reveal no evidence of a threshold, therefore, impacts could be associated with ambient PM concentrations below background. Counting impacts beginning at concentrations lower than the LOAELs is not done, however, because the available data is limited in allowing understanding of the form of the relationship below this level, and there is no clear definition or acceptance of 24-hour background concentrations.

A.1.1 Relative Risks Per $\mu\text{g}/\text{m}^3$ PM

The risk analysis focuses on two health effects reported in the epidemiology literature: increased daily mortality associated with both PM_{10} and $\text{PM}_{2.5}$ and increased hospital admissions for cardio-respiratory causes associated with $\text{PM}_{2.5}$. Although the analysis does not address all of the various human health endpoints, all potential effects are characterized in Chapter 3.2 of the Rationale Document and Chapter 12 of the Science Assessment Document. Table A1 lists the selected relative risks (mean, 95 % CI) for each of the health effects for both PM_{10} and $\text{PM}_{2.5}$.

For this analysis, estimates of the change in mortality associated with PM_{10} utilized the relative risks from ten short-term exposure mortality studies (Ito & Thurston, 1996; Kinney et al., 1995; Pope et al., 1992; Schwartz, 1993; and Schwartz et al, 1996; each reviewed in the Science Assessment Document). The relative risks from each study were combined to produce an “all-averaging times” pooled estimate of relative risk using a random effects model. The methodology for deriving the pooled relative risk estimate is described elsewhere (Abt Associates, 1996). The pooled estimate is superior to a single relative risk estimate since it combines studies that used the average PM concentration on a single day as well as studies that used the average PM concentration over a 2- 3- or 5-day period.

Estimates of the change in mortality associated with $\text{PM}_{2.5}$ are based upon a combined estimate of the concentration-response coefficients reported in the re-analysis of the Six Cities Study (Schwartz et al., 1996; cited in Science Assessment Document). Both combined estimates (PM_{10} and $\text{PM}_{2.5}$) mitigate the concern about the transferability of the concentration-response relationships from the city in which they were estimated to other places and times.

For the analysis of cardiac and respiratory hospitalizations, the $\text{PM}_{2.5}$ equivalent of the statistical association between SO_4 and hospital admissions developed by Burnett et al. 1995 (cited in the Science Assessment Document) are used (Table A1). The concentration-response relationships are combined estimates of the relationship between sulphates and hospitalization in many locations and populations in Southern Ontario. These results are used because they are based on Canadian data, and the effects were observed in both the summer and winter quarters, in both males and females and across all age groups.

Table A1 Relative Risk Estimates (95% Confidence Interval)		
		RR per $\mu\text{g}/\text{m}^3$
Relative Risk of mortality per 50 $\mu\text{g}/\text{m}^3$ change in daily PM_{10} concentration, all averaging times	1.04 (1.053, 1.025)	1.00078 (1.001, 1.0005)
Relative Risk of mortality, per 25 $\mu\text{g}/\text{m}^3$ change in daily $\text{PM}_{2.5}$ concentrations	1.036 (1.047, 1.026)	1.0014 (1.0018, 1.001)
Relative Risk of respiratory hospital admissions (RHA) per 10 $\mu\text{g}/\text{m}^3$ change in daily $\text{PM}_{2.5}$ concentrations	1.0074 (1.0099, 1.0049)	1.00074 (1.00099, 1.00049)
Relative Risk of cardiac hospital admissions (CHA) per 10 $\mu\text{g}/\text{m}^3$ change in daily $\text{PM}_{2.5}$ concentrations	1.0070 (1.0102, 1.0036)	1.0007 (1.001, 1.00036)

A.1.2 Baseline Health Effects Incidence Rates

Baseline health effects incidence rates for mortality (18.4/1,000,000 nonaccidental deaths per day) and hospital admissions for respiratory causes (16 RHA per million per day) or cardiac causes (14.4 daily CHA per million) were obtained from Supplemental Report 2, Environmental Health Benefits of Cleaner Vehicles and Fuels¹. Location-specific baseline incidence information for hospital admissions and mortality was not used because it is not as readily available as national data.

A.2 HEALTH IMPACTS ATTRIBUTABLE to CURRENT AMBIENT PM CONCENTRATIONS

The current ambient data set consists of both PM_{10} and $\text{PM}_{2.5}$ dichotomous sampler measurements from 18 NAPS sites (16 cities) for the period January 1992 through December 1994. The data set was normalized to 365 days per year to account for the 1 in 6 days sampling schedule and missing measurements (See Appendix B for details). For each site, the number of days/year are sorted into 5 $\mu\text{g}/\text{m}^3$ bins. The sum of all daily 1 $\mu\text{g}/\text{m}^3$ increments above the LOAEL are summed to produce a site-specific annual cumulative concentration, $\mu\text{g}/\text{m}^3 \times \text{days}$, shown in Table A2 for both PM_{10} and $\text{PM}_{2.5}$.

Walpole Island has the largest cumulative concentration for both PM_{10} and $\text{PM}_{2.5}$, 5536 $\mu\text{g}/\text{m}^3 \times \text{days}$ and 3474 $\mu\text{g}/\text{m}^3 \times \text{days}$, respectively. There were approximately 200 days/year (Table B2) when the 24-hour PM_{10} concentration was greater than or equal to 25 $\mu\text{g}/\text{m}^3$. A similar number of days (Table B3) exceeded the LOAEL for $\text{PM}_{2.5}$. The smallest cumulative concentration, 72 $\mu\text{g}/\text{m}^3 \times \text{days}$, occurs in Sutton, with approximately 15 days/year (Table B2) when the 24-hour PM_{10} concentration was greater than or equal to 25 $\mu\text{g}/\text{m}^3$. Sutton's $\text{PM}_{2.5}$ concentrations were equal to or greater than 15 $\mu\text{g}/\text{m}^3$ for approximately 40 days/year.

¹Chestnut, L. & Ostro, B. (1995), Environmental and Health Benefits of Cleaner Vehicles and Fuels (Supplemental Report 2).

Table A2 Current Ambient PM Cumulative Concentrations			
Station	City	Cumulative Concentration ($\mu\text{g}/\text{m}^3 \times \text{day}$)	
		PM ₁₀	PM _{2.5}
30118	Halifax	192	303
30501	Kejimkujic	309	375
40203	Saint John	462	309
50104	Montreal	1446	946
50109	Montreal	3826	1661
54101	Sutton	72	179
60104	Ottawa	1032	709
60204	Windsor	2148	1350
60211	Windsor	3395	2006
60424	Toronto	2545	1728
60512	Hamilton	3515	2442
61901	Walpole Is.	5536	3474
64401	Egbert	819	714
70119	Winnipeg	1929	396
90130	Edmonton	1403	321
90227	Calgary	1370	214
100111	Vancouver	738	662
100303	Victoria	229	291

Each $\mu\text{g}/\text{m}^3$ increment is associated with an incidence of adverse health effect, therefore, more increments (a greater cumulative concentration) result in more human health impacts. Every 1 $\mu\text{g}/\text{m}^3$ increase in PM₁₀ concentration results in an increase of 0.014 deaths/10⁶ population/day, calculated as the $(RR_1 - 1)$ multiplied by the number of nonaccidental deaths per day $[(1.00078 - 1) \times 18.4/1,000,000]$. Table A3 shows the average number of deaths per million population per year associated with current PM₁₀ concentrations.

Similarly, the number of PM_{2.5} associated deaths and hospitalizations for cardiorespiratory causes per $\mu\text{g}/\text{m}^3$ per million population per day (also shown in Table A3) equals the $(RR_1 - 1)$ multiplied by the baseline number of daily hospitalizations. Each $\mu\text{g}/\text{m}^3$ increase in PM_{2.5} will result in an increase of 0.026 deaths/10⁶ population/day, 0.0118 hospitalizations for respiratory related causes/10⁶ population/day, and 0.010 hospitalizations for cardiac related causes/10⁶ population/day.

For example, in Calgary, Alberta, the cumulative concentration (the sum of all one $\mu\text{g}/\text{m}^3$ increments above the LOAEL) for the normalized year equals 1370 $\mu\text{g}/\text{m}^3 \times \text{days}$ for PM₁₀ and 214 $\mu\text{g}/\text{m}^3 \times \text{days}$ for PM_{2.5} (Table A2). Multiplying the cumulative concentrations by the

respective incidence of adverse health impacts per million population per day results in the estimate of annual impacts. Therefore, $1370 \mu\text{g}/\text{m}^3 \times \text{days}$ multiplied by $0.014 \text{ deaths}/10^6 \text{ population}/\text{day}$ equals an estimated 19.8 deaths per million population per year resulting from current ambient PM_{10} concentrations.

Table A3 Health Impacts Resulting from Current Ambient PM Levels					
Station	City	PM₁₀ Related Impacts	PM_{2.5} Related Impacts		
		Annual Mortality	Annual Mortality	Annual RHA ¹	Annual CHA ²
		per million population			
30118	Halifax	2.8	7.9	3.6	3.0
30501	Kejimkujic	4.5	9.8	4.4	3.8
40203	Saint John	6.7	8.1	3.6	3.1
50104	Montreal	20.9	24.7	11.2	9.5
50109	Montreal	55.2	43.3	19.6	16.7
54101	Sutton	1.0	4.7	2.1	1.8
60104	Ottawa	14.9	18.5	8.4	7.1
60204	Windsor	31.0	35.2	15.9	13.6
60211	Windsor	49.0	52.3	23.7	20.2
60424	Toronto	36.7	45.0	20.4	17.4
60512	Hamilton	50.8	63.6	28.8	24.5
61901	Walpole Is.	79.9	90.5	41.0	34.9
64401	Egbert	11.8	18.6	8.4	7.2
70119	Winnipeg	27.9	10.3	4.7	4.0
90130	Edmonton	20.3	8.4	3.8	3.2
90227	Calgary	19.8	5.6	2.5	2.1
100111	Vancouver	10.7	17.2	7.8	6.6
100303	Victoria	3.3	7.6	3.4	2.9

¹ RHA = respiratory hospital admissions

² CHA = cardiac hospital admissions

Estimates of the number of deaths and hospitalizations in Calgary resulting from current ambient $\text{PM}_{2.5}$ concentrations are calculated as follows:

$214 \mu\text{g}/\text{m}^3 \times \text{days}$ (the cumulative concentration) multiplied by $0.026 \text{ deaths}/10^6 \text{ population}/\text{day} = 5.6 \text{ deaths}/10^6 \text{ population}/\text{year}$.

Similarly, $214 \mu\text{g}/\text{m}^3 \times \text{days}$ multiplied by $0.0118 \text{ hospitalizations for respiratory related causes}/10^6 \text{ population}/\text{day} = 2.5 \text{ hospitalizations for respiratory related causes}/10^6 \text{ population}/\text{year}$, and

214 $\mu\text{g}/\text{m}^3 \times \text{days}$ multiplied by 0.010 hospitalizations for cardiac related causes/ 10^6 population/day = 2.1 hospitalizations for cardiac related causes/ 10^6 population/year.

A.3 POTENTIAL CHANGES IN HEALTH IMPACTS WITH ADJUSTED AMBIENT DATA

The potential number of avoided impacts corresponding to a reduction in 24-hour ambient PM concentrations can be calculated by adjusting the ambient PM distribution to simulate no exceedances² of the range of incremental target levels. The incremental targets are concentration limits (identified in Sections A.3.1 for PM_{10} , and A.3.2 for $\text{PM}_{2.5}$) that encompass the range identified in Chapter 4.

Adjusting current ambient concentrations involves decreasing PM levels on all days by the same percentage (a proportional linear rollback). The rollback itself involves calculating, at each site, the percent reduction - which is the ratio between the specified concentration limit and the observed maximum concentration from the normalized data set at a given site. Appendix C provides a comparison of the observed maximum concentration, from the 1-in-6 day sampling regime, with the annual maximum concentration from a daily sampling schedule. The analysis reveals that the observed maximum approximates the 98th percentile of the ambient frequency distribution. For the purpose of this analysis, therefore, the maximum concentrations are considered to be the 98th percentile of the ambient concentration distribution.

All daily values are subsequently adjusted (rolled back) by the same percentage - by multiplying the 24-hour concentration by (1 - percent reduction). Appendix D compares the proportional linear rollback with alternative approaches to reducing ambient concentrations below current levels. The analysis concludes that the proportional rollback methodology is the most suitable method for simulating attainment of targets given the limited source-receptor modelling and understanding of how emissions reductions would actually translate into reductions in ambient PM concentrations. An analysis by the US EPA indicates that PM levels have historically decreased in a proportional manner, i.e., concentrations at different points in the distribution have decreased by approximately the same percentage. Note that it is not possible, based on our current understanding, to estimate the corresponding changes in emissions, by type and location, that would correspond to the percent reductions in ambient air concentrations under the incremental rollback scenarios.

² An exceedance is defined here as any site experiencing a daily average concentration greater than the specified level one or more times per year.

A.3.1 Avoided Impacts Resulting from Proportional Rollback of Ambient PM10 Data

The 24-hour maximum concentration limits for PM₁₀ range from 50 µg/m³ to 25 µg/m³, decreasing in 5 µg/m³ steps. The upper limit arbitrarily reflects the existing PM10 air quality criterion adopted by British Columbia, Ontario, and Newfoundland. The lower increment is the lowest level, identified in the Science Assessment Document, at which there is a statistically significant association between ambient concentrations of PM10 and adverse health effects. The lower limit does not reflect the existence of a threshold; the epidemiological evidence supports the conclusion that PM has adverse health effects at concentrations below the LOAEL (24-hour 25 µg/m³). Its utilization simply reflects the need for a scientifically robust limit without entering into the discussion of thresholds and background concentrations.

For illustrative purposes, the normalized PM₁₀ distribution profile for Edmonton, AB is presented in Table A4 (row 3) along with the percent reduction calculations (the final column) and the adjusted ambient concentration distribution profile. The maximum value reported in the table is the observed maximum concentration measured in the three-year period. With each proportional linear rollback, the observed 24-hour maximum concentration (77.6 µg/m³), and the lognormal frequency distributions are reduced by the calculated percentage such that the new 24-hour maximum concentration equals one of 50, 45, 40, 35, 30, or 25 µg/m³. Correspondingly, the number of days/year in each of the bins (<15, 15-20, 20-25, etc.) decreases with each rollback until the 24-hour maximum concentration equals the LOAEL of 25 µg/m³.

	N	< 15	15-20	20-25	25-30	30-35	35-40	40-45	45-50	>50	Max	% Red
Normalized data set	167	125	72	66	31	31	4	7	13	17	77.6	-
Rollback to 50 µg/m ³	167	238	55	35	9	17	4	4	0	2	50	$\frac{(77.6 - 50)}{77.6} = 36$
Rollback to 45 µg/m ³	167	269	55	9	20	7	2	4	0	0	45	$\frac{(77.6 - 45)}{77.6} = 42$
Rollback to 40 µg/m ³	167	282	46	17	11	4	4	0	0	0	40	$\frac{(77.6 - 40)}{77.6} = 48$
Rollback to 35 µg/m ³	167	317	17	20	7	2	2	0	0	0	35	$\frac{(77.6 - 35)}{77.6} = 55$
Rollback to 30 µg/m ³	167	328	24	7	4	2	0	0	0	0	30	$\frac{(77.6 - 30)}{77.6} = 61$
Rollback to 25 µg/m ³	167	337	22	4	2	0	0	0	0	0	25	$\frac{(77.6 - 25)}{77.6} = 68$

For the normalized data set in Edmonton, a decrease in PM10 levels of 42% would be needed so the 24-hour peak concentration just equals 45 µg/m³. If the reduced concentration limit is 40 or 35 µg/m³, peak concentrations would have to be reduced by 48% and 55%, respectively. Similar rollback calculations were done for each of the 16 cities. The adjusted frequency distributions are presented in Table A5. The estimated percent reductions are in Appendix D.

Using the adjusted ambient data set, annual cumulative concentrations (the sum of all daily PM10 concentrations greater than LOAEL) are summed under each rollback scenario (Table A6). The number of avoided impacts, Δy, associated with a reduction in ambient PM levels such that the 24-hour maximum meets the proposed targets is calculated as follows:

$\Delta y = y (RR_1 - 1) \Delta SUM$, where ΔSUM is cumulative concentration greater than 25 µg/m³ for the current year minus the cumulative concentration following the rollback. The resulting cumulative concentration difference (ΔSUM) is multiplied by the relative risk associated with an increase in death for each 1 µg/m³ increase in PM10 concentrations, and the baseline health effects incidence rates. However, the number of avoided impacts are underestimates of the potential benefits of reducing ambient PM10 concentrations. Applying a proportional linear rollback to all daily concentrations will result in some concentrations being reduced below the LOAEL. The calculation of avoided impacts does not include the reductions in ambient concentrations to less than the LOAEL.

Table A5 Number of Days/year when 24-hour PM10 Concentrations Fall Within Specified Concentration Limits											
Rollback to 50 µg/m³											
Station	N	<15	15-20	20-25	25-30	30-35	35-40	40-45	45-50	>50	Max
40203	557	282	42	24	11	5	1	0	0	1	50.0
30118	140	224	76	26	23	13	0	3	0	0	42.4
30501	454	311	27	10	5	4	5	2	0	1	50.0
50104	440	285	42	21	11	4	1	0	1	1	50.0
50109	110	256	60	17	13	3	7	7	3	0	50.0
54101	136	282	38	32	11	0	0	3	0	0	42.1
60104	128	228	68	20	31	6	3	0	9	0	50.0
60204	123	288	39	21	12	0	0	3	0	3	50.0
60211	490	218	74	34	19	10	5	1	3	1	50.0
60424	488	245	59	34	12	7	3	3	1	1	50.0
60512	156	222	63	33	33	2	7	2	0	2	50.0
61901	137	288	29	13	13	11	0	5	3	3	50.0
64401	279	275	47	17	17	5	3	0	1	0	50.0
70119	146	280	63	13	3	3	3	0	0	3	50.0
90130	167	238	55	35	9	17	4	4	0	2	50.0
90227	173	213	72	46	13	11	6	2	0	2	50.0
100111	94	144	70	54	39	31	12	12	4	0	45.8
100303	131	187	86	53	22	11	3	3	0	0	41.3

Rollback to 45 µg/m³											
Station	N	<15	15-20	20-25	25-30	30-35	35-40	40-45	45-50	>50	Max
40203	557	302	36	18	6	3	0	1	0	0	45.0
30118	140	224	76	26	23	13	0	3	0	0	42.4
30501	454	324	18	9	5	4	4	1	0	0	45.0
50104	440	301	41	14	6	1	0	1	1	0	45.0
50109	110	282	46	13	7	7	3	3	3	0	45.0
54101	136	282	38	32	11	0	0	3	0	0	42.1
60104	128	268	46	26	11	6	0	6	3	0	45.0
60204	123	303	33	24	0	0	3	0	3	0	45.0
60211	490	250	60	30	13	8	0	4	1	0	45.0
60424	488	271	48	30	7	4	4	1	1	0	45.0
60512	156	243	63	28	16	7	5	2	0	0	45.0
61901	137	293	35	13	11	3	5	3	3	0	45.0
64401	279	293	42	14	10	3	1	0	1	0	45.0
70119	146	315	33	8	3	5	0	0	3	0	45.0
90130	167	269	55	9	20	7	2	4	0	0	45.0
90227	173	238	72	27	13	11	2	0	2	0	45.0
100111	94	148	82	43	39	31	8	12	4	0	45.0
100303	131	187	86	53	22	11	3	3	0	0	41.3

Rollback to 40 µg/m3											
Station	N	<15	15-20	20-25	25-30	30-35	35-40	40-45	45-50	>50	Max
40203	557	314	33	13	4	0	0	1	0	0	40.0
30118	140	242	63	29	26	3	0	3	0	0	40.0
30501	454	332	16	6	6	5	0	1	0	0	40.0
50104	440	318	30	12	4	0	1	1	0	0	40.0
50109	110	309	23	17	7	0	7	3	0	0	40.0
54101	136	293	38	24	8	0	0	3	0	0	40.0
60104	128	279	37	34	6	0	6	3	0	0	40.0
60204	123	318	30	12	0	3	0	3	0	0	40.0
60211	490	282	44	22	10	2	4	1	0	0	40.0
60424	488	289	49	15	5	4	1	1	0	0	40.0
60512	156	274	44	33	5	7	0	2	0	0	40.0
61901	137	312	19	21	3	5	3	3	0	0	40.0
64401	279	311	27	17	7	1	0	1	0	0	40.0
70119	146	333	23	3	5	0	0	3	0	0	40.0
90130	167	282	46	17	11	4	4	0	0	0	40.0
90227	173	270	61	17	11	4	0	2	0	0	40.0
100111	94	163	85	47	43	12	12	4	0	0	40.0
100303	131	195	89	47	20	8	6	0	0	0	40.0

Rollback to 35 µg/m3											
Station	N	<15	15-20	20-25	25-30	30-35	35-40	40-45	45-50	>50	Max
40203	557	333	24	7	1	0	1	0	0	0	35.0
30118	140	269	55	26	13	0	3	0	0	0	35.0
30501	454	342	10	6	6	1	0	0	0	0	35.0
50104	440	335	22	6	1	1	1	0	0	0	35.0
50109	110	325	17	7	7	7	3	0	0	0	35.0
54101	136	311	38	13	0	0	3	0	0	0	35.0
60104	128	305	37	11	3	9	0	0	0	0	35.0
60204	123	332	27	0	3	0	3	0	0	0	35.0
60211	490	303	40	13	4	4	1	0	0	0	35.0
60424	488	313	38	7	6	1	1	0	0	0	35.0
60512	156	304	33	16	9	0	2	0	0	0	35.0
61901	137	320	21	13	0	8	3	0	0	0	35.0
64401	279	334	20	9	1	1	0	0	0	0	35.0
70119	146	348	10	3	3	0	3	0	0	0	35.0
90130	167	317	17	20	7	2	2	0	0	0	35.0
90227	173	304	36	17	6	0	2	0	0	0	35.0
100111	94	202	70	58	19	16	0	0	0	0	35.0
100303	131	234	84	31	11	3	3	0	0	0	35.0

Rollback to 30 µg/m3											
Station	N	<15	15-20	20-25	25-30	30-35	35-40	40-45	45-50	>50	Max
40203	557	347	14	3	0	1	0	0	0	0	30.0
30118	140	305	39	18	0	3	0	0	0	0	30.0
30501	454	348	8	7	1	1	0	0	0	0	30.0
50104	440	348	15	1	1	1	0	0	0	0	30.0
50109	110	332	17	7	7	3	0	0	0	0	30.0
54101	136	330	30	3	3	0	0	0	0	0	30.0
60104	128	317	34	6	6	3	0	0	0	0	30.0
60204	123	347	12	0	3	3	0	0	0	0	30.0
60211	490	326	26	8	4	1	0	0	0	0	30.0
60424	488	339	16	6	3	1	0	0	0	0	30.0
60512	156	318	33	9	5	0	0	0	0	0	30.0
61901	137	330	21	3	8	3	0	0	0	0	30.0
64401	279	339	21	4	0	1	0	0	0	0	30.0
70119	146	355	3	5	0	3	0	0	0	0	30.0
90130	167	328	24	7	4	2	0	0	0	0	30.0
90227	173	331	19	11	2	2	0	0	0	0	30.0
100111	94	249	62	39	16	0	0	0	0	0	30.0
100303	131	284	59	11	8	3	0	0	0	0	30.0

Rollback to 25 µg/m3											
Station	N	<15	15-20	20-25	25-30	30-35	35-40	40-45	45-50	>50	Max
40203	557	358	6	0	1	0	0	0	0	0	25.0
30118	140	326	34	3	3	0	0	0	0	0	25.0
30501	454	353	9	2	1	0	0	0	0	0	25.0
50104	440	358	5	1	1	0	0	0	0	0	25.0
50109	110	345	10	10	0	0	0	0	0	0	25.0
54101	136	352	11	3	0	0	0	0	0	0	25.0
60104	128	348	9	6	3	0	0	0	0	0	25.0
60204	123	359	0	3	3	0	0	0	0	0	25.0
60211	490	346	15	4	1	0	0	0	0	0	25.0
60424	488	351	10	4	1	0	0	0	0	0	25.0
60512	156	351	9	2	2	0	0	0	0	0	25.0
61901	137	344	11	8	3	0	0	0	0	0	25.0
64401	279	356	8	0	1	0	0	0	0	0	25.0
70119	146	358	5	3	0	0	0	0	0	0	25.0
90130	167	337	22	4	2	0	0	0	0	0	25.0
90227	173	344	17	2	2	0	0	0	0	0	25.0
100111	94	283	62	16	4	0	0	0	0	0	25.0
100303	131	323	28	14	0	0	0	0	0	0	25.0

Table A6: Cumulative Concentrations >25 µg/m³ (µg/m³ × days) for PM10 rollback targets.

City	50 µg/m ³	45 µg/m ³	40 µg/m ³	35 µg/m ³	30 µg/m ³	25 µg/m ³
	Sum	Sum	Sum	Sum	Sum	Sum
Halifax	191.6	191.6	127.2	42	13	0
Kejimkujic	161.7	106.8	60.4	24.7	4.6	0
Saint John	83	44.2	19.1	7.2	3.3	0
Montreal	92.3	56	29.9	16.2	7.4	0
Montreal	326.4	229.7	146.3	76.1	27.3	0
Sutton	71.9	71.9	52.1	26.8	13.4	0
Ottawa	343.2	220.6	143.2	83.5	39.5	0
Windsor	134.7	98.8	71.3	43.9	16.4	0
Windsor	277.4	169.8	95.2	43.4	17	0
Toronto	191.5	120.5	68.1	30.8	8	0
Hamilton	308.2	180.3	102.2	48.7	12.6	0
Walpole	327	216.3	134.9	79.5	30.1	0
Egbert	142.8	77.4	38.2	16.8	6.5	0
Winnipeg	123.1	85.8	55.1	26.3	12.5	0
Edmonton	333.6	209.5	106.9	50.5	16.2	0
Calgary	269.7	167.6	87	35.8	11.5	0
Vancouver	738.3	680.8	367.8	153.8	43	0
Victoria	228.7	228.7	192.7	88.4	25.2	0

For example, the annual cumulative concentration for the current ambient distribution in Edmonton (1403 µg/m³ × days, Table A2) includes the difference between the 24-hour maximum and the LOAEL (77.6 - 25 = 52.6 µg/m³) as part of its sum. Applying a 42% reduction to the 24-hour maximum concentration in Edmonton reduces the observed maximum concentration from 77.6 µg/m³ to 45 µg/m³. The difference (77.6 - 45 = 32.6 µg/m³) would have contributed to the calculation of avoided impacts, i.e. the number of 1 µg/m³ increments above the LOAEL of 25 µg/m³ has been reduced by 32.6. Applying the 42% reduction to a different day in the current ambient distribution, when the 24-hour maximum is a 37 µg/m³, reduces the concentration to 21.5 µg/m³. The current annual cumulative concentration included the difference between the 24-hour maximum and the LOAEL in its sum (37 - 25 = 12 µg/m³). The twelve 1- µg/m³ increments above the LOAEL factor into the calculation of the number of avoided impacts, while the reduction in the 24-hour concentration below 25 µg/m³ (25 - 21.5 = 3.5 µg/m³) do not.

This in no way implies that the health effects are not associated with PM₁₀ concentrations below 25 µg/m³. The concentration-response relationships are robust down to the LOAEL. At concentrations below that, the health impacts could be calculated, although the uncertainties could not be quantified.

The number of deaths that could be avoided if the 24-hour maximum ambient PM₁₀ concentration is reduced from current levels to the target concentrations (50 µg/m³ to 25 µg/m³) are shown in tabular format on the following page (Table A7). The results are displayed graphically in Chapter 5.

Table A7 Number of Avoided Deaths Resulting from Decreased Ambient PM10 Concentrations

Station	City	50 µg/m3			45 µg/m3			40 µg/m3			35 µg/m3			30 µg/m3			25 µg/m3		
		central	upper	lower	central	upper	lower	central	upper	lower	central	upper	lower	central	upper	lower	central	upper	lower
30118	HALIFAX	0.00	0.00	0.00	0.00	0.00	0.00	0.93	1.23	0.61	2.16	2.84	1.41	2.58	3.40	1.69	2.77	3.64	1.81
30501	KEJIMKUJIC	2.13	2.80	1.39	2.92	3.85	1.91	3.59	4.73	2.35	4.11	5.41	2.69	4.40	5.79	2.88	4.46	5.88	2.92
40203	SAINT JOHN	5.48	7.21	3.58	6.04	7.95	3.95	6.40	8.43	4.19	6.57	8.65	4.30	6.63	8.73	4.34	6.68	8.79	4.37
50104	MONTREAL	19.55	25.74	12.79	20.07	26.43	13.13	20.45	26.93	13.38	20.64	27.19	13.51	20.77	27.35	13.59	20.88	27.49	13.66
50109	MONTREAL	50.54	66.55	33.07	51.93	68.39	33.98	53.14	69.98	34.77	54.15	71.31	35.43	54.85	72.24	35.89	55.25	72.76	36.15
54101	SUTTON	0.00	0.00	0.00	0.00	0.00	0.00	0.29	0.38	0.19	0.65	0.86	0.43	0.84	1.11	0.55	1.04	1.37	0.68
60104	OTTAWA	9.94	13.09	6.50	11.71	15.42	7.66	12.83	16.89	8.39	13.69	18.03	8.96	14.32	18.86	9.37	14.89	19.61	9.75
60204	WINDSOR	29.07	38.29	19.02	29.59	38.97	19.36	29.99	39.49	19.62	30.39	40.02	19.88	30.78	40.54	20.14	31.02	40.85	20.30
60211	WINDSOR	45.02	59.29	29.46	46.57	61.33	30.48	47.65	62.75	31.18	48.40	63.74	31.67	48.78	64.24	31.92	49.02	64.56	32.08
60424	TORONTO	33.98	44.74	22.23	35.00	46.09	22.90	35.76	47.09	23.40	36.30	47.80	23.75	36.63	48.23	23.97	36.74	48.38	24.04
60512	HAMILTON	46.30	60.97	30.30	48.15	63.41	31.51	49.28	64.89	32.24	50.05	65.91	32.75	50.57	66.59	33.09	50.75	66.83	33.21
61901	WALPOLE	75.21	99.04	49.21	76.81	101.15	50.26	77.98	102.69	51.03	78.78	103.75	51.55	79.49	104.69	52.02	79.93	105.26	52.30
64401	EGBERT	9.76	12.85	6.39	10.70	14.10	7.00	11.27	14.84	7.37	11.58	15.25	7.58	11.73	15.44	7.67	11.82	15.57	7.74
70119	WINNIPEG	26.08	34.34	17.07	26.62	35.05	17.42	27.06	35.64	17.71	27.48	36.18	17.98	27.68	36.45	18.11	27.86	36.68	18.23
90130	EDMONTON	15.43	20.33	10.10	17.23	22.69	11.27	18.71	24.64	12.24	19.52	25.71	12.77	20.02	26.36	13.10	20.25	26.67	13.25
90227	CALGARY	15.88	20.92	10.39	17.36	22.86	11.36	18.52	24.39	12.12	19.26	25.37	12.60	19.61	25.83	12.83	19.78	26.05	12.94
100111	VANCOUVER	0.00	0.00	0.00	0.83	1.09	0.54	5.35	7.04	3.50	8.44	11.11	5.52	10.04	13.22	6.57	10.66	14.04	6.98
100303	VICTORIA	0.00	0.00	0.00	0.00	0.00	0.00	0.52	0.68	0.34	2.03	2.67	1.33	2.94	3.87	1.92	3.30	4.35	2.16

(1) not adjusted for area population exposed to concentrations measured at ambient monitor

(2) proportional linear rollback to simulate attainment of increment

(3) 365 days based on a 3y data set, Jan 92 to dec 94

(4) baseline annual mortality rate - 6700/1,000,000; WHO, 1994

A.3.2 Avoided Impacts Resulting from Proportional Rollback of Ambient PM2.5 Data

The 24-hour maximum concentration limits for PM_{2.5} range from 30 µg/m³ to 15 µg/m³, also decreasing in 5 µg/m³ steps. The upper limit for the analysis was selected by examining the distribution of PM_{2.5} concentrations across Canada; 30 µg/m³ is greater than the 75th percentile of ambient concentrations in all locations across the country. The lower increment is the lowest level, identified in the Science Assessment Document, at which there is a statistically significant association between ambient concentrations of PM_{2.5} and adverse health effects. The lower limit does not reflect the existence of a threshold. The epidemiological evidence supports the conclusion that PM has adverse health effects at concentrations below the LOAEL (24-hour 15 µg/m³). Its utilization simply reflects the need for a scientifically robust limit without entering into the discussion of thresholds and background concentrations. The adjusted distribution profiles and percent reduction calculations for PM_{2.5} concentrations in Edmonton are presented in Table A8 for illustrative purposes. Table A9 contains the adjusted PM_{2.5} distribution profiles for all 16 cities. The estimated percent reductions are in Appendix D.

	N	<10	10-15	15-20	20-25	25-30	>30	Max	% Red
Normalized data set	169	266	56	24	9	6	4	56.3	-
Rollback to 30 µg/m ³	169	339	22	2	0	2	0	30	$\frac{(56.3 - 30)}{56.3} = 47$
Rollback to 25 µg/m ³	169	352	9	2	0	2	0	25	$\frac{(56.3 - 25)}{56.3} = 56$
Rollback to 20 µg/m ³	169	361	2	0	2	0	0	20	$\frac{(56.3 - 20)}{56.3} = 64$
Rollback to 15 µg/m ³	169	363	0	2	0	0	0	15	$\frac{(56.3 - 15)}{56.3} = 73$

The sum of all daily PM_{2.5} concentrations greater than 15 µg/m³ for the current ambient data for Edmonton is 321 µg/m³xdays (Table A2). A decrease in PM_{2.5} levels of 47% would be needed so the peak level just equals 30 µg/m³. If the target concentration limits were 25 or 20 µg/m³, PM_{2.5} levels in Edmonton would have to be reduced by 56% and 64%, respectively. Reducing the ambient distribution by 47% results in a cumulative concentration equal to 39.7 µg/m³xdays (Table A10). Therefore, reducing the 24-hour maximum from 56.3 µg/m³ to 30 µg/m³ resulted in a decrease in annual cumulative concentration by 282 µg/m³xdays (321 minus 39.7).

Table A9 Number of Days/year when 24-hour PM2.5 Concentrations Fall Within Specified Concentration Limits

Rollback to 30 µg/m3									
Station	City	N	<10	10-15	15-20	20-25	25-30	>30	Max
30118	HALIFAX	140	276	57	21	8	0	3	30.0
30501	KEJIMKUJIC	454	332	16	8	3	6	0	30.0
40203	SAINT JOHN	558	283	55	17	7	2	1	30.0
50104	MONTREAL	440	324	32	3	4	1	1	30.0
50109	MONTREAL	110	295	43	17	7	0	3	30.0
54101	SUTTON	136	284	56	16	5	0	3	30.0
60104	OTTAWA	146	313	35	8	8	3	0	30.0
60204	WINDSOR	130	278	59	22	3	0	3	30.0
60211	WINDSOR	490	312	35	12	5	0	1	30.0
60424	TORONTO	490	276	57	23	7	1	1	30.0
60512	HAMILTON	156	234	75	30	19	5	2	30.0
61901	WALPOLE ISLAND	138	333	19	11	0	3	0	30.0
64401	EGBERT	279	284	47	21	10	1	1	30.0
70119	WINNIPEG	152	355	2	2	2	0	2	30.0
90130	EDMONTON	169	339	22	2	0	2	0	30.0
90227	CALGARY	173	287	55	15	6	2	0	30.0
100111	VANCOUVER	122	266	60	24	9	3	3	30.0
100303	VICTORIA	131	242	56	42	14	11	0	29.7

Rollback to 25 µg/m3									
Station	City	N	<10	10-15	15-20	20-25	25-30	>30	Max
30118	HALIFAX	140	310	42	10	0	3	0	25.0
30501	KEJIMKUJIC	454	342	11	6	5	1	0	25.0
40203	SAINT JOHN	558	318	34	10	3	1	0	25.0
50104	MONTREAL	440	342	16	6	1	1	0	25.0
50109	MONTREAL	110	319	37	7	0	3	0	25.0
54101	SUTTON	136	309	40	13	3	0	0	25.0
60104	OTTAWA	146	323	33	3	5	3	0	25.0
60204	WINDSOR	130	317	39	6	0	3	0	25.0
60211	WINDSOR	490	334	21	10	0	1	0	25.0
60424	TORONTO	490	306	42	13	4	0	0	25.0
60512	HAMILTON	156	271	58	26	7	2	0	25.0
61901	WALPOLE ISLAND	138	346	16	0	0	3	0	25.0
64401	EGBERT	279	314	31	16	4	0	0	25.0
70119	WINNIPEG	152	355	5	2	0	2	0	25.0
90130	EDMONTON	169	352	9	2	0	2	0	25.0
90227	CALGARY	173	316	36	8	2	2	0	25.0
100111	VANCOUVER	122	293	45	21	3	3	0	25.0
100303	VICTORIA	131	265	67	20	14	0	0	25.0

Table A9 Continued Rollback to 20 µg/m3

Station	City	N	<10	10-15	15-20	20-25	25-30	>30	Max
40203	SAINT JOHN	558	338	21	5	1	0	0	20.0
30118	HALIFAX	140	334	26	5	0	0	0	20.0
30501	KEJIMKUJIC	454	348	10	6	1	0	0	20.0
50104	MONTREAL	440	356	6	2	1	0	0	20.0
50109	MONTREAL	110	338	23	0	3	0	0	20.0
54101	SUTTON	136	341	21	0	3	0	0	20.0
60104	OTTAWA	146	348	10	5	3	0	0	20.0
60204	WINDSOR	130	337	25	0	3	0	0	20.0
60211	WINDSOR	490	347	16	1	1	0	0	20.0
60424	TORONTO	490	333	27	4	1	0	0	20.0
60512	HAMILTON	156	309	37	16	2	0	0	20.0
61901	WALPOLE ISLAND	138	352	11	0	3	0	0	20.0
64401	EGBERT	279	331	27	5	1	0	0	20.0
70119	WINNIPEG	152	358	5	0	2	0	0	20.0
90130	EDMONTON	169	361	2	0	2	0	0	20.0
90227	CALGARY	173	342	17	4	2	0	0	20.0
100111	VANCOUVER	122	326	30	6	3	0	0	20.0
100303	VICTORIA	131	295	56	11	3	0	0	20.0

Rollback to 15 µg/m3

Station	City	N	<10	10-15	15-20	20-25	25-30	>30	Max
40203	SAINT JOHN	558	355	9	1	0	0	0	15.0
30118	HALIFAX	140	355	8	3	0	0	0	15.0
30501	KEJIMKUJIC	454	356	9	0	0	0	0	15.0
50104	MONTREAL	440	359	5	1	0	0	0	15.0
50109	MONTREAL	110	355	7	3	0	0	0	15.0
54101	SUTTON	136	357	5	3	0	0	0	15.0
60104	OTTAWA	146	355	10	0	0	0	0	15.0
60204	WINDSOR	130	359	3	3	0	0	0	15.0
60211	WINDSOR	490	359	5	1	0	0	0	15.0
60424	TORONTO	490	356	8	1	0	0	0	15.0
60512	HAMILTON	156	339	23	2	0	0	0	15.0
61901	WALPOLE ISLAND	138	362	3	0	0	0	0	15.0
64401	EGBERT	279	352	12	1	0	0	0	15.0
70119	WINNIPEG	152	360	2	2	0	0	0	15.0
90130	EDMONTON	169	363	0	2	0	0	0	15.0
90227	CALGARY	173	357	6	2	0	0	0	15.0
100111	VANCOUVER	122	350	12	3	0	0	0	15.0
100303	VICTORIA	131	340	22	3	0	0	0	15.0

Table A10 Cumulative Concentrations > 15 µg/m³ (µg/m³ × days) for PM2.5 Rollback Targets				
Cities	Target Concentration Limits			
	30 µg/m ³	25 µg/m ³	20 µg/m ³	15 µg/m ³
Halifax	127	50.4	14	0
Kejimkujic	111.6	59.2	20.6	0
Saint John	120	48.7	11.2	0
Montreal	58.8	29.1	7.4	0
Montreal	108.5	52.9	16.6	0
Sutton	111.5	41.8	13.4	0
Ottawa	114.7	58.6	19.4	0
Windsor	94	38.2	14	0
Windsor	71.9	25.3	4	0
Toronto	122.3	47.6	10.3	0
Hamilton	297.3	136	29.8	0
Walpole	50	26.4	13.2	0
Egbert	149.3	62.9	15.8	0
Winnipeg	50.5	29.2	12	0
Edmonton	39.7	22.3	10.8	0
Calgary	112.7	50	14.4	0
Vancouver	221.8	102	30.1	0
Victoria	290.7	141.8	44.8	0

The number of avoided impacts are calculated by subtracting the adjusted annual cumulative concentration from the current ambient cumulative concentration, and multiplying the difference by the number of health impacts per µg/m³: 0.026 deaths/10⁶ population/day, 0.0118 hospitalizations for respiratory related causes/10⁶ population/day, and 0.010 hospitalizations for cardiac related causes/10⁶ population/day. Results are shown in the Tables A11, A12, and A13, for avoided deaths, avoided respiratory-related hospitalizations and avoided cardiac-related hospitalizations, respectively. In Edmonton, for example, the number of hospital admissions for respiratory related causes could be reduced by an average of three. In Victoria, benefits are only realized once the ambient PM2.5 concentration is reduced to a 24-hour maximum of 25 µg/m³ because the current 24-hour maximum is 29.7 µg/m³.

Table A11 Number of Avoided Deaths Resulting from Decreased Ambient PM2.5 Concentrations													
Stations	Cities	Central estimate				Upper 95% Confidence Interval				Lower 95% Confidence Interval			
		30	25	20	15	30	25	20	15	30	25	20	15
30118	HALIFAX	4.59	6.59	7.54	7.90	5.97	8.56	9.79	10.26	3.33	4.78	5.47	5.73
30501	KEJIMKUJIC	6.85	8.22	9.22	9.76	8.90	10.67	11.98	12.67	4.97	5.96	6.69	7.08
40203	SAINT JOHN	4.92	6.78	7.76	8.05	6.40	8.81	10.08	10.46	3.57	4.92	5.63	5.84
50104	MONTREAL	23.12	23.89	24.46	24.65	30.03	31.04	31.77	32.02	16.78	17.34	17.75	17.89
50109	MONTREAL	40.43	41.88	42.82	43.26	52.52	54.40	55.62	56.19	29.34	30.39	31.07	31.39
54101	SUTTON	1.76	3.58	4.32	4.67	2.29	4.65	5.61	6.06	1.28	2.60	3.13	3.39
60104	OTTAWA	15.49	16.95	17.97	18.48	20.12	22.02	23.35	24.00	11.24	12.30	13.04	13.41
60204	WINDSOR	32.71	34.16	34.79	35.16	42.48	44.37	45.19	45.66	23.73	24.79	25.24	25.51
60211	WINDSOR	50.38	51.60	52.15	52.25	65.44	67.02	67.74	67.87	36.56	37.44	37.84	37.92
60424	TORONTO	41.83	43.77	44.74	45.01	54.33	56.86	58.12	58.47	30.35	31.76	32.47	32.66
60512	HAMILTON	55.86	60.06	62.82	63.60	72.55	78.01	81.60	82.61	40.53	43.58	45.59	46.15
61901	WALPOLE Isle.	89.18	89.80	90.14	90.49	115.84	116.64	117.08	117.53	64.71	65.16	65.41	65.66
64401	EGBERT	14.72	16.97	18.20	18.61	19.12	22.04	23.64	24.17	10.68	12.31	13.20	13.50
70119	WINNIPEG	8.99	9.54	9.99	10.30	11.68	12.40	12.98	13.38	6.52	6.92	7.25	7.48
90130	EDMONTON	7.32	7.77	8.07	8.35	9.51	10.10	10.48	10.85	5.31	5.64	5.86	6.06
90227	CALGARY	2.63	4.26	5.19	5.57	3.41	5.54	6.74	7.23	1.91	3.09	3.77	4.04
100111	VANCOUVER	11.45	14.58	16.45	17.23	14.88	18.93	21.36	22.38	8.31	10.58	11.93	12.50
100303	VICTORIA	0.00	3.88	6.41	7.57	0.00	5.04	8.32	9.83	0.00	2.81	4.65	5.49

Table A12 Number of Avoided Respiratory Hospital Admissions Resulting from Decreased Ambient PM2.5 Concentrations

Stations	Cities	Central Estimate				Upper 95% Confidence Interval				Lower 95th % Confidence Interval			
		30	25	20	15	30	25	20	15	30	25	20	15
30118	HALIFAX	2.08	2.98	3.41	3.58	2.78	3.99	4.56	4.78	1.38	1.98	2.26	2.372739
30501	KEJIMKUJIC	3.10	3.72	4.18	4.42	4.15	4.97	5.58	5.91	2.06	2.47	2.77	2.93022
40203	SAINT JOHN	2.23	3.07	3.52	3.65	2.98	4.11	4.70	4.87	1.48	2.04	2.33	2.417585
50104	MONTREAL	10.47	10.82	11.08	11.17	14.00	14.47	14.81	14.92	6.94	7.18	7.35	7.403017
50109	MONTREAL	18.32	18.97	19.40	19.60	24.48	25.35	25.93	26.19	12.14	12.58	12.86	12.99055
54101	SUTTON	0.80	1.62	1.96	2.11	1.07	2.17	2.61	2.83	0.53	1.07	1.30	1.401471
60104	OTTAWA	7.02	7.68	8.14	8.37	9.38	10.26	10.88	11.19	4.65	5.09	5.40	5.54954
60204	WINDSOR	14.82	15.48	15.76	15.93	19.80	20.68	21.06	21.28	9.82	10.26	10.45	10.5577
60211	WINDSOR	22.82	23.37	23.63	23.67	30.50	31.24	31.57	31.63	15.13	15.49	15.66	15.69272
60424	TORONTO	18.95	19.83	20.27	20.39	25.32	26.50	27.09	27.25	12.56	13.15	13.44	13.51785
60512	HAMILTON	25.30	27.21	28.46	28.81	33.82	36.36	38.03	38.50	16.77	18.04	18.87	19.10013
61901	WALPOLE Isle.	40.40	40.68	40.84	40.99	53.99	54.36	54.57	54.78	26.78	26.97	27.07	27.17407
64401	EGBERT	6.67	7.69	8.24	8.43	8.91	10.27	11.02	11.27	4.42	5.10	5.46	5.588218
70119	WINNIPEG	4.07	4.32	4.53	4.67	5.44	5.78	6.05	6.24	2.70	2.87	3.00	3.094471
90130	EDMONTON	3.32	3.52	3.66	3.78	4.43	4.71	4.89	5.06	2.20	2.33	2.42	2.508486
90227	CALGARY	1.19	1.93	2.35	2.52	1.59	2.58	3.14	3.37	0.79	1.28	1.56	1.671267
100111	VANCOUVER	5.19	6.60	7.45	7.81	6.93	8.82	9.96	10.43	3.44	4.38	4.94	5.175084
100303	VICTORIA	0.00	1.76	2.90	3.43	0.00	2.35	3.88	4.58	0.00	1.16	1.92	2.273726

Table A13 Number of Avoided Cardiac Hospital Admissions Resulting from Decreased Ambient PM2.5 Concentrations

Stations	Cities	central estimate (arithmetic average)				upper 95% confidence interval				Lower 95% confidence interval			
		30	25	20	15	30	25	20	15	30	25	20	15
30118	HALIFAX	1.77	2.54	2.91	3.05	2.58	3.70	4.23	4.43	0.91	1.31	1.50	1.57
30501	KEJIMKUJIC	2.64	3.17	3.56	3.76	3.85	4.61	5.18	5.48	1.36	1.63	1.83	1.94
40203	SAINT JOHN	1.90	2.62	2.99	3.11	2.76	3.81	4.36	4.52	0.98	1.35	1.54	1.60
50104	MONTREAL	8.92	9.22	9.43	9.51	12.98	13.41	13.73	13.84	4.59	4.75	4.86	4.90
50109	MONTREAL	15.60	16.16	16.52	16.69	22.69	23.51	24.04	24.28	8.03	8.32	8.51	8.59
54101	SUTTON	0.68	1.38	1.67	1.80	0.99	2.01	2.42	2.62	0.35	0.71	0.86	0.93
60104	OTTAWA	5.98	6.54	6.93	7.13	8.70	9.52	10.09	10.37	3.08	3.37	3.57	3.67
60204	WINDSOR	12.62	13.18	13.42	13.56	18.36	19.17	19.53	19.73	6.50	6.79	6.91	6.99
60211	WINDSOR	19.43	19.90	20.12	20.16	28.28	28.96	29.27	29.33	10.01	10.25	10.36	10.38
60424	TORONTO	16.13	16.89	17.26	17.36	23.48	24.57	25.11	25.27	8.31	8.70	8.89	8.94
60512	HAMILTON	21.55	23.17	24.23	24.53	31.35	33.71	35.26	35.70	11.10	11.93	12.48	12.64
61901	WALPOLE Isle.	34.40	34.64	34.77	34.91	50.06	50.40	50.60	50.79	17.72	17.84	17.91	17.98
64401	EGBERT	5.68	6.55	7.02	7.18	8.26	9.53	10.21	10.44	2.92	3.37	3.62	3.70
70119	WINNIPEG	3.47	3.68	3.85	3.97	5.05	5.36	5.61	5.78	1.79	1.90	1.99	2.05
90130	EDMONTON	2.82	3.00	3.11	3.22	4.11	4.36	4.53	4.69	1.45	1.54	1.60	1.66
90227	CALGARY	1.01	1.64	2.00	2.15	1.48	2.39	2.91	3.12	0.52	0.85	1.03	1.11
100111	VANCOUVER	4.42	5.62	6.34	6.65	6.43	8.18	9.23	9.67	2.28	2.90	3.27	3.42
100303	VICTORIA	0.00	1.50	2.47	2.92	0.00	2.18	3.60	4.25	0.00	0.77	1.27	1.50

APPENDIX B - NORMALIZATION OF AMBIENT PM DATA

The data set used in the incremental risk analysis is normalized to 365 days per year to account for the 1 in 6 days sampling schedule and missing measurements (see Box 1). The normalized ambient frequency distributions are presented in Tables B2 and B3 below. The maximum values reported in the tables are the maximum value observed in the three-year period. (**Editor's Note:** This dataset was selected in order to take advantage of the largest national dataset available at the time, which contained both PM₁₀ and PM_{2.5} information. This was to facilitate an across the country comparison of the risk information.)

Box 1: Normalization of Ambient PM Data

Objective: *to compute the average number of days from <15 to >50 $\mu\text{g}/\text{m}^3$ in 5 $\mu\text{g}/\text{m}^3$ increments*

- Use three years of data (January 1992 to December 1994) in order to have a larger number of values and a better approximation of the data distribution.
- Sort into concentration bins:
 - For PM₁₀ <15, 15-20, 20-25, 25-30, 30-35, 35-40, 40-45, 45-50, and > 50
 - For PM_{2.5} <10, 10-15, 15-20, 20-25, and >30
- Normalization of the data to represent one year is performed after the data is binned or summed.

Procedure:

- 1) Step through daily data for each site, binning all values.
- 2) Determine the maximum value (DAYMAX) for the data set.
- 3) Normalize all bin counts by multiplying by 365 and dividing by the total number of samples (N).

Table B2: Normalized PM10 Frequency Distribution profile (average # days/yr) Jan 1992 - Dec 1994

Site	Site #	<15 µg/m ³	15-20 µg/m ³	20-25 µg/m ³	25-30 µg/m ³	30-35 µg/m ³	35-40 µg/m ³	40-45 µg/m ³	45-50 µg/m ³	>50 µg/m ³	Max µg/m ³
Halifax	30118	224	76	26	23	13	0	3	0	0	42.4
Kejimkujik	30501	289	35	14	8	6	3	3	4	2	60.5
Saint John	40203	200	84	51	11	5	8	0	0	5	54.0
Montreal ₁	50104	109	78	56	48	22	14	17	7	15	98.6
Montreal ₂	50109	60	40	53	40	60	33	27	10	43	118.0
Sutton*	54101	282	38	32	11	0	0	3	0	0	42.1
Ottawa ₂	60104	193	73	43	24	10	6	6	4	6	58.0
Windsor ₁	60204	77	77	56	42	42	24	12	12	24	109.9
Windsor ₂	60211	45	55	51	57	42	36	19	15	46	104.8
Toronto	60424	93	55	47	43	38	25	16	18	28	101.9
Hamilton	60512	66	61	44	40	35	35	23	9	51	104.7
Walpole Isle.	61901	80	37	32	24	32	59	24	5	72	149.5
Egbert	64401	205	46	37	27	21	7	10	5	7	77.5
Winnipeg	70119	68	83	60	45	40	28	23	5	15	110.6
Edmonton	90130	71	71	73	52	23	23	13	15	25	132.0
Calgary	90227	101	65	68	49	32	21	8	6	15	75.9
Vancouver	100111	133	109	52	26	26	14	5	0	0	42.0
Victoria	100303	187	86	53	22	11	3	3	0	0	41.3

Table B3: Normalized PM2.5 Frequency Distribution Profile (average # days/yr) Jan 1992 - Dec 1994

Site	Site #	<10 µg/m ³	10-15 µg/m ³	15-20 µg/m ³	20-25 µg/m ³	25-30 µg/m ³	>30 µg/m ³	Max µg/m ³
Saint John	40203	235	77	28	15	6	4	38.3
Halifax	30118	235	73	31	16	8	3	37.6
Kejimkujic	30501	289	40	15	8	3	10	46.7
Montreal	50104	165	92	49	26	12	21	69.6
Montreal	50109	116	73	73	50	17	37	68.9
Sutton	54101	271	56	24	11	0	3	33.2
Ottawa	60104	213	65	45	18	10	15	53.8
Windsor	60204	124	76	76	42	20	28	60.6
Windsor	60211	98	80	80	38	22	47	85.6
Toronto	60424	134	75	48	40	24	43	66.4
Hamilton	60512	89	80	56	51	23	66	61.0
Walpole Island	61901	98	53	45	48	40	82	126.6
Egbert	64401	224	55	42	17	12	16	47.7
Winnipeg	70119	264	70	14	7	0	10	71.3
Edmonton	90130	266	56	24	9	6	4	56.3
Calgary	90227	255	72	23	8	4	2	35.6
Vancouver	100111	183	99	39	18	18	9	41.5
Victoria	100303	242	56	42	14	11	0	29.7

APPENDIX C THE EFFECT OF SAMPLING FREQUENCY ON THE ANNUAL MAXIMUM

Particulate matter levels exhibit a strongly skewed distribution, dominated by a large number of low values that mask the frequency or magnitude of the extremes of the distribution (Figure 1). The upper portion of the particulate matter frequency distribution in Canadian datasets may not be well characterized due to the limited number of observations available (a one in six day sampling schedule). The outcome is an underestimation of the annual maximums. The effect of sampling frequency on annual maximum values is calculated in Table C1.

Figure 1: Distribution of 24-h PM_{10} and $PM_{2.5}$ Concentrations for Station 50109 - Montréal.

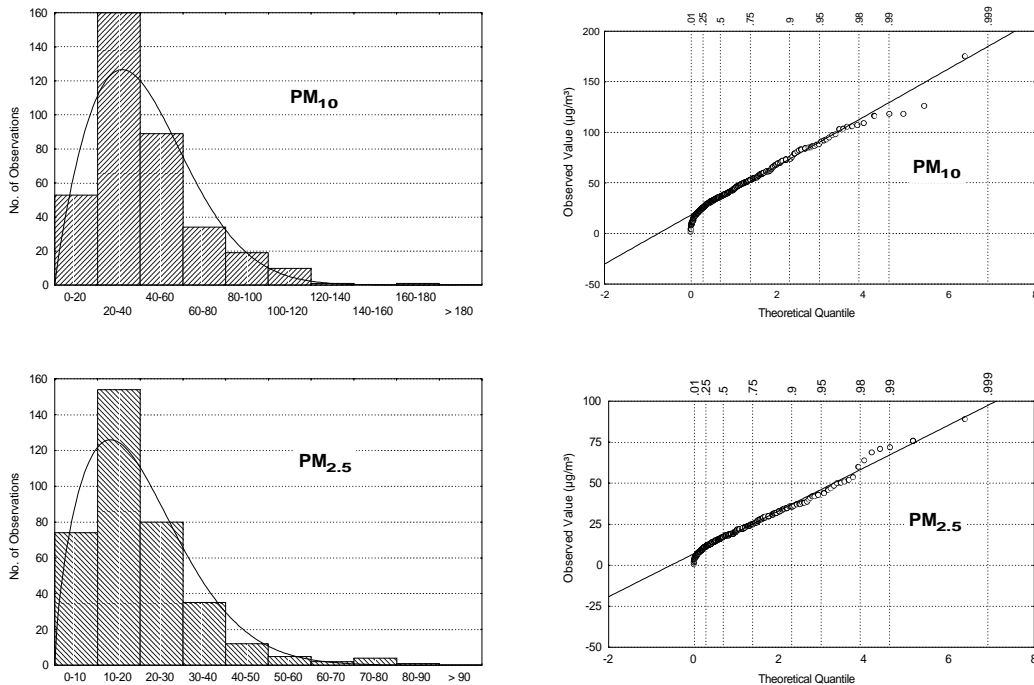


Table C1: Deviation of the observed annual mean and maximum from true values for 42 U.S. TSP sites (from Nehls and Ackland, 1973)

Sampling frequency in days	Average percent error (%)	
	Annual Mean	Annual Maximum
1 in 12	7.1	30.1
1 in 6	4.1	22.0
1 in 3	2.1	13.6
1 in 2	1.9	8.9

Dann (1994) estimates that measured and calculated maximum 24 hour average PM10 concentrations at NAPS sites differ by up to 61 $\mu\text{g}/\text{m}^3$ and 88 $\mu\text{g}/\text{m}^3$ at sites with few data samples (site 60104 in Ottawa and 61901 at Walpole Island, respectively). Sites with more observations show a smaller discrepancy between calculated and measured maximum 24-hour average PM10 concentrations. For example, for data from site 60211 in Windsor, the difference was only 16 $\mu\text{g}/\text{m}^3$.

Table C2 illustrates the relationship between annual PM_{2.5} maximums observed on the one-in-six-day NAPS sampling schedule and those observed on a daily sampling schedule. Table C3 illustrates the same relationship for PM₁₀. The data in these tables are from the Canadian Acid Aerosol Measurement Program, May through September 1993 to 1995. The first line for each station # / city is the ambient data from daily sampling, and the second line is ambient data from a one-in-six day sampling schedule collected on the NAPS sampling days. For PM_{2.5} the maximum concentrations measured during the one-in-six-day sampling schedule are 20% to 30% lower than those concentrations measured on a daily sampling schedule. The PM₁₀ the one-in-six-day sampling schedule measures 24-hour maximums which are 15% to 40% lower than the corresponding one-in-six day sampling schedule maximum PM₁₀ concentrations.

Table C2: Frequency Distribution for PM_{2.5} Mass ($\mu\text{g}/\text{m}^3$) for Daily vs. One-in-Six-Day Sampling (May-September 1993-1995)

Station #	City	N	Min.	10.0	30.0	50.0	60.0	70.0	80.0	90.0	95.0	99.0	Max.	Mean	Std. Dev.
40203	Saint John	432	0.6	2.6	5.1	7.7	9.0	11.1	13.2	17.3	21.3	29.5	53.2	9.2	6.5
40203	Saint John	75	1.1	2.3	4.6	6.9	8.0	9.3	12.6	14.8	17.6	37.6	37.6	8.1	5.8
30118	Halifax	233	0.8	4.6	6.1	7.6	8.6	9.7	12.3	15.4	20.8	32.1	43.4	9.3	5.8
30118	Halifax	41	1.5	4.9	6.4	7.9	8.6	9.7	13.0	15.6	22.4	32.1	32.1	9.9	6.4
30501	Kejimkujic	307	0.0	1.9	3.3	5.1	6.1	7.3	10.1	14.2	22.3	41.1	45.6	7.3	7.2
30501	Kejimkujic	66	0.7	1.4	2.6	5.0	5.9	6.2	7.3	12.0	22.3	36.5	36.5	6.2	6.6
50104	Montreal	317	1.4	4.5	7.6	10.0	11.8	14.1	17.1	22.3	29.6	33.1	58.8	12.1	7.7
50104	Montreal	55	2.6	3.7	6.5	9.3	10.1	11.6	12.5	19.6	29.9	30.8	30.8	10.6	7.1
54101	Sutton	136	1.0	2.4	4.1	6.1	7.3	9.1	11.5	15.6	19.4	22.6	33.2	7.7	5.5
54101	Sutton	24	1.7	2.4	3.5	4.8	6.1	7.0	9.3	12.0	19.4	33.2	33.2	7.1	6.9
64401	Egbert	260	0.7	2.7	5.3	8.0	9.8	12.8	17.4	24.1	31.0	39.3	47.7	10.9	8.8
64401	Egbert	60	0.7	2.7	4.6	7.1	9.5	12.8	19.3	25.8	31.8	39.3	39.3	10.8	9.3
60424	Toronto	376	2.0	5.0	7.8	11.5	14.4	17.3	21.6	28.3	34.5	42.5	52.2	14.4	9.5
60424	Toronto	69	2.7	4.1	6.6	9.1	10.6	16.3	20.1	28.3	35.0	37.4	37.4	12.8	9.2
60512	Hamilton	284	1.9	6.1	11.3	16.2	19.3	22.4	26.9	33.4	41.7	61.0	74.1	18.6	12.0
60512	Hamilton	52	2.7	6.1	10.7	13.4	16.7	22.2	27.6	32.5	40.2	71.3	71.3	18.0	12.7
60211	Windsor	273	3.9	7.0	11.5	16.0	17.9	21.2	26.7	36.6	47.9	62.2	85.6	19.4	13.1
60211	Windsor	59	4.7	6.8	9.9	15.5	17.3	20.8	25.5	38.6	48.3	56.1	56.1	18.1	12.2

Table C3: Frequency Distribution for PM10 Mass ($\mu\text{g}/\text{m}^3$) for Daily vs. One-in-Six-Day Sampling (May-Sep 1993-1995)

Station #	City	N	Min.	10.0	30.0	50.0	60.0	70.0	80.0	90.0	95.0	99.0	Max.	Mean	Std. Dev.
40203	Saint John	432	1.4	5.3	9.6	13.6	16.0	18.5	21.8	28.2	32.6	45.0	67.7	15.5	9.4
40203	Saint John	75	1.6	4.9	8.7	11.4	13.4	16.2	20.8	24.0	28.7	45.0	45.0	13.5	7.9
30118	Halifax	235	2.9	8.6	11.2	13.2	14.6	16.1	19.1	23.1	28.6	40.8	51.5	14.9	6.8
30118	Halifax	44	4.6	8.4	11.2	12.8	13.5	15.3	17.5	23.1	30.4	40.6	40.6	14.5	7.2
30501	Kejimkujic	307	0.4	3.9	6.0	8.1	9.5	11.2	14.7	19.8	27.6	45.2	50.0	10.6	8.1
30501	Kejimkujic	66	1.5	3.5	4.9	7.5	8.4	9.8	11.5	16.3	25.8	42.8	42.8	9.2	7.5
50104	Montreal	340	4.3	9.8	14.6	18.8	21.5	24.8	29.0	35.4	41.5	50.7	85.8	21.2	10.7
50104	Montreal	59	5.3	9.2	13.5	17.1	18.5	21.0	28.2	37.0	41.3	50.7	50.7	19.4	9.8
54101	Sutton	136	2.5	4.6	6.9	9.5	11.3	13.4	15.7	20.8	24.0	29.5	42.1	11.3	6.5
54101	Sutton	24	2.6	4.5	6.5	7.6	9.2	10.2	13.4	15.1	24.0	42.1	42.1	10.2	8.2
64401	Egbert	263	1.2	5.3	9.2	13.9	16.8	21.8	26.6	33.0	43.0	54.9	77.5	17.4	12.4
64401	Egbert	63	1.2	4.9	9.2	12.1	16.8	22.6	29.7	37.7	44.1	54.9	54.9	17.4	13.1
60424	Toronto	376	6.5	11.1	15.8	21.4	25.2	29.2	34.9	43.5	49.7	68.6	74.0	24.7	13.1
60424	Toronto	69	6.8	8.9	13.7	17.4	20.8	27.3	32.8	39.0	46.9	70.2	70.2	22.2	12.7
60512	Hamilton	284	3.9	12.7	19.6	28.2	32.3	37.7	43.8	54.0	62.0	104.7	176.8	31.5	19.9
60512	Hamilton	52	5.1	13.8	18.6	24.5	30.0	38.5	43.7	48.6	61.6	102.4	102.4	30.2	19.0
60211	Windsor	288	8.2	14.9	23.2	29.3	33.6	37.2	45.8	57.1	68.8	100.7	104.8	33.5	18.0
60211	Windsor	74	9.2	14.2	19.0	26.8	28.7	33.6	41.1	53.8	61.6	79.1	79.1	29.6	15.5

Figures 2 and 3 graphically display the relationship between the observed annual maximums and the 98th, 95th and 90th percentiles based upon the 1-in-6 day sampling schedule. Based upon these plots, the annual maximums are ~30% greater than the 98th percentile. Given that the one-in-six day schedule underestimates the annual maximums by 20% to 30%, if one assumes that the relationships remain the same for a daily sampling schedule, then the observed annual maximums on the 1-in-6 day sampling schedule approximate the 98th percentile of the actual ambient distribution.

Figure 2: Relationship between Maximum PM₁₀ Concentrations (µg/m³) and Selected Percentile Values - All Dichot Sites 1988-1996

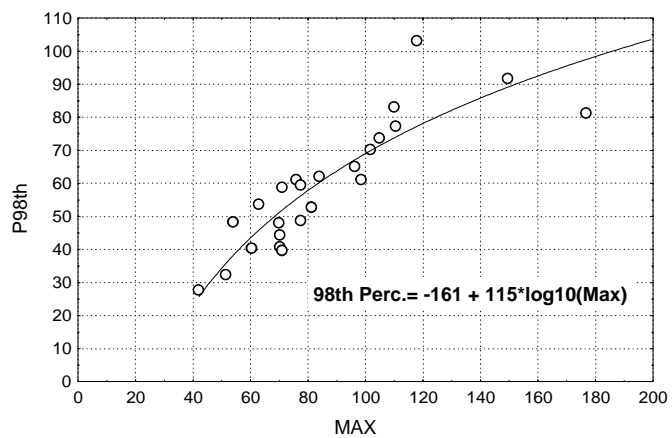
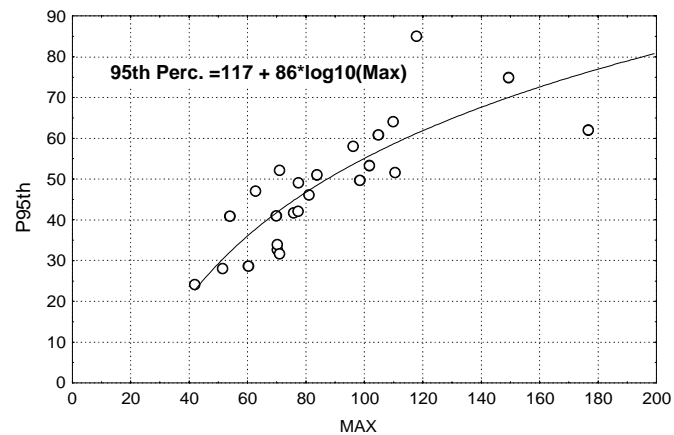
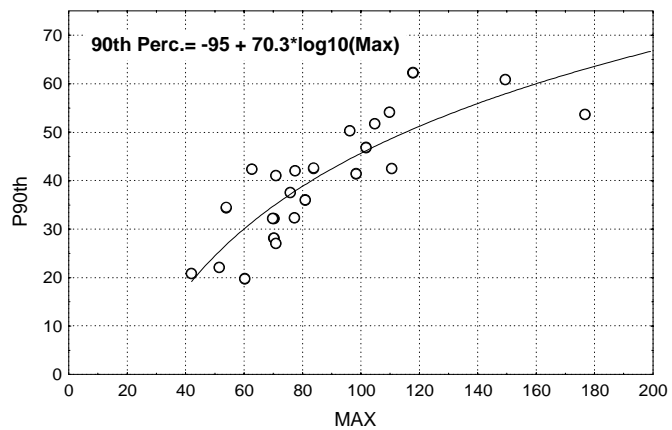
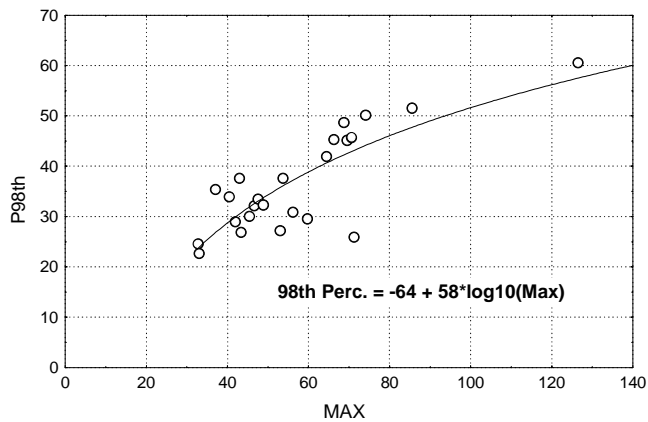
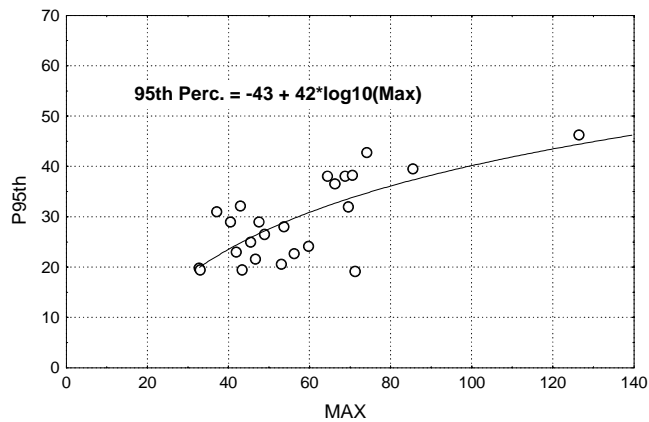
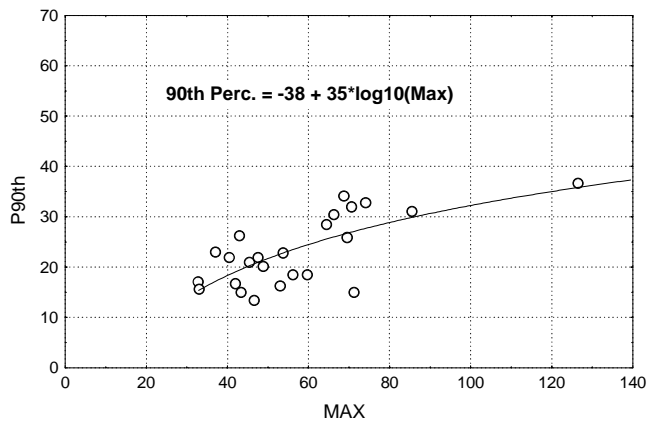


Figure 3: Relationship between Maximum PM_{2.5} Concentrations (µg/m³) and Selected Percentile Values - All Dichot Sites 1988-1996



APPENDIX D PROPORTIONAL LINEAR ROLLBACK

The rollback calculations used the maximum 24-hour PM concentration to estimate the percentage reduction required in the ambient frequency distribution to simulate no exceedances of a range of incremental concentrations. The 24-hour maximum PM concentration is representative of the 98th percentile of ambient concentrations, not actual peak concentrations. To estimate the required rollback percentage in order to reduce the maximum concentration to various incremental target levels, in each case, the ratio between the target level and the maximum concentration at a given site was used to scale back the data for that site (the linear proportional rollback analysis). Given that there are no perfect empirical schemes to generate predicted ambient particulate matter concentrations, the linear proportional rollback approach does provide a useful perspective. (**Editor's Note:** The analysis described here represents a preliminary approach using the data and information available at the time.) The details of the calculations are also noted in Box 2.

Box 2: Rollback Calculations

Objective: *to calculate the percentage of ambient concentration reductions required to reach various targets.*

- Use the normalized ambient PM data
- Recalculate the frequency distribution under each rollback scenario, adjusting the data such that the 24-hour maximum is less than or equal to the following targets:
 - For PM₁₀: 25, 30, 35, 40, 45, and 50 µg/m³
 - For PM_{2.5}: 15, 20, 25 and 30 µg/m³
- the level above which PM values are summed for the year is set to 25 µg/m³ for PM₁₀ and 15 µg/m³ for PM_{2.5}
- the lower limit could be equal to "background" values of 15 µg/m³ for PM₁₀ and 10 µg/m³ for PM_{2.5} (selected to be representative of conditions in south western Ontario, which has the highest background concentrations likely to be observed in Canada.)

Procedure:

- 1) For each rollback scenario, first compute the reduction factor as $REDN = (DAYMAX - ROLLBACKVAL) / DAYMAX$
- 2) Step through daily data, adjusting each value by multiplying it by $(1-REDN)$.
- 3) Bin the adjusted values, and normalize the adjusted values by multiplying by 365 and dividing by the total number of samples, N.

Note 1: The background value may be adjusted and subsequently the % reduction for alternate background or threshold values using: $REDN = DAYMAX - ROLLBACKVAL / (DAYMAX - BKGD)$.

There are other approaches, (Abt Associates, 1996) such as peak shaving or a weighted proportional rollback, to adjusting the frequency distribution such that they do not exceed a target concentration limit that is less than current ambient levels. Peak shaving involves cutting off the upper end of the PM frequency distribution above a specified level while leaving the daily concentrations at or below the target level unchanged. This method is unlikely to reflect the actual changes in ambient concentration distributions that would occur under various attainment

scenarios. A weighted proportional rollback involves applying one reduction factor to days with higher concentrations and a different reduction factor to days with lower concentrations. The maximum is still reduced to at or below the target level. Currently, there is insufficient information to determine the multiple factors that a weighted proportional rollback technique would require.

The method by which daily concentrations are reduced to meet the concentration targets will affect the resulting concentration distribution profile, and consequently any estimated associated health risks. In the development of risk estimates the U.S. EPA reviewed various rollback methodologies, and analysed the extent to which historical changes in particulate matter air quality have been linear (Abt, 1996). Proportional linear rollbacks accounted for the vast majority of the variation between consecutive years of data, however the relationships become less linear at either extreme, that is, at the low and high ends of the concentration distributions. Incorporating background estimates in the rollback calculations improved the predictive power of the linear regressions between consecutive years. Exponential and logarithmic forms of rollbacks performed more poorly than the linear approach.

In airsheds that are not subject to significant regional transport of particles, all sources are potentially subject to management actions aimed at reducing PM. If those actions target all PM sources, then an x% reduction of PM emissions should on average lead to an x% reduction in ambient PM. For airsheds without significant regional PM contribution, ambient levels are, in effect, the sum of two independent PM distributions. PM distributions resulting from local sources will be subject to local management actions and can be appropriately reduced; the PM distributions not subject to local management actions will not change. So only the PM distribution from local sources is being reduced and the regional distribution (background) is not. The net effect is a much less than proportional decrease in ambient PM. Also, airsheds in which management actions to decrease PM focus on only part of the PM/PM precursor sources will have a less than proportional decrease in ambient PM. In these areas, proportional rollback generates an optimistic reduction in the mid-range PM concentrations, and, as such, the rollback overestimates the potential health benefits.

The percent reduction in ambient concentrations required to rollback the normalized frequency distribution to meet the targets (in $5 \mu\text{g}/\text{m}^3$ intervals, between 25 and 50 for PM_{10} , and between 15 and $30 \mu\text{g}/\text{m}^3$ for $\text{PM}_{2.5}$) are presented in Table D1 and D2, respectively. The variation between the estimated lowest and highest ambient reductions reflect the variability among geographic regions of the country.

Table D1: Estimated Percent Reduction in Ambient Concentrations for PM₁₀ Rollback Targets.

City	Station	50 µg/m³ %	45 µg/m³ %	40 µg/m³ %	35 µg/m³ %	30 µg/m³ %	25 µg/m³ %
Halifax	30118	0	0	6	18	29	41
Kejimikujic	30501	17	26	34	42	50	59
Saint John	40203	7	17	26	35	44	54
Montreal ₁	50104	49	54	59	65	70	75
Montreal ₂	50109	58	62	66	70	75	79
Sutton	54101	0	0	5	17	29	41
Ottawa	60104	14	22	31	40	48	57
Windsor ₁	60204	55	59	64	68	73	77
Windsor ₂	60211	52	57	62	67	71	76
Toronto	60424	51	56	61	66	71	75
Hamilton ₁	60512	52	57	62	67	71	76
Walpole Island	61901	67	70	73	77	80	83
Egbert	64401	35	42	48	55	61	68
Winnipeg	70119	55	59	64	68	73	77
Edmonton	90130	62	66	70	73	77	81
Calgary	90227	34	41	47	54	60	67
Vancouver	100111	0	0	5	17	29	40
Victoria	100303	0	0	3	15	27	40

Table D2 Estimated Percent Reduction in Ambient Concentration for PM2.5 Rollback Targets

City	Incremental Targets			
	15 µg/m³	20 µg/m³	25 µg/m³	30 µg/m³
Saint John	61	48	35	22
Halifax	60	47	34	20
Kejimikujic	68	57	46	36
Montreal	78	71	64	57
Montreal	78	71	64	56
Sutton	55	40	25	10
Ottawa	72	63	53	44
Windsor	75	67	59	51
Windsor	82	77	71	65
Toronto	77	70	62	55
Hamilton	75	67	59	51
Walpole island	88	84	80	76
Egbert	69	58	48	37
Winnipeg	79	72	65	58
Edmonton	73	64	56	47
Calgary	58	44	30	16
Vancouver	64	52	40	28
Victoria	50	33	16	0

APPENDIX E.

ANNUAL AVERAGE BACKGROUND CONCENTRATIONS FOR PM₁₀ and PM_{2.5}

When estimating benefits of avoided annual health or environmental impacts (i.e., annual cases of a particular endpoint, for example, premature mortality or hospital admissions) a cumulative estimate of exposure is required. This is calculated from a sum of particulate matter concentrations for each day of the year.

In order to avoid estimation of benefits which no amount of broad-based emission reduction strategy could achieve, it is appropriate to exclude impacts due to naturally occurring background concentrations. Observations at rural or remote locations with none or minimal anthropogenic contributions are used to estimate background concentrations. Ideally one would identify a background concentration for each day of the year for each area of the country to account for the temporal and spatial variation in the background, to provide a more complete picture of concentration distribution. Since this is not available, average estimates are made.

Since the future risk or benefit estimates are developed on an annual basis (summed from individual daily risks), an annual average estimate of background will suffice for the purpose of this analysis. The annual average background estimate is subtracted from each daily concentration for estimating the risks. Thus an annual average natural non-anthropogenic concentration is considered an appropriate metric to use in estimating risks. In estimating the future risks (or the benefits of avoided impacts) the annual average background is subtracted from the predicted concentration distribution based on application of the rollback algorithms. This ensures that the sum of avoided impacts (i.e., the benefits) does not include attribution of benefits to levels below natural non-anthropogenic background.

Rationale for the selection of the background estimate for PM₁₀.

The following considerations led to the selection of 5 µg/m³ as the annual average estimate for natural non-anthropogenic background.

- 1) “*Natural* background” PM₁₀ concentrations range between 4 - 6 µg/m³ (west vs east) in the U.S. Data from remote sites along with emission estimates for natural sources were used in estimating this “natural background”. (WGAQOG, 1998 citing Trijonis et al., 1990).
- 2) The US EPA Criteria Document for PM provides estimates of regional background PM₁₀ for annual or longer averaging times. The range for the eastern U.S. is 5-11 µg/m³ and for the western U.S. the range is 4-8 µg/m³. It is noted that the *lower* bounds of the estimates are based on analyses of *natural* versus anthropogenic and/or *natural* trace species. The *upper* bounds are derived from multi-year annual averages of the “clean” remote monitoring sites in the IMPROVE network and reflect the effects of *both natural and anthropogenic* PM and precursor emissions from within North America (WGAQOG 1998 citing NAPAP, 1991; Trijonis, 1982; Malm et al., 1994;). Rural sites in eastern Canada can have annual background concentrations as high as 15 µg/m³ but these sites are influenced by strong anthropogenic contributions (PM SAD, 1998).
- 3) An inspection of levels between 1988 and 1994 of concentrations observed at some of the cleaner rural Canadian NAPS sites such as Sutton, Quebec, Egbert, Ontario and Kejimikujik, Nova Scotia also provides direction for estimating background concentrations. On the cleanest days, as determined by using 10th percentile concentrations, since no back trajectory analyses

are available, the concentrations were 4, 5 and 5 $\mu\text{g}/\text{m}^3$ respectively (WGAQOG 1998, Table 5-10).

Rationale for the selection of the background estimate for $\text{PM}_{2.5}$.

The following considerations led to the selection of 2.5 $\mu\text{g}/\text{m}^3$ as the annual average estimate for natural non-anthropogenic background.

Minimal information exists upon which to base a background estimate for $\text{PM}_{2.5}$ due to lack of monitoring in rural or remote areas. However, it is assumed that the ratio observed between ambient concentrations of PM_{10} and $\text{PM}_{2.5}$ in areas where both have been monitored would apply generally. Thus the $\text{PM}_{2.5}$ background estimate is half of that for PM_{10} . The observed relationships between PM size fractions is presented in WGAQOG 1998, Chapter 5.

Editor's note: Though this estimate of background for $\text{PM}_{2.5}$ was not developed as part of the analysis undertaken to estimate PM risks and benefits, it is included here for the sake of completeness.

APPENDIX F CUMULATIVE EXPOSURE INDEX

Environmental quality criteria have historically been expressed as concentration limits, set at or below the concentration at which known effects on human health or the environment occur. Averaging times varied from 30 minutes to more than one year. Total suspended particulate matter (TSP) acceptable limits, for example, are $120 \mu\text{g}/\text{m}^3$ averaged over 24-hours, and $70 \mu\text{g}/\text{m}^3$ averaged annually. The public interpreted ambient concentrations of suspended particulate that were below the maximum acceptable limits to be without risk, i.e., provide complete protection. Therefore, many believed that effects on human health or the environment occurred only when concentrations exceeded the objective(s). Air quality managers viewed the “air quality problem” as one of controlling episodes of high pollutant concentrations and directed their efforts towards reducing the number of AQO “exceedances”. As a result, ambient TSP concentrations in the air were below the acceptable limits most of the time.

The results of scientific research on PM over the past decade have invalidated both the assumption that episodes of high PM concentrations are associated with adverse effects, and the assumption that low levels of ambient PM are without adverse effects. The “no observed effect level” for PM cannot be reliably identified and current ambient concentrations of PM in many parts of the country are often in the range where effects on health have been found. Public health impacts, including deaths, hospitalizations, and increased symptoms and medication use, are associated not only with episodes of poor air quality, but with every day ambient PM concentrations. The significance of the new evidence on PM warrants a paradigm shift both in the form of the objective and in how air quality is managed.

There are biologically significant adverse effects, ranging in severity from mild to life-threatening depending on the frequency and duration of exposure, occurring at the Reference Levels. The RL represents the statistically derived LOAEL, a level above which the increase in the incidence of health effects in the population can be quantified and the uncertainty estimated. This is similar in concept to the benchmark dose, which is a statistical lower confidence limit for a dose that produces a predetermined change in response rate of an adverse effect. However, no margin of safety is incorporated into the identification of the LOAEL, whereas a margin of safety is applied when calculating a benchmark dose. Setting an ambient air target at or below the Reference Level will provide a reasonable guide, for identifying the periods when air quality may have effects on human health. If the target is set above the Reference Level, there may be failure to appreciate and address impacts of concentrations in the range from the Reference Level to the recommended target. Reliance on the ambient air targets above the Reference Level may lead to an under-estimation of the frequency, duration and magnitude of the PM “problem” in an airshed. It may create the impression that air pollution is a rare or episodic event in areas where this is not the case.

The effects of PM pollution are related to both the concentration and duration of exposure in the effects range, therefore, an ambient target that incorporates both factors is inherently a more accurate descriptor of effects on human health than a measure that considers only time above a fixed concentration.

Estimates of the extent of PM related impacts on health have used PM_{10} increments (typically 10

µg/m³-days; Vedal, 1995). Although the biological basis is different, the form is analogous to the SUM60 proposed for ozone related effects on vegetation (ref. NO_x-VOC Veg. Report). For PM related effects on human health, a target could be tied directly to the Reference Level, or the Reference Level with an appropriate margin of safety. It would be a measure of the exposure to PM levels above the lower limit. For example, if expressed in terms of PM_{2.5} with a Reference Level of 15 µg/m³, one day where ambient PM_{2.5} was 35 µg/m³ would accumulate an exposure of 35-15= 20 µg/m³-days. The target could be set in terms of the exposure, in µg/m³-days, over 30 days or year. A monthly integration would focus attention on the seasonal variation of PM problems and consequently on the identification of the various causes of PM pollution at different times of the year. The calculation of a 30 day cumulative effects index (CEI) would be as follows:

CEI = sum over 30 days of (daily ambient PM concentration - PM Reference Level or RL with a margin of safety) when the difference was positive

Possible advantages to using a cumulative exposure AQO are that it:

- directly relates to the human health effects air quality programs strive to minimize;
- explicitly incorporates the Reference Level and allows for a reduction in the ambient target without revision of the long term target or incorporation of new science that causes the Reference Level to change;
- is a measure of exposure to PM in the range where effects are expected;
- provides a base from which health impacts can be estimated;
- provides a solid basis for building airshed plans and management strategies to protect human health by incorporating all the factors elevating PM levels not just those responsible for peaks in PM concentrations;
- would help air quality managers and the public understand that in many communities, elimination of PM-related health outcomes will involve more than episode management.

The link between the anticipated health impacts and the selection of an ambient target is much more transparent if based on the cumulative effects index. It is also possible to structure a two component ambient target that has both a 30 day cumulative effects index and a 24 hour concentration limit. Adding the concentration limit to the cumulative effects index effectively adds a technique for addressing PM episodes and allows the continued application of the episode management approaches honed over the past several decades.

The PM_{2.5} and PM₁₀ 30 day cumulative health indices are selected in recognition of the existing index values (Table x or Table xxx), annual health impacts per million people listed in Table xx and a desire to maximize the protection from PM-related health impacts. Although the PM related mortality was a key factor in selecting the index values, it is equally true that all of the other health outcomes related to ambient PM played a significant role in the selection deliberations. The selected index values are:

Table F1: Annual Health Impact as a Function of PM₁₀ and PM_{2.5} Cumulative Effects Index

PM₁₀: Annual Health Impacts vs. 30 Day Cumulative Effects Index				
30 Day Cumulative Effects Index	Equivalent Annual Cumulative Effects Index *	Estimated Annual PM ₁₀ Health Impacts (per million people)		
µg/m ³ -day		Mortality		
0	0	0		
10	120	1.7		
20	240	3.4		
30	365	5.1		
40	485	6.8		
50	605	8.5		
100	1215	17		
x/12	X	x(0.014)		

PM_{2.5}: Annual Health Impacts vs. 30 Day Cumulative Effects Index				
30 Day Cumulative Effects Index	Equivalent Annual Cumulative Effects Index*	Estimated Annual PM _{2.5} Health Impacts (per million people)		
µg/m ³ -day		Mortality	Respiratory Hospital Admissions	Cardiac Hospital Admissions
0	0	0	0	0
5	60	1.6	0.7	0.6
10	120	3.1	1.4	1.2
15	180	4.7	2.1	1.8
20	240	6.2	2.8	2.4
25	300	7.8	3.5	3.0
30	365	9.5	4.3	3.7
50	605	15.6	7.1	6.1
x/12	x	x(0.026)	x(0.0118)	x(0.010)

*Equivalent Annual Cumulative Effects Index determined, for the purpose of estimating annual health impacts, by assuming that the ambient PM levels were such that the 30 Day Index value was equaled for any 30 day period during the year.

Table F2: HEALTH OUTCOMES EXPECTED - Based on Ambient PM₁₀ and P_{2.5} Levels Measured over 1992 to 1994, Relative Risks from various studies; national health outcome incidence (annual, per million people)

PM ₁₀ : Expected Health Impacts at Current Ambient Levels and Avoided Impacts if a 30 Day Cumulative Effects Index of 50 µg/m ³ -day is Achieved					
Station+	City	PM ₁₀ Cumulative Health Index		Annual Mortality	Avoided Annual Mortality if a 30 Day Cumulative Effects Index of 50 µg/m ³ -day is Achieved*
		µg/m ³ -day		per million people	per million people
		Annual Index	Equivalent 30 Day Index **		
30118	Halifax	192	16.0	2.8	0
30501	Kejimkujic	309	25.8	4.5	0
40203	Saint John	462	38.5	6.7	0
50104	Montreal	1446	121	20.9	12.4
50109	Montreal	3826	319	55.2	46.7
54101	Sutton	72	6.0	1.0	0
60104	Ottawa	1032	86.0	14.9	6.4
60204	Windsor	2148	179	31.0	22.5
60211	Windsor	3395	283	49.0	40.5
60424	Toronto	2545	212	36.7	28.2
60512	Hamilton	3515	293	50.8	42.3
61901	Walpole Island	5536	461	79.9	71.4
64401	Egbert	819	68.3	11.8	3.3
70119	Winnipeg	1929	161	27.9	19.4
90130	Edmonton	1403	117	20.3	11.8
90227	Calgary	1370	114	19.8	11.3
100111	Vancouver	738	61.5	10.7	2.2
100303	Victoria	229	19.1	3.3	0

+ Stations selected from the NAPS database; included if both PM₁₀ and PM_{2.5} were available

* Avoided Impacts calculation assumed a uniform distribution of contributions to the Cumulative Effects Index throughout the year; the Avoided Impacts would be higher if the contributions were concentrated in only a few 30 day periods.

** The Equivalent 30 Day Index is determined by dividing the Annual Index by 12; this is equivalent to assuming that the daily PM levels that contributed to the Annual Index were uniformly distributed throughout the year.

PM_{2.5}: Expected Health Impacts at Current Ambient Levels and Avoided Impacts if a 30 Day Cumulative Effects Index of 25 µg/m³-day is Achieved									
Station+	Name	PM _{2.5} Cumulative Health Index µg/m ³ -day		Impacts at Current Ambient Levels per million people			Avoided Impacts if 30 day CEI of 25 µg/m ³ -d is achieved *(per million people)		
		Annual Index	Equivalent 30 Day Index**	Annual Mortality	Annual RHA	Annual CHA	Annual Mortality	Annual RHA	Annual CHA
30118	Halifax	303	25.3	7.9	3.6	3	0.1	3.6	3
30501	Kejimikujic	375	31.3	9.8	4.4	3.8	2	4.4	3.8
40203	Saint John	309	25.8	8.1	3.6	3.1	0.3	0.1	0.1
50104	Montreal	946	78.8	24.7	11.2	9.5	16.9	7.7	6.5
50109	Montreal	1661	138	43.3	19.6	16.7	35.5	16.1	13.7
54101	Sutton	179	14.9	4.7	2.1	1.8	0	0	0
60104	Ottawa	710	59.2	18.5	8.4	7.1	10.7	4.9	4.1
60204	Windsor	1350	113	35.2	15.9	13.6	27.4	12.4	10.6
60211	Windsor	2006	167	52.3	23.7	20.2	44.5	20.2	17.2
60424	Toronto	1728	144	45	20.4	17.4	37.2	16.9	14.4
60512	Hamilton	2442	204	63.6	28.8	24.5	55.8	25.3	21.5
61901	Walpole Island	3474	290	90.5	41	34.9	82.7	37.5	31.9
64401	Egbert	714	59.5	18.6	8.4	7.2	10.8	4.9	4.2
70119	Winnipeg	396	33	10.3	4.7	4	2.5	1.2	1
90130	Edmonton	321	26.8	8.4	3.8	3.2	0.6	0.3	0.2
90227	Calgary	214	17.8	5.6	2.5	2.1	0	0	0
100111	Vancouver	662	55.2	17.2	7.8	6.6	9.4	4.3	3.6
100303	Victoria	291	24.3	7.6	3.4	2.9	0	0	0

+Stations selected from the NAPS database; included if both PM₁₀ and PM_{2.5} were available

* Avoided Impacts calculation assumed a uniform distribution of contributions to the Cumulative Effects Index throughout the year; the Avoided Impacts would be higher if the contributions were concentrated in only a few 30 day periods.

** The Equivalent 30 Day Index is determined by dividing the Annual Index by 12; this is equivalent to assuming that the daily PM levels that contributed to the Annual Index were uniformly distributed throughout the year.

APPENDIX G Source Apportionment of PM₁₀ and PM_{2.5}

Characterizing the sources and atmospheric processes that contribute to ambient PM₁₀ and PM_{2.5} levels is a complex process, as ambient particle composition is the result of primary emissions and secondary formation from diverse biogenic and anthropogenic source types. Chow and Ono (1992) summarize the results of PM₁₀ source apportionment analyses in urban areas and provide the following list of (potentially) major contributors to PM₁₀.

Sources of primary particulate:

- fugitive dust from roads, construction and agriculture
- wood smoke from domestic and industrial combustion of lumber and forest fires
- primary particulate emissions from the mobile sector (diesel and gasoline engines)
- other fuel combustion and industrial processes
- crustal material

Primary particles, those which are emitted directly from sources, undergo few physical or chemical changes between source and receptor. Consequently, atmospheric concentrations are approximately proportional to their emission concentrations. Secondary particles, those which form in the atmosphere from their precursor gases, undergo physical and chemical transformations masking the original source chemical composition. This creates a challenge to identify source types for secondary particles. Most sulphate and nitrate particles are of secondary origin, resulting from sulphur dioxide, nitrogen dioxide and ammonia emissions. Some organic carbon is also of secondary origin, resulting from volatile organic gas emissions.

This chapter will review common methods used to identify sources of particulate matter, review recent source apportionment studies (with particular emphasis on Canadian efforts), and illustrate the key source types which contribute to particulate matter levels in the atmosphere.

1. Methods of Source Apportionment

There are a variety of approaches for determine the contributing sources to PM₁₀ and PM_{2.5} loadings. Source apportionment, or source attribution, allows for the identification (qualitatively and quantitatively) of contributing sources to support the development of atmospheric models and air quality management strategies. A brief summary of common source apportionment methodologies with particular emphasis on the difference between PM₁₀ and PM_{2.5} sources follows.

Characteristic Source Profiles

Particles (aerosols) from a specific source have unique physical and chemical characteristics. The characterization of a particular source type based upon the relative size fractions and chemical composition of emitted particles provides a source fingerprint or source profile. For example, the composition of forest fire aerosols and sea spray aerosols is very different.

Source profiles, describing the unique chemical composition of the emissions, have been developed for many aerosol sources types. As there are many sources of each type, all potential sources must be analyzed to obtain the characteristic source type profile. This requires detailed chemical speciation and analysis of the particulate samples. The characterization of fine particulate emissions from motor vehicles requires many samples from different vehicles under various representative driving conditions. It is possible to borrow source profiles developed for other locations or similar source operating conditions. However, these borrowed signatures may not always be truly

representative of the sources in the particular region of interest. For example, available motor vehicle source profiles are almost exclusively based on leaded gas information from studies in the United. The need for care in applying this information to the Canadian situation is obvious.

Descriptive Analysis - Natural Tracers

The chemical composition of particulate matter is key to determining the originating source types. However, if component species quantification is not possible, much can be inferred from the relative abundances of different species, avoiding complicated source characterization techniques. For instance, silicon, iron, aluminium, and calcium are indicative of crustal materials, providing evidence of the relative importance of soil-related contributions to the sample. In addition, the correlation between different species concentration may be used to identify sources. For example, if sodium ion concentration is correlated with the concentration of chloride ions, then a marine source contribution may be significant.

Artificial Tracers

In some source apportionment studies, interest is focused on selectively identifying one or two sources or source types which may have similar source profiles. In such situations, a unique tracer is introduced into the emission stream of the sources of interest. Analysis of samples for the unique tracer identifies the contributor. The WHITEX project (Malm et al., 1990) identified the contributions of one electrical power generating plant to wintertime haze in the lower Colorado River area, using this technique.

Receptor Modelling

Receptor models start with observed particle concentrations at a receptor (i.e. at a monitoring site) and seek to apportion the observed concentrations between several source types based on knowledge of the compositions of the source and receptor materials (Henry et al., 1984). There are two primary types of receptor models: chemical mass balance (CMB) and multivariate analysis techniques.

CMB modelling is based upon several assumptions (Hopke, 1985): chemical source profiles are known and constant, the source emissions are linearly independent in time and space, and the chemical species are non-reactive. The model simply states that the concentration of an element measured at a receptor is a linear addition of all source contributions (Henry, 1987). The linear equations are:

$$C_{ik} = \sum_{j=1}^p G_j S_{jk} F_{ij} + e_{ik} \quad (1)$$

where C_{ik} , the concentration of the i^{th} species in the k^{th} sample, is expressed as the sum of the contributions of the i^{th} species from p sources, S_{jk} is the mass per unit volume of air contributed by the j^{th} source to the k^{th} sample, F_{ij} is the fractional abundance of the i^{th} species in the j^{th} source profile, and e_{ik} is the random error of measurement for the i^{th} species in the k^{th} sample. The solution expresses each receptor concentration of a chemical species as a linear sum of products of source profile species and source contributions (Friedlander, 1973).

One of the difficulties in receptor modelling is source identification of secondary aerosols. The sources of the precursor gases are often masked by the formation process. A variant of CMB can be used to estimate a source contribution to secondary aerosols if additional assumptions are imposed

which take into account, for example, reaction rates and atmospheric transport times. The net effect of these assumptions is to attribute the secondary aerosol in direct proportion to the apportioned primary mass and the relative emission strength of each source (Malm et al., 1994).

Examples of the CMB analysis, summarized in Chow et al. (1993), are summarized in Table 1. Additionally, in urban areas, primary industrial emissions may contribute significantly to ambient PM₁₀; in coastal locations, marine aerosols may also be important. Wood smoke has also been identified as a significant contributor to PM₁₀ in urban areas of Canada (Intera Environmental Consultants Ltd., 1993).

Table 1 Source type contributions to PM₁₀ (µg/m³), as determined with CMB analysis, in the United States between 1984 and 1990 (adapted from Chow et al., 1993a)

Sampling Location	Average Contributions to PM ₁₀ (µg/m ³)						Measured PM ₁₀
	Primary particulate				Secondary Particulate		
	Geological	Construction	Motor Vehicle Exhaust	Vegetation Burning	Ammonium Sulphate	Ammonium Nitrate	
Palm Springs, CA	16.4	1.4	2.3	5.1	3.7	4.2	35.1
Indio, CA	33.0	3.0	4.4	7.1	3.6	4.1	58.0
Tucson, AZ	26.0	5.1	14.0	0.0	1.0	0.2	48.0
Orange Grove, AZ	20.0	0.0	15.0	0.0	0.4	0.4	34.2
Craycroft, AZ	13.0	0.0	8.3	0.0	0.6	0.6	23.4
Coruna de Tucson, AZ	17.0	0.0	1.6	0.0	0.0	0.0	19.1

The traditional approach to CMB modelling involves a least squares solution to Equation 1. This model requires source profiles as input. Ideally, source profiles should be determined concurrently with ambient aerosol measurements. For practical reasons, this is usually not done, and source profiles are obtained from previous studies at other locations. The major assumption for applying the CMB model is that the source profiles are comprehensive (account for all of the ambient aerosol) and representative. If these conditions hold, then the uncertainty of the source contribution estimates (SCEs) may be random and can be evaluated in statistical terms. If these assumptions are not met, then the uncertainty in the SCEs is not random but systematic. This form of error is known as bias. Bias is undoubtedly the most important source of error in CMB source apportionment but for practical purposes, it is difficult to quantify. The problem is confounded by the fact that the CMB least squares solution is not unique; more than one combination of source profiles can give equally good solutions. Fortunately, the potential for bias error can be minimized through careful and reasonable selection of source profiles.

Owing to the uncertainties involving contributing emission sources, and the applicability of source profiles from other studies, a number of alternative (typically receptor-based) multivariate approaches may be employed to make inferences regarding source contributions. These multivariate receptor models are distinguished from other receptor models by their attempt to determine both the source apportionment and composition of the source material from a series of ambient observations. Some examples of this approach are: principal component analysis (e.g. Hopke et al., 1976; Pryor et al., 1994a), principal component analysis and regression analysis (e.g. Thurston and Spengler, 1985) and procrustes target analysis/target transformation factor analysis (Richman and Vermette, 1993). The advantages of these over the CMB analysis is that detailed and complete source emission profiles are not required. Multivariate techniques allow a large data set

(consisting of speciated ambient samples) to be simplified into fewer representative variables that reflect the attributes of the data. The number of representative variables indicates the number of source types of ambient particles (Pryor and Steyn, 1994a). Like CMB analyses, these techniques typically yield estimates of the contribution of each source type, not individual sources, to ambient particulate matter.

In summary, multivariate analysis may be used where detailed source fingerprint information is not available, as it requires only a limited amount of *a priori* information on the nature of the sources. However, multivariate analysis lacks the physical basis of CMB modelling, and the results are dependent on subjective interpretation of factors. Furthermore, multivariate techniques require an adequate number of ambient samples and variability within the data set to deduce the number of major factors or sources in the data. CMB modelling may be applied to a single sample.

Spatial and Temporal Analysis

Spatial and temporal trend analysis may be used to determine the likelihood that a local source or a combination of regional influences is dominating a given geographic area. This approach assumes that there are sufficient sampling sites to generate spatial particulate concentration contours. If concentrations diurnal variations are relatively homogeneous throughout a region, distant sources are likely major contributors.

A refinement of the spatial and temporal analysis technique involves the use of multivariate analysis. For each corresponding sampling period, contour maps of species concentrations over the region are generated. The multivariate analysis simplifies the many individual maps to a few representative patterns which are linearly recombined to reproduce the observations for all time periods. Malm et al. (1994) applied this methodology to sulphur in $PM_{2.5}$ measured over a wide region in Northwest Washington. The resulting representative patterns highlighted contributing sources from the Seattle-Tacoma area (including a large power plant), local refineries and, to a lesser degree, sources within British Columbia.

Trajectory analysis is another mechanism used to identify sources in which the attendant meteorological conditions are reviewed to identify the geographic regions over which an air mass may have travelled. This is useful when examining episodes of poor visibility or high particulate loadings. Based on a knowledge of source locations, one infers the geographical areas from where particulate transport was possible. This type of analysis indicated that sources south of the border may be contributing to periods of high sulphate loading during the REVEAL study in the Lower Fraser Valley (Pryor and Steyn, 1995).

2. Summary of Selected North American Source Apportionment Studies - $PM_{2.5}$ and PM_{10}

Source apportionment studies are performed for the purposes of supporting atmospheric modelling and developing air quality management strategies associated with meeting ambient air quality objectives or standards for particulate matter. This brief review includes studies for which both fine and coarse particulate matter sources were assessed in Canada and the United States.

Canadian Urban Studies

Canadian Cities Source Apportionment (Environmental Applications Group, 1984)

The CMB model was applied to $PM_{2.5}$ and to coarse (2.5 - 15 μ m) diameter particle samples collected in seven Canadian cities (Halifax, Montreal, Ottawa, Toronto, Winnipeg, Edmonton, and Vancouver) from August, 1983 through January, 1984. While the potentially important sources were

identified for each location, source profiles were not measured and were taken from previous studies. Focusing on the PM_{2.5} source apportionments, the CMB model performance measures were not good, i.e., the model did not fit the data very well. The percent of mass accounted for ranged from 66.7% at Edmonton to 84.5% at Winnipeg. Underfitting of the total mass combined with poor model performance suggest that the source profiles used were simply not representative of the true source profiles in these urban areas. It is not possible to analytically estimate the magnitude of systematic error introduced to CMB source contributions by the use of non-representative profiles.

With the exception of Halifax, the major contribution to PM₁₀, as determined by CMB analysis came from road dust and other crustal sources. The report's authors observed that in Halifax, the fraction of the mass identified was low, but there was an indication of a substantial heavy oil contribution. The transportation sector appeared to constitute more of the coarse fraction in Edmonton and Vancouver.

The well-defined source contributions to PM₁₀ and PM_{2.5} for the seven cities are summarized in Tables 2 and 3. The CMB results indicate that 13-92% of PM₁₀ may be attributed to soil-derived materials. This was consistent with the semi-qualitative accounting approach concerning the relative abundances of soil-related elements in the chemical analysis data. This approach estimated that 25-75% of PM₁₀ would be soil-derived.

For the fine, PM_{2.5} fraction, the authors note that the contribution was dominated by secondary sulphate and transportation sources in all cities. The transportation source component was higher in the west, and the sulphate contribution was higher in the east. Table 2 summarizes the range of well-described source contributions nationally. The heavy fuel oil contribution was highest in Halifax (7.7%), in agreement with the coarse fraction results for this city. Contributions from this source occurred in all eastern cities and Vancouver, but at levels below 0.6%.

Table 2: National CMB source apportionment for fine and coarse particulate matter fractions.

Source	Contribution (%) to PM _{2.5}	Contribution (%) of PM ₁₀
Crustal	na	13 - 92
Transportation	9 - 39	2 - 19
Sulphate	14 - 48	0 - 4
Marine air	na	0 - 8
Heavy fuel oil	0.1 - 7.7	16 (Halifax only)
Steel	0.1 - 0.7	na

Note: Uncertainties for individual cities are provided in EAG(1984).

The average estimated motor vehicle contributions to PM_{2.5} for Canadian cities are noted in Table 3. These estimates are well within ranges found in other U.S. and Canadian studies.

Table 3: Average motor vehicle contributions to PM_{2.5} in Canadian cities (EAG, 1984)

City	Average motor vehicle contributions to PM _{2.5}
Halifax	9.4 - 0.5%
Montreal	26.5 - 1.1%
Ottawa	17.0 - 0.9%
Toronto	14.0 - 0.7%
Winnipeg	25.8 - 1.9%
Edmonton	35.1 - 1.8%
Vancouver	38.7 - 2.5%

Descriptive Analysis: TSP, PM₁₀, PM_{2.5}

Brook et al. (1996) presented a summary of TSP, PM₁₀, and PM_{2.5} aerosol concentrations measured at 14 urban and 4 rural Canadian sites over different time periods between 1984 and 1993. These data do not directly address the issue of source apportionment, but some descriptive insights are possible. Concentrations for PM₁₀ mass and chemical species were higher at the urban than rural sites. Crustal species concentrations were higher at the urban sites while sulfate concentrations were more similar at urban and rural locations. This probably reflects the regionality of secondary sulfate and its large sources in the eastern U.S.. The ratio of PM_{2.5}/PM₁₀ was remarkably consistent at the urban sites (0.51"0.17) and at the rural sites (0.60 - 0.65). The higher ratios at the rural sites probably reflect the relatively greater influence of regional sulfate at those sites.

Organic carbon and elemental carbon concentrations were not measured. Carbon concentrations were estimated from the differences between mass and the other measured species and ranged from 40 to 65% of PM_{2.5} with higher values at western sites. This could simply reflect the greater concentrations of regional sulfate at the eastern sites. In any case, the high carbonaceous component is consistent with the presence of motor vehicle and other combustion sources in urban areas. Sulfate was always more concentrated than nitrate but the sulfate/nitrate ratios were higher at the eastern sites. Most nitrate in urban areas is assumed to come from vehicle emissions. In western U.S. cities like Los Angeles and Phoenix, particle nitrate concentrations are higher than sulfate concentrations (Chow et al., 1991; Chow et al., 1994) in winter, but not necessarily in summer. Particulate nitrate and sulfate concentrations depend on thermodynamic factors, eg., temperature and relative humidity and also on the stoichiometry of the gas and particle phase components (Watson et al, 1994a). For example, fine particle ammonium nitrate cannot be present in the absence of ammonia and particulate ammonium nitrate formation is favoured under cold and humid conditions.

Evans Ave. Toronto CMB Source Apportionment Study (Lowenthal, 1997)

Data were provided for PM_{2.5} and coarse-particle samples collected at Evans Ave., Toronto, from the end of August to the beginning of October, 1995 (n = 23). The carbon data was taken from concurrent PM₁₅ samples. The composition of the average measured PM_{2.5} mass was 33% ammonium sulfate, 5.6% ammonium nitrate, and 58% carbon. These percentages constrain the motor vehicle contribution to primary PM_{2.5}. The motor vehicle contribution to sulfate, which is mainly secondary, could not be estimated from these data as some would be local and some regional (including U.S. sources).

Source profiles were taken from the Desert Research Institute Master Source Profile Library, a compilation of source profiles from numerous previous studies. The combination of profiles which provided the best fit to the data were:

- 1) a paved road dust profile from Long Beach, CA (PRLBPC, Watson et al., 1994b);
- 2) a 75% diesel/25% unleaded gasoline motor vehicle composite from Denver (MD75U25S, Watson et al., 1988);
- 3) a wood smoke profile from Bakersfield (BAMAJC, Chow et al., 1992);
- 4) pure sodium chloride (NACL) profile, which could represent road salt or marine aerosol;
- 5) pure ammonium sulfate (AMSUL) and ammonium nitrate (AMNIT); and
- 6) a municipal incinerator profile (DCI, recently measured, unpublished data).

The source contributions to $PM_{2.5}$, coarse, and PM_{10} ($PM_{2.5}$ plus coarse) are summarized in Table 4. The values in Table 4 represent averages of CMB results over all of the individual samples. Two uncertainties are given for each average source contribution estimate: 1) the propagated uncertainty, or root mean squared error; and 2) the standard deviation, which represents the variability of the estimated contributions over the sample series. The major contributors to $PM_{2.5}$ were motor vehicles (52%), secondary sulfate (33%) and secondary nitrate (5.6%). Road dust accounted for only 3.6% of estimated $PM_{2.5}$. Conversely, the major contributors to the coarse fraction were road and construction dust (56%), motor vehicles (32%) secondary nitrate (5.0%), secondary sulfate (3.3%), and industrial sources (2.8%). The major contributors to PM_{10} were motor vehicles (42%), road and construction dust (28%), secondary sulfate (19%), and secondary nitrate (5.3%).

The relatively large (32%) motor vehicle contribution to the coarse fraction is probably due to the manner in which elemental and organic carbon were apportioned among the fine and coarse modes. In Phoenix, only 15% of coarse mass was apportioned to motor vehicles (Chow et al., 1991). The higher motor vehicle contribution to coarse mass in Toronto suggests that too much carbon (probably organic carbon) was assigned to the coarse fraction. Thus, the motor vehicle contributions to the fine and coarse fractions could be somewhat higher and lower, respectively, than those reported in Table 4. However, without actual fine and coarse carbon concentration data, it is impossible to quantify this uncertainty.

Table 4: Average of Individual CMB Source Contributions to PM_{2.5}, Coarse Particles, and PM_{2.5} at Evans Ave., Toronto, :g/m3 and Percent of Estimated Mass (from Lowenthal, 1997).

	PM _{2.5}		Coarse (PM _{2.5} - PM ₁₀)		PM ₁₀	
	µg/m3	(%)	µg/m3	(%)	µg/m3	(%)
Measured Mass	10.8 ± 0.4	(6.1) ^a	8.1 ± 0.3	(3.7)	18.9 ± 0.5	
Estimate Source Contribution	10.6 ± 1.3	(6.1)	9.1 ± 1.3	(4.1)	19.7 ± 1.8	
Geological ¹	0.38 ± 0.09	(0.36)	5.1 ± 0.8	(2.3)	5.4 ± 0.8	28
Motor Vehicles	5.4 ± 1.2	(3.4)	2.9 ± 1.0	(1.4)	8.4 ± 1.6	42
Wood Combustion ²	0.29 ± 0.25	(0.30)	-	-	0.29 ± 0.25	1.5
Salt ³	0.06 ± 0.03	(0.13)	0.05 ± 0.19	(0.15)	0.12 ± 0.19	0.6
Incineration	0.30 ± 0.01	(0.33)	0.08 ± 0.01	(0.13)	0.38 ± 0.01	1.9
Sulphate ⁴	3.5 ± 0.2	(3.3)	0.30 ± 0.01	(0.27)	3.8 ± 0.2	19
Nitrate ⁵	0.59 ± 0.04	(0.97)	0.46 ± 0.01	(0.38)	1.0 ± 0.04	5.3
Industrial	-	-	0.25 ± 0.04	(0.22)	0.25 ± 0.04	1.3

a Average contribution " root mean squared uncertainty. Value in parentheses is the standard deviation of the source contributions.

1 Road dust plus construction

2 Vegetative burning

3 Road salt/marine salt

4 Secondary sulfate

5 Secondary nitrate

Overall, four conclusions emerged from this study:

1. The average of primary motor vehicle contributions to $PM_{2.5}$ estimated using the CMB model in various western U.S. urban areas is 44 to 23%. The average motor vehicle source contribution to $PM_{2.5}$ estimated by CMB at Evans Ave., Toronto during fall, 1995, is 51%. The Toronto estimate falls close to the center of the range of motor vehicle contributions estimated using CMB for U.S. urban areas.
2. Motor vehicle contributions to $PM_{2.5}$ in the Vancouver region estimated using CMB in independent studies done at different times (Lowenthal et al., 1996; EAG, 1984) agreed closely (~39%).
3. Distinguishing gasoline and diesel-powered vehicle contributions to ambient aerosol using CMB modelling is an active research topic. Thus, it may be premature to base emissions control strategies on this type of analysis.
4. While it is clear that motor vehicles are a significant source of primary particles and secondary nitrate in urban areas, evaluation of motor vehicle contributions to secondary sulfate in Canadian urban areas, especially those in eastern Canada, must account for regional and trans-boundary sulfur sources.

Canadian Non-urban Studies

REVEAL - Lower Fraser Valley Study on $PM_{2.5}$ (Sakiyama, 1994)

The B.C. Ministry of Environment, Lands and Parks, Air Resources Branch, conducted REVEAL as part of a larger photo-oxidants field study (Pacific 93) in the Lower Fraser Valley. Pacific 93 provided a massive information platform (time and space variations in meteorology, photochemistry and emissions) from which REVEAL data analysis could be conducted. Under REVEAL, daily collection (July - August) of $PM_{2.5}$ using IMPROVE samplers occurred at 10 sites, three of which were outside the LFV. Source apportionment analysis focuses on three sites: Pitt Meadows, Chilliwack and Clearbrook. Different source apportionment receptor modelling techniques were applied to the REVEAL data set, the results of which are presented briefly here. The broad objectives of REVEAL were to:

- (1) determine the spatial and temporal patterns of visibility-reducing aerosol concentrations,
- (2) determine estimates of light extinction budgets near the eastern end of the valley, and
- (3) apportion summertime haze constituents to general source types found in the region (both natural and anthropogenic).

Source apportionment via multivariate receptor modelling techniques (Pryor and Steyn, 1994), was used to provide an overview of valley situation. Owing to the lack of local source fingerprint data, the multivariate receptor modelling techniques provided source apportionment of ambient aerosols only to general source types. This analysis showed that:

- (1) There is evidence that a portion of the organic aerosols are the result of secondary formation.
- (2) Source apportionment of ambient aerosols indicated that contributions from secondary aerosols (sulphates and nitrates), soil/road dust and vehicle emissions dominated fine aerosol

mass during the study period. Marine and vegetation/wood burning sources were identified as relatively small contributors to PM_{2.5} loadings. The absolute mass contributions of all source types should be interpreted with care, as they are only first order estimates with unknown uncertainty resulting from limitations due to the small size of the data set.

(3) After secondary aerosols, soil/road dust was the largest contributor.

Also using multivariate receptor modelling, Pryor and Steyn (1994b) focused on the fine aerosol contributions from vegetative burning/wood processing and direct vehicle emissions for three sites: Pitt Meadows, Clearbrook, and Chilliwack. The key results are that direct motor vehicle emissions contribute an average of one-sixth of the total PM_{2.5}, and vegetation burning/wood processing contributed an average of one-fifth and one-twentieth of the total fine mass aerosol at Pitt Meadows and Clearbrook. For the Chilliwack data, although there were some hints of a vegetation contribution, this component was not resolvable. The authors caution, that these source attributions need care in interpretation as they are only first order approximations. Further, the evaluation of the data was hampered by a relatively small data set, and the lack of local source profile information.

Lowenthal et al. (1994) conducted CMB receptor modelling analysis for the REVEAL aerosol data at two LFV sites (Chilliwack and Pitt Meadows). As locally determined source fingerprint data were not available for the Lower Fraser Valley, representative source profiles were used. The CMB results for the period average concentration profile (Table 5) showed that direct motor vehicle emissions, secondary sulphate and secondary nitrate dominate the contribution to PM_{2.5} at the Chilliwack site. The crustal and marine components are relatively small (3% each). The uncertainty estimates shown in the figure. The results for Pitt Meadows showed similar contributions from marine, vegetation burning and crustal source types. The motor vehicle contribution was greater (43% vs. 34% at Chilliwack) and there was less secondary nitrate (12% vs. 27% at Chilliwack).

Table 5: PM_{2.5} chemical mass balance analysis for Chilliwack, British Columbia, 1993 (from Lowenthal et al., 1994).

PM_{2.5} Fractions during REVEAL	
Crustal	3 ± 0.5%
Vegetation Burning	8 ± 2.5%
Motor Vehicles	34 ± 8%
Marine	3 ± 0.5%
Secondary Sulphate	25 ± 1%
Secondary Nitrate	27 ± 1%

Given the available ambient and source data, it was not possible to distinguish emissions from gasoline and diesel-powered vehicles or to quantify the biogenic sources of the measured organic carbon. Source apportionment according to these source types was also conducted for each day through the sampling period. This allowed estimates of a range of contributions over the study period. For example, at Pitt Meadows vegetation burning sources can contribute up to 19% to PM_{2.5} on some days and as low as 5% on others.

Although the source apportionments using multivariate analysis and CMB yield quantitatively different results, there are similarities. Both methodologies indicate a contribution from direct

motor vehicle emissions (although the magnitude of the contribution differs) and identify vegetation burning and marine sources. The differences in source attribution are not surprising, given the lack of local source fingerprints, the relatively small data set (important in multivariate techniques) and other factors related to the fundamental assumptions associated with both methods. Both studies note that sulphates, nitrates and organic carbon make up a major fraction of PM_{2.5}. This observation is independent of the source apportionment methodology. At Chilliwack, sulphates, nitrates and organic carbon comprise 25, 27 and 39% of the PM_{2.5}, respectively.

U.S. Urban Studies

South Coast Air Basin of California (Gray et al., 1988)

CMB modelling was applied to PM₁₀ data in the South Coast Air Basin of California to determine source contributions to annual PM₁₀ mass. The chemical composition of 150 source types emitting PM₁₀ was compiled. Where necessary, source testing was performed to construct chemical signatures for source types unique to the air basin. Three of the ten basin monitoring sites are discussed here. Lennox (LX) is a coastal site near the Los Angeles International Airport. Los Angeles (LA and LA2) is a downtown site surrounded by freeways, heavy surface street traffic and industry. Rubidoux (RB), 75 km east of LA, is in a mixed urban/rural area. Tables 6 and 7 provide the source apportionment for 1986 as determined from CMB analysis. Note that the uncertainties associated with the apportionment were not provided in this study.

Table 6: PM₁₀ source apportionment for Lennox and Rubidoux, California, 1986. (Gray et al., 1988).

Lennox PM ₁₀		Rubidoux PM ₁₀	
Paved Road Dust	20%	Paved Road Dust	47%
Freeway Dust	14%	Soil Dust	2%
Marine	7%	Lime	5%
Motor Vehicle Leaded Diesel	8%	Marine	3%
Motor Vehicle Leaded Gas	1%	Motor Vehicle Leaded Diesel	5%
Secondary Carbon	16%	Motor Vehicle Leaded Gas	1%
NH ₄ SO ₄	16%	Secondary Carbon	7%
NH ₄ NO ₃	17%	NH ₄ SO ₄	7%
Unexplained	1%	NH ₄ NO ₃	14%

Table 7: PM₁₀ and PM_{2.5} source apportionment for Los Angeles, California, 1986 (Gray et al., 1988).

Los Angeles PM ₁₀		Los Angeles PM _{2.5}	
Paved Road Dust	40%	Paved Road Dust	9%
Marine	2%	Marine	1%
Motor Vehicle Leaded Diesel	9%	Motor Vehicle Leaded Diesel	16%
Motor Vehicle Leaded Gas	1%	Motor Vehicle Leaded Gas	2%
Secondary Carbon	13%	Secondary Carbon	29%
NH ₄ SO ₄	13%	NH ₄ SO ₄	20%
NH ₄ NO ₃	19%	NH ₄ NO ₃	19%
Unexplained	3%	Unexplained	4%

The paved road dust profile consists of soil-related constituents (Al, Si, Ca, Fe), while carbon

and lead are also present, indicating contribution from automobile exhaust, tire wear and motor oil deposits accumulating on road surfaces. At Lennox, no single source type dominates (paved road dust: 20%, freeway road dust: 14%). Secondary nitrate, sulphate and carbon comprise 17, 16 and 16 % of the total mass respectively. This is in contrast to Rubidoux, where paved road dust (47%) dominates and 24% of the mass is made up of secondary nitrate. For Los Angeles, the largest source of PM₁₀ was paved road dust (40%). Secondary nitrate comprised 19% of the total mass. As expected, the marine contribution is highest at Lennox (7%).

PM_{2.5} was also monitored at the Los Angeles site (Table 7). The secondary species (ammonium nitrate, ammonium sulphate and secondary organic carbon) comprise most of the total mass. The paved road dust contribution, although dominant in the PM₁₀ source attribution (40%), contributes only 9%. It is also interesting to note that the leaded diesel motor vehicle sources contribute far more than the leaded motor vehicle direct emissions. The difference in the PM₁₀ and PM_{2.5} source apportionment for an urban location is likely the result mobile sources, causing a dominance of secondary compounds in the PM_{2.5} from primary sources of NO_x, SO_x and organic compounds - all released in fossil fuel combustion. The paved road dust contribution is much smaller in the PM_{2.5} samples, indicating that the particles associated with this source are in the coarse particle size range.

Quail Roost - Houston Source Apportionment (Stevens and Pace, 1984)

Eighteen sequential fine (PM_{2.5}) and coarse (PM_{2.5} - PM₁₅) particle fraction samples were made in September 1980 in Houston, Texas. Aerosols on three different filter media were collected in order to perform a wide range of chemical analyses. Source profiles specific to Houston's major aerosol emission sources were unavailable, so source fingerprints obtained in other cities were used. The results of the CMB analysis are shown in Table 8. Uncertainties associated with the source contributions are were not given in the paper. Note that for the coarse fraction, crustal material dominates the source contribution (62%). For the fine fraction, sulphate and other cations comprise the greatest amount of PM_{2.5}. The crustal contribution in the fine fraction is considerably less (4%).

Table 8: Fine (PM_{2.5}) and coarse (PM_{2.5} - PM₁₅) source apportionment for Houston, Texas (from Stevens and Pace, 1984).

Houston Coarse (PM _{2.5} - PM ₁₅)		Houston Fine (PM _{2.5})	
Crustal	62%	Crustal	4%
Steel	1%	Steel	3%
Marine	1%	Other Metals	2%
Vehicle Exhaust	2%	Vehicle Exhaust	7%
Other Carbon	5%	Other Carbon	14%
Nitrate	3%	Sulphate + Cations	43%
Sulphate + Cations	3%	Other	27%
Other	23%		

Detroit Source Apportionment (Wolff and Korsog, 1985)

An aerosol characterization study was conducted in Detroit, Michigan, in the summer of 1981, during a severe pollution episode (poor visibility and elevated ozone). Dichotomous filter samples were collected every four hours during this period. The size ranges were defined as

coarse (PM_{2.5} to PM₁₅) and fine (PM_{2.5}). The sampling site was located in an urban area, 3 km north of the urban core. Heavy industry was farther south, and a freeway and major traffic arteries were in close proximity. Multivariate receptor modelling techniques were employed. The results are shown in Table 9. Uncertainties associated with the percent contribution are shown in the figure.

Table 9: Fine (PM_{2.5}) and coarse (PM_{2.5} - PM₁₅) source apportionment for Detroit, Michigan, 1981 (from Wolff and Korsog, 1985).

Detroit Fine (PM _{2.5})		Detroit Coarse (PM _{2.5} - PM ₁₅)	
Oil Burning	3 ± 2%	Crustal	63 ± 6%
Incineration	4 ± 2%	Iron and Steel	12 ± 3%
Motor Vehicles	5 ± 2%	Motor Vehicles	16 ± 5%
Sulphate Factor	55 ± 7%	Organic Carbon	6 ± 3%
Unidentified	33 ± 4%	Unidentified	3 ± 13%

Crustal material was the dominant source for the coarse particles, accounting for about two-thirds of the mass. Emissions that appear to be associated with the iron and steel industry represent about 12% of the mass, and particles associated with motor vehicles account for approximately 16%. The authors note that most of this contribution appeared to be resuspended road dust. For PM_{2.5}, a sulphate source, which appeared to be from coal burning, was the dominant contributor (55%). Less important sources, which account for 10% or less of the PM_{2.5}, include motor vehicles, fuel oil combustion and incineration. Most of the motor vehicle contribution was due to heavy-duty diesels, which accounted for most of the elemental carbon. An organic carbon source, which represents about 10% of PM_{2.5}, appeared to be from secondary atmospheric reactions.

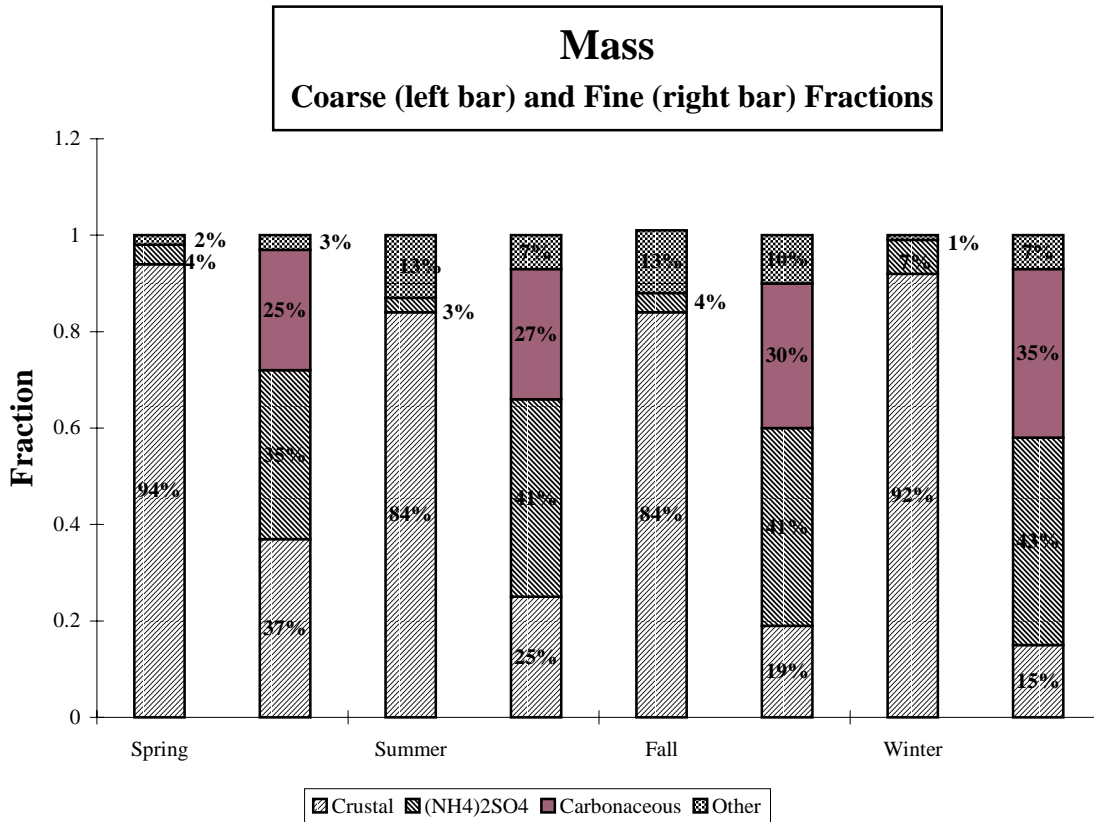
U.S. Non-urban Studies

Southwestern U.S. Visibility Study (SCENES - Vasconcelos et al., 1995)

A long-term monitoring study designed to characterize regional haze and visibility impacts in the Southwestern United States utilized seven long-term aerosol monitoring stations in Utah, Arizona and Nevada. Fine fraction (PM_{2.5}) and total particulates (here defined as PM₁₅) were monitored. Chemical analysis provided a platform for the source apportionment work based upon an eight year data record. Although no receptor modelling work was performed, the summaries provided a useful means of comparing the chemical differences between the fine and total particulate, and a preliminary identification of contributing sources.

Chemical speciation for all days averaged over all locations for the fine (PM_{2.5}) and coarse (defined here as between PM_{2.5} and PM₁₅) fractions is shown in Figure 6.1. Note that for the coarse fraction, the crustal components are dominant in all seasons. For the fine fraction, the crustal contribution is much less, with ammonium sulphate and carbonaceous particulates (organic and elemental carbon) comprising the largest fraction of the total mass. Although the authors did not delineate the sources of the sulphates and carbonaceous material, these compounds commonly result from activities related to fossil fuel combustion. The SCENES program indicates evidence of particulate matter transport to the Grand Canyon basin from Los Angeles.

Figure 1 Chemical speciation of particulates during the SCENES study in the Southwestern United States (from Vasconcelos et al., 1995).



PREVENT - Visibility and Aerosol Study in the Pacific Northwest (Malm et al., 1994

During the summer of 1990, extensive aerosol (34 sites - PM_{2.5}) and visibility monitoring was conducted over a wide area in the Pacific Northwest. The objective was to attribute visibility degradation to various source types. Supporting data (emissions, meteorology, ambient pollutant concentrations, and atmospheric visibility data) were collected to facilitate the source apportionment analysis which focused on three sites: two in Mount Ranier National Park and one in Cascades National Park.

Sulphates, nitrates and organics comprised the largest fraction of the fine particulate matter, as illustrated in Table 10. Overall, organics comprised the largest fraction (about 50%), light-absorbing particles (elemental carbon) contributed 10%. Sulphates, presumed to be in the form of ammonium sulphate, contributed about 30% of PM_{2.5}, and nitrates contributed approximately 5%.

Table 10: PM_{2.5} Chemical mass balance analysis for Mount Rainier and Cascades National Park during PREVENT (from Malm et al., 1994)

PM_{2.5} Fraction	
Soil	7%
Sea Salt	1%
Organics	47%
Nitrates	5%
Sulphate	30%
Light Absorbing	10%

Quantitative apportionment of sulphates, complicated by the atmospheric transport, dispersion and chemistry of the region, showed strong source - receptor trends indicating that elevated sulphur in particulate matter was attributable to a large power plant and the Seattle-Tacoma urban area. Nitrate sources appeared to be related to pulp and paper mill or lime kiln activities, with contributions from coal-fired power plants, transportation and burning. The majority of organic carbon was associated with transportation-related emissions in the Seattle - Tacoma urban area, and most of the elemental carbon was associated with forest fire events.

Canadian National Emission Inventory Information - 1990 (from Altech, 1997)

A national PM_{10/2.5} emission inventory for 1990 has been prepared by Environment Canada that includes primary particulate emissions for all anthropogenic source sectors as well as meteorologically-dependent open sources and biogenic emissions. This inventory was also based on applying U.S. emission source fine particulate profile information to the estimated Canadian total particulate emissions.

A ranking of the major types of primary fine particle emission sources in Canada is shown in Table 11. The tabulated emission quantities have been extracted from recently compiled inventory data, supplied by Environment Canada, and includes contributions of 17 traditional anthropogenic source sectors which account for ~90% of the total PM_{2.5} emissions (i.e., excluding open and biogenic sources) with ranking in descending order of contribution. Thus, significant PM_{10/2.5} emission contributions might be expected for sources such as:

- pulp and paper manufacturing
- coal industry
- mining and rock quarrying
- heavy-duty diesel vehicles
- off-road use of diesel
- wood industries
- railroads
- electric power generation
- iron and steel production
- tire wear
- wood waste incineration

However, it should also be noted that other types of non-traditional open contributing sources, which have been included in the Environment Canada emission totals, are known to release major and sometimes overwhelming quantities of fine particulate matter such as:

- paved and unpaved road dust
- forest fires
- prescribed burning
- agricultural tilling
- construction activities
- mine tailings

In several instances, emissions from these major open-source and biogenic sectors are often the least well quantified in emission inventory compilations and may fluctuate extensively between years and within Canadian regions.

Table 11: Preliminary Estimates of Major Sources of Fine Particle Emissions in Canada (1990)^a
[Table to be revised by Pollution Data Branch in collaboration with provinces]

Source: Altech, 1997. Smog Plan National Tracking System

Sector Description	Total Particulate (kt)	% Total	PM ₁₀ (kt)	% Total	PM _{2.5} (kt)	% Total
Residential Fuelwood Combustion	113.0	8.8	113.0	14.8	107.3	20.7
Pulp and Paper Industry	109.4	8.5	80.0	10.5	64.3	12.4
1Coal Industry	70.2	5.5	64.6	8.4	56.7	10.9
Mining and Rock Quarrying	124.0	9.7	66.2	8.9	38.1	7.3
Heavy Duty Diesel Vehicles	31.3	2.4	31.3	4.1	28.8	5.5
Off Road Use of Diesel	24.7	1.9	24.7	3.2	22.7	4.4
Wood Industry	76.7	6.0	39.3	5.1	20.7	4.0
Other Industries (not classified)	113.8	8.9	42.6	5.6	20.4	3.9
Railroads	22.0	1.7	22.0	2.9	20.2	3.9
Electric Power Generation	154.0	12.0	45.7	6.0	17.7	3.4
Iron and Steel Production	50.0	3.9	28.2	3.7	14.3	2.8
Tire Wear	37.0	2.9	20.4	2.7	14.4	2.8
Wood Waste Incineration	33.9	2.6	18.7	2.4	13.6	2.6
Non-Ferrous Mining & Smelting	16.0	1.2	11.0	1.4	8.3	1.6
Asbestos Production	23.9	1.9	13.0	1.7	7.9	1.5
Aluminum Production	19.8	1.5	11.2	1.5	7.5	1.4
Iron Ore Mining & Beneficiation	54.1	4.2	38.0	5.0	6.4	1.2
Pesticide & Fertilizer Application	20.7	1.6	10.4	1.4	4.7	0.9
Total Canadian Emissions (excluding open and Biogenic Source Categories)	1281.4 ^b	85.4	764.3 ^b	89.0	519.7 ^b	91.2

NOTES

- A Ranked based on PM_{2.5} totals from emission information provided by Environment Canada
 B Total emissions include totals from all traditional emission categories not just major listed sectors (excluding open and biogenic emissions)

Major Open Source and Biogenic Emissions	Total Particulate (kt)	PM ₁₀ (kt)	PM _{2.5} (kt)
Dust from Unpaved Roads	34,246.7	23,287.1	10,273.7
Dust from Paved Roads	1,197.2	526.8	299.3
Forest Fires	293.1	290.2	263.8
Prescribed Burning	183.4	181.6	163.1
Agricultural Operations	4,985.0	398.8	99.7
Construction	23,831.8	6,434.6	71.5
Mine Tailings	45.4	24.7	15.0
Total Canadian Emissions from all Sources	66,093.5	31,910.5	11,708.4

Summary

The varied sources of primary and secondary particulate matter complicate the source apportionment analysis. This has led to the application of various techniques of varying degrees of complexity, suitable for a range of available data. In this brief overview, there are common features associated with coarse and fine source apportionment results.

- 1) In the coarse fraction, which in some studies has been defined as PM_{10} or the range from $PM_{2.5}$ to PM_{15} , there is a predominance of crustal material (or road dust, which has components of crustal material). The exact contribution of coarse depends on the location, the season, etc. The contribution of crustal sources greatly diminishes in $PM_{2.5}$ samples, generally less than 10%. In the arid regions of the Southwestern United States, the SCENES data indicate a crustal contribution of only 15 - 37% of the total.
- 2) The greatest contributions to $PM_{2.5}$ come from organic compounds and secondary sulphates and nitrates. In urban areas, the sources of these compounds and their precursor gases (SO_x , NO_x , and $VOC=s$) are typically from combustion processes - motor vehicles, industrial processes and vegetative burning. Even remote areas may be impacted by these sources.
- 3) The early estimates from the EAG source apportionments for all of the Canadian cities are consistent with these more recent analysis - estimated motor vehicle source contribution ranging from 9 - 39% of $PM_{2.5}$. REVEAL (Lowenthal et al., 1996) and EAG (EAG, 1984) CMB source motor vehicle source apportionments for the Vancouver area were highly consistent, about 40% of $PM_{2.5}$. REVEAL (Pryor and Steyn, 1994) indicates motor vehicle contribution (15 - 20%) to $PM_{2.5}$. In Toronto, the motor vehicle contribution is approximately 50% (Lowenthal, 1997).

The 1990 Canadian national emission inventory for PM_{10} and $PM_{2.5}$ includes primary emissions only, and is based upon estimates using U.S. source profiles on the Canadian Total Suspended Particulate emission information. It is anticipated that the 1995 emission inventory will refine these estimates based upon available Canadian source profile information.

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