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CANADIAN SMOG SCIENCE ASSESSMENT

Highlights and Key Messages

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Canadian Smog Science Assessment – Highlights and Key Messages

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Preamble

This "Highlights and Key Messages" document, a distillation of the material contained in the *Canadian Smog Science Assessment*, was co-authored by Environment Canada and Health Canada and presents a summary of the Assessment's key findings[1]. This initiative is the most recent effort[2] by the Government of Canada to assess the adverse effects of smog on Canadians and their environment, quantify emissions of smog-forming pollutants, describe their behaviour in the atmosphere at a regional scale, and report on recent and projected levels of smog in the air we breathe. It represents the first time all the scientific material related to smog in Canada has been addressed together, combining characterization of both particulate matter and ground-level ozone in a single document.

This summary document is intended to provide credible and relevant scientific information to guide current policy and regulatory decision-making in Canada. With this purpose in mind, the document has been arranged in a logical manner to inform risk management decision-making.

1. The *Canadian Smog Science Assessment* will be available in 2012.
2. Previous assessments include:
   Federal-Provincial Working Group on Air Quality Objectives and Guidelines, 1999. *National ambient air quality objectives for ground-level ozone: Science assessment document.* Health Canada and Environment Canada, Ottawa.
   Federal-Provincial Working Group on Air Quality Objectives and Guidelines, 1999. *National ambient air quality objectives for particulate matter, Part 1: Science assessment document.* Health Canada and Environment Canada, Ottawa.
   Joint Action Implementation Coordinating Committee, 2005. *An Update in Support of the Canada-wide Standards for Particulate Matter and Ozone: Part A – 2003 science review.* Report to the Canadian Council of Ministers of the Environment [Available at www.ccme.ca].

Contents

Introduction to Smog

Air pollution has received significant attention over the past several decades from both the scientific and the environmental risk management communities in Canada and internationally. This is especially true of smog's two major components: particulate matter (PM) and ground-level ozone (O_3). The primary reason for this attention has been the scientific literature indicating the extensive impacts exerted by these pollutants on human health. Although they can also impact the environment, these impacts are less critical at the levels of smog currently experienced in Canada. In the last decade alone, at least 4500 new primary scientific publications on the health effects of PM and O_3 have been published, with 1200 of these in the 2002–2006 period covered by this review.

The World Health Organization (WHO) periodically undertakes an analysis titled ***Global Health Risks***. In the most recent 2009 analysis[3], the WHO lists the most important quantifiable human health risk factors. While these risk factors vary considerably between developing and developed countries, urban air pollution ranks 13th in the top 20. In developed countries, urban air pollution is eighth among the top 10 risk factors, and is the most important population health risk factor that cannot be significantly reduced by changes to personal behaviour, arguably making it the most important risk factor for which the primary control mechanism is regulatory in nature.

In Canada, efforts have been made over the last decade to quantify the health burden of smog, facilitated by the evolving health effects literature. While a number of different endpoints have been associated with air pollution, mortality has received the most attention due to the availability of data for analysis and the severity of the endpoint. A recent 2008 analysis by the Canadian Medical Association[4] estimated that some 21 000 deaths per year were associated with air pollution in Canada, along with tens of thousands of hospital visits, hundreds of thousands of asthma and respiratory symptom days and millions of minor illnesses and restricted-activity days.

There is evidence that all Canadians are at some risk from the effects of ongoing exposure to air pollution, especially PM. However, much of the evidence indicates a heightened level of sensitivity for those with cardiovascular or respiratory disease. This is important due to the prevalence of such diseases in the Canadian population where cardiovascular causes account for 30% of mortality and respiratory disease for 10%. Asthma, which is exacerbated by both PM and O_3, has been diagnosed in at least 8% of the Canadian population over the age of 12. Diabetes, a common and increasing disease currently affecting one in ten Canadians, is also adversely affected by smog.

Not only does smog affect a broad swath of the population, but adverse health effects occur across the full range of ambient levels experienced in Canada, even very low levels. For all practical purposes, both PM and O_3 need to be treated as having no safe level (i.e., on a population basis, there is no lower threshold at which there are no effects). This is likely due to the ubiquitous exposure to smog and the common nature of the illnesses and conditions affected by air pollution. One of the ramifications of this is that, while smog days are the most noticeable, the cumulative effects of smog on what are normally regarded as "good"

3. World Health Organization, 2009. *Global health risks: mortality and burden of disease attributable to selected major risks.* WHO Press, Geneva.
4. Canadian Medical Association, 2008. *No breathing room: National illness costs of air pollution.* CMA, Ottawa.

days are significant and, in total, exceed the total effects on "smog" days. For example, a 2001 analysis for the City of Toronto[5] found that over 90% of the health impacts of air pollution in that city occurred on days classified as "good" under the province's Air Quality Index.

SMOG FACTS

Smog **is a general term used to describe poor air quality that is often associated with reduced visibility. Although smog is a mixture of pollutants, the *Canadian Smog Science Assessment* focuses on its two main components; particulate matter and ground-level ozone. *Particulate matter* (PM) is a complex mixture of small airborne liquid and solid particles that are characterized by their size. PM can be emitted directly (primary PM) or formed in the atmosphere (secondary PM) by reactions involving nitrogen oxides (NO_x), sulphur dioxide (SO_2), volatile organic compounds (VOCs), and ammonia (NH_3). *Ground-level ozone* (O_3) is a gas that forms in the atmosphere through reactions between NO_x and VOCs in the presence of sunlight.**

Primary PM, and the gaseous precursors to O_3 and secondary PM originate from both natural (e.g., forest fires, windblown soil, sea salt spray, volcanic dust) and anthropogenic sources (e.g., fossil fuel burning, various industrial processes, agricultural activity, road dust), both within and outside Canada.

The PM of most concern is that which can penetrate into the human lungs and is classified by size as coarse PM (between 10µm and 2.5µm in diameter), fine PM (less than 2.5µm) and ultra-fine PM (less than 0.1µm). Since smaller particles penetrate deeper into the lungs, the *Canadian Smog Science Assessment* focuses on $PM_{2.5}$ but, where appropriate, discusses results oriented to other sizes as well.

While human health effects are regarded as the primary impact of concern, smog also exerts considerable adverse effects on a range of environmental factors, most notably in relation to acid deposition (or acid rain) and the effects of O_3 on vegetation. Visibility is also degraded by air pollution (primarily PM) and this has an aesthetic impact, especially in areas where natural vistas are culturally and economically important.

Because of its impacts, smog pollutants have been the subject of significant attention from policy makers and risk managers for several decades. Measures to address acid rain in eastern Canada and the eastern U.S. have delivered major reductions in sulphur dioxide and nitrogen oxides (two of the precursors of PM and O_3). This has also led to some improvement in levels of $PM_{2.5}$ and, to a lesser extent, levels of O_3. Canada-wide Standards for $PM_{2.5}$ and O_3 were established in June 2000 in order to reduce the risks to human health and the environment associated with these pollutants.

5. Toronto Public Health, 2001. *Toronto Air Quality Index: Health Links Analysis.* Toronto, Ontario.

THE CANADA-WIDE STANDARDS FOR $PM_{2.5}$ AND O_3

The Canada-wide Standards for $PM_{2.5}$ and O_3 are numeric targets published in June 2000 by the Canadian Council of Ministers of the Environment (CCME) in order to minimize the exposure risk to human health and the environment. These targets were based on the scientific understanding of the issue at the time and represented a balance between the desire to achieve the best health and environmental protection possible in the near-term and the feasibility and costs of reducing the emissions that contribute to PM and ozone levels in ambient air. The numeric target for $PM_{2.5}$ is 30 µg m^{-3} (24-hour average) to be achieved by 2010 based on the 98th percentile annual value averaged over three consecutive years. The numeric target for O_3 is 65 parts per billion (8-hour average) to be achieved by 2010 based on the fourth highest annual value averaged over three consecutive years.

Effects on Human Health

Mortality

The air health effects literature which examines linkages with mortality consists of two general study types: those that examine the relationship with long-term, or chronic, exposure (usually in terms of multi-year exposure); and those that examine short-term, or acute, exposure (in terms of same-day to a few days of exposure) and associated changes in mortality rates.

Long-term studies are usually based on cohorts of individuals who have been enrolled in a database for the purposes of studying their health relationships with a variety of risk factors (e.g., smoking, lifestyle). The focus on these studies is on the individual and factors that alter individual risk that can be examined to provide information on group risk. Short term studies generally utilize administrative records collected for various purposes such as billing and health care tracking to examine population-level reactions to risk factors. In both of these types of study, exposure to air pollution is estimated as the concentrations measured at air monitoring stations that have been established to monitor compliance with ambient air quality targets.

The cohort studies on which chronic effects are based have the advantage of containing extensive information on individual risk factors that can be examined in addition to the risk of air pollution over time. Additionally, because of their long-term (or "longitudinal") structure, these studies make it possible to estimate the life shortening impact of an individual risk factor such as air pollution. However, longitudinal cohorts are relatively rare due to their significant financial and logistical demands. Additionally, because many of the outcomes associated with air pollution are also associated with other risk factors (e.g., cardiovascular disease, smoking and diet), extensive analyses are required to distinguish air pollution-related effects.

Acute studies are generally more common since the databases on which they depend are usually maintained by public health agencies and utilize standardized mortality classifications and collection methods over entire populations. In addition, because the time scale of these studies is comparatively short (in that they examine changes in mortality on a day-to-day basis), most other risk factors generally associated with mortality are not correlated with air pollution and do not need to be accounted for in these analyses. The most significant factors which must be dealt with in short-term studies, primarily weather variables, are available on a scale amenable to analysis, though this requires extensive statistical modelling. Unlike long-term studies, however, these analyses provide little additional information on risk factors and other factors (e.g., socio-economic status factors) which modify the air pollution relationship and cannot be assessed directly.

Both short- and long-term studies provide information on the magnitude of risk from air pollution exposure. Their results are likely additive (i.e., the mortalities observed in one type of study are not accounted for in the other) and can provide information on susceptibility by examining all-cause, respiratory, cardiovascular and other causes of death, as well as age-specific analyses. However, long-term cohort studies are regarded as more robust and informative because they can account for other risk factors, and the inherent follow-up time period which allows examination of important issues such as length of life lost.

Subsequent to the 1999 Canadian science assessments[6] for particulate matter (PM) and ozone (O_3), it was determined that the availability of Canadian studies examining short-term relationships with mortality, combined with questions concerning the analysis of the key U.S. chronic-exposure studies[7] and their applicability to Canadians, favoured an emphasis on short-term associations. Thus the focus for both PM and O_3 was on short-term exposure and resulted in the derivation of 8-hour (for O_3) and 24-hour (for $PM_{2.5}$) Canada-wide Standards. Work undertaken in the last decade indicates that, while short-term standards continue to reflect a robust body of evidence, there is now evidence which implicates long-term exposure in premature mortality and supplies a clear rationale for the derivation of a long-term (annual) air quality standard for Canada.

Chronic Exposure Mortality

The major effort examining relationships between chronic air pollution exposure and mortality has been in the re-analysis of results from several cohorts, principally the American Cancer Society (ACS) Cohort, and the Harvard Six City Cohort. While these studies have been available since the mid-1990s, some issues related to analysis raised questions about interpretation of their results. Subsequent re-analyses have dealt with these issues and, in addition, have added a number of new insights to the overall understanding of effects. As well, several additional, though smaller, cohorts have also been analyzed. All of these analyses show positive, usually significant, relationships between long-term exposure to air pollution and premature mortality, including all-cause mortality as well as several cardiovascular and respiratory sub-categories.

One of the important results of these analyses is the predominant adverse health effect of fine PM, with coarse PM and other co-pollutants (such as nitrogen dioxide and carbon monoxide) showing little effect.

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6. Federal-Provincial Working Group on Air Quality Objectives and Guidelines, op. cit.
7. American Cancer Society Study: Pope III, C.A. 1995. Particulate air pollution as a predictor of mortality in a prospective study of U.S. adults. *Am. J. Respir. Crit. Care Med.* 151: 669–74.
   Harvard Six City Study: Dockery, D.W. et al. 1993. An association between air pollution and mortality in six U.S. cities. *New England Journal of Medicine* 329: 1753–1759.

Sulphur dioxide (SO_2) is, in some cases, also significantly related to adverse effects, although it has been argued that SO_2 is as an indicator of PM from specific sources. Cardiac outcomes have also been better characterized; additionally, while respiratory mechanisms appear to play a role, it has become increasingly clear that cardiovascular effects underlie the majority of these air-pollution related mortalities.

U.S. studies of chronic exposure mortality have been used worldwide by agencies (WHO, EU, etc.) to characterize PM-related mortality. Canadian cohorts, until recently, had not been available. However, two cohorts encompassing millions of Canadians have now been developed and are undergoing analysis for relationships with chronic exposure to several air pollutants including PM and O_3. The advantage of these cohorts is that they will provide for examination of the relationships in the relatively low pollution levels of the Canadian environment, have a large amount of information on susceptibility not available in other cohorts, can be used in relation to a number of Canadian-specific risk factors, and will provide analysis related to the initiation of disease. First results are expected in 2011.

While other cohorts have been analyzed, the ACS and Six City studies remain the most prominent and provide the most reliable risk estimates due principally to their design (e.g., large size, based on the general population). In these and other studies, the newer analyses have confirmed previous results but have also provided more precise evidence of the impact of PM on susceptible subgroups, such as those with ischaemic heart disease. The ACS cohort, due to its larger size and extended follow-up, is generally used in estimating risk; however the Six City study is also used, and observed even higher risk estimates than in the ACS study. Both of these studies have been analyzed to estimate the length of life lost due to air pollutant exposure and, while there is some variation, the estimates are substantial, ranging from several months to two years. Additionally, the ACS cohort highlighted a significant link to lung cancer mortality, though the results do not definitively establish an association with the initiation of lung cancers. A few recent Canadian cohort studies, though with less statistical power due to smaller sample size and issues related to study design, also found significant mortality risk, with some results indicating that socio-economic factors may play a role in the exact nature of the effects (i.e., greater effects for lower socio-economic status).

While outside the review period of this assessment, a 2009 analysis of the ACS cohort is potentially of considerable importance and is, therefore, mentioned here. Among other issues, it used additional years of data to assess the relationship of O_3 and PM with respiratory and cardiac causes of death. A primary finding was that, unlike previous ACS analyses, a significant relationship was found between long-term O_3 exposure and premature mortality. Additionally, this relationship appeared to be confined to respiratory causes of mortality (PM was primarily associated with cardiac causes), was a function of summer time average (PM effects were a function of annual averages), and, the results appeared to suggest a threshold for the effect (unlike PM which exhibited no apparent threshold). This O_3 finding may be due to improvement in analytical techniques, but is more likely due to the additional years of data, which improve the statistical power of the study (respiratory mortality is much less common than cardiac mortality, and thus it can be more difficult to observe these effects).

Overall, the database on chronic exposure mortality has been greatly enhanced since 1999, with new analyses and re-analysis of existing cohort data indicating a significant impact of fine PM on public health.

Fine PM continues to be the dominant factor in explaining the long-term impacts of air pollution on premature mortality, and provides a strong scientific rationale for the establishment of a long-term ambient air quality target. Though little information was previously available to indicate that chronic exposure to O_3 was related to mortality, very recent analysis of the ACS cohort study indicates the possible importance of O_3 in explaining respiratory mortality, though these findings need confirmation, and additional insight is needed to interpret this result in the Canadian context.

Acute Exposure Mortality

In the 1999 Canadian science assessment documents, it was concluded that there was substantial evidence of an association between short-term exposures to increased O_3 and fine PM and premature mortality. Since that time, a number of single-city and multi-city studies, as well as meta-analyses of studies, have become available.

The majority of these studies (both old and new) demonstrate positive and usually significant effects of PM on mortality. While these studies place an emphasis on the results of fine particulates, the coarse fraction of PM ($PM_{2.5-10}$) may be implicated to some degree.

While single-city studies are more common, multi-city studies are generally regarded as more robust. Multi-city studies often have greater variability in response (due to the wide variety of inter-city conditions incorporated), but they can be superior to single-city studies since they examine a greater variety of conditions, do not suffer from publication bias (the possibility that negative single-city studies would not get published) and having greater statistical power due to the larger sample size. This provides the ability to examine more specific age and disease categories than would otherwise be possible. Meta-analysis combines the results of multiple (usually single-city) studies.

However, the risk estimates are almost always higher for fine particles, and especially in cases where studies have examined specific causes of death (i.e., respiratory and cardiovascular). In many studies, more specific mortality categories have been examined (e.g., stroke, myocardial infarction, ischaemic heart disease, chronic heart failure, pneumonia, and chronic obstructive pulmonary disease) and, in virtually all these cases, the risk is greater where there is pre-existing disease. Lags between increased exposure and resultant mortality are quite short, usually in the one day range, although longer multi-day lag periods indicate a prolonged effect with larger overall risk than for any single-day lag structure.

In the Canadian multi-city study, the effects of fine PM were similar to other analyses (both single and multi-city); however the study also noted a significant impact of nitrogen dioxide (NO_2) and O_3. In a large U.S. multi-city study (NMMAPS[8]) PM was the primary pollutant predicting mortality. Due to the large size of the NMMAPS study a number of important sub-issues could be examined. Among other findings, it was determined that weather variables could not explain the associations between air pollution and acute exposure mortality; effects were not confined to a single season, but appeared relatively uniform year-round. The statistical methodologies [i.e., the use of generalized additive models (GAM)] that had called into question earlier single-city results were not a significant issue. Model specifications (the exact manner

8. National Morbidity, Mortality and Air Pollution Study.

in which the data are statistically treated), while modifying results somewhat, did not have significant impacts on the overall results. While PM was the major predictor of mortality in the NMMAPS study, O_3 was also implicated as a significant risk factor, with the original study finding a positive and significant warm season effect and a positive, non-significant year-round effect. A follow-up analysis with more years of data reported similar effects for summer and a significant effect in the all-year analysis (possibly reflecting the increased statistical power of the larger dataset). As was the case for other studies, the NMMAPs analysis also indicated that: risk estimates for cardiopulmonary causes were greater than for all-cause mortality; O_3 results were not affected by the inclusion of co-pollutants; and the highest relative risks were found for the shortest (0 and 1 day) lags, though there was evidence that elevated risks occurred for several days beyond this.

A major European multi-city study[9] found results similar to those in North America, including greater overall impacts in the warm season, as well as greater risks attached to the categories of cardiovascular and respiratory mortality. This and other studies also examined the importance of specific O_3 concentration averaging times (e.g., 1-hour versus 8-hour versus 24-hour) and, while shorter exposure times resulted in somewhat higher risk estimates, the differences were not statistically significant.

Single-city studies are generally consistent in finding significant relationships for premature mortality with both acute PM and O_3 exposures. For O_3, these types of studies reinforce a more pronounced effect in the warm season, while for PM, risks appear to be relatively consistent year-round.

Many of the short-term studies have examined the impact of adjusting for co-pollutant exposures on pollutant-mortality associations. For particulate matter, the risk estimates are fairly robust to adjustment for other pollutants. While they are sometimes reduced in models that include gaseous pollutants (particularly NO_2, which is often highly correlated with PM), they remain positive and most often statistically significant.

Overall and with few exceptions, the O_3 effect on premature mortality was not greatly influenced by particulate matter or other pollutants. Because O_3 levels and temperature can be highly correlated, a number of studies have examined different ambient air temperature adjustments in their mortality models to determine if the O_3 effects were sensitive to this variable. Results indicate that the methods used in epidemiological studies to account for temperature are adequate to account for its effects, and O_3 and PM mortality relationships are not significantly affected by air temperature.

A number of population health studies have examined the phenomenon of thresholds for mortality effects of O_3 and PM. Approaches have included the fitting of alternative models, and segregating high from low pollutant days (usually defined as above and below an air quality standard) to determine if risk estimates differ. The shape of the short-term PM-mortality dose-response curve has been examined in several studies. In each instance, the analysis revealed a quasi-linear association, with no apparent threshold. For O_3, most of the available studies also fail to support the existence of a “no effect” level.

The relative lack of data at low exposure levels, the potential for exposure misclassification, and the inherent heterogeneity of human populations imposes some limits on the interpretation of the form of the exposure-mortality relationship, especially at low concentrations. However, an NMMAPS analysis

9. Air Pollution and Health: A European Approach.

specifically for O_3 on this issue could find no evidence that risk was reduced at the lowest concentrations and concluded that a threshold, should it exist, could only occur at levels well below background. This is similar to analyses conducted for the 1999 Canadian science assessment documents that indicated that the associations with mortality and hospital admissions for both O_3 and PM existed at the lowest ambient levels that could be examined. While it is very likely that effect thresholds exist for individuals, the presence of very large "subpopulations" with pre-existing cardio-respiratory disease, the large inter-individual variability in effects exhibited in controlled exposure studies, the existence of genetically-based susceptibility traits, and other factors indicate that thresholds for population level effects may not exist, or exist only at very low levels.

The issue of mortality displacement (the displacement of a date of death by only a few days) due to exposure to air pollutants has been examined in several acute exposure mortality studies. While difficult to study and interpret, those studies that examined the issue (usually by extending the analysis period over several days to a few weeks) could find little evidence that the date of death had been brought forward by only short time periods. This is important since a change by only a few days would have little public health significance. These results support the hypothesis that exposure to air pollution can significantly change date of death, and that these deaths were not occurring in frail individuals who were already dying and happened to die a few days earlier. Rather, short-term exposure to air pollution appears to advance deaths of susceptible people by weeks or months.

Intervention Studies

There have been a significant number of opportunities, especially over the last 10 years, to examine the public health benefits of situations where major decreases in air pollution result from intended or unintended emission reductions. For mortality, there were three cases of dramatic reductions in PM air pollution for long enough periods that permitted the study of associated health benefits. In each of these cases (a strike at a steel plant in the Utah Valley; a ban on coal use for home heating in Dublin; and significant improvement in fuel quality in Hong Kong), ambient PM pollution was reduced precipitously and persistently. Subsequent analysis indicated a concomitant reduction in mortality (and other adverse effects). Other studies have revealed that more incremental actions directed at a broader range of sources, such as those taken over the last 30 years or so in North America and in Europe, also reduced PM levels and resulted in attendant decreases in cardiovascular and respiratory mortality, and in children's respiratory symptoms. While the results of such studies have some limitations due to analytic constraints, they can provide support for observations made in the standard epidemiological analyses discussed previously.

Conclusions

Overall, newer studies have confirmed earlier results and have also increased understanding of the impact of PM and O_3 on premature mortality. Of particular importance is the confirmation of the impacts of long-term exposure to PM and the direction this gives to the adoption of an ambient target to reflect this relationship (i.e., an annual standard). While daily peaks in air pollution are important and indicate that short-term (i.e., 24-hours or less) standards are warranted, the long-term exposure findings indicate an effect significantly larger than the short-term. Since the risk management actions to address long-term exposure may be different than those for the short-term, standards that address both are warranted. The findings for cardiovascular outcomes, including specific endpoints such as ischaemic heart disease,

indicate that the respiratory system, while being the first point of attack, is not the only system subject to the adverse effects of air pollution. In addition, many of the results indicate that in addition to exacerbating existing disease, PM is implicated in the development of cardiac disease.

Development of disease is regarded as more significant than exacerbation of existing disease since it implies the development of new cases and/or the progression of the disease to more advanced states. This indicates that current methods of valuing adverse effects of air pollution produce underestimates. More recent findings on the relationship of respiratory mortality with O_3 indicates the potential for a future regulatory approach on O_3 that incorporates a chronic exposure-related standard (possibly warm-season average), though more analysis appears warranted before this is formally considered. Newer information and the overall database also confirm the need to retain short-term average standards (24-hour for $PM_{2.5}$, and 8-hour for O_3). The scientific database also continues to indicate that mortality effects occur throughout the continuum of exposure experienced by Canadians, and that a population, non-threshold approach is the most appropriate interpretation of the data at this time.

Morbidity

Chronic Exposure Morbidity

The focus for chronic exposure morbidity continues to be cardiovascular and respiratory disease endpoints, though there are few new studies and to date the most important findings have been respiratory in nature. Previous studies had revealed significant impacts of O_3 on lung health in children, but these results were largely based on high exposure levels not experienced in Canada. Results at more relevant O_3 concentrations were equivocal, and overall there are relatively few studies examining chronic exposure and morbidity outcomes.

A significant research effort is ongoing, especially in relation to understanding the potential for air pollution to cause chronic disease, or contribute to the progression of chronic disease. Much of this work focuses on cardiovascular outcomes due to the general importance of this in modern society. This newer work provides indications of an involvement of air pollution (especially fine PM) in the progression of several important adverse cardiac effects.

The most important new study in this area of investigation is the California Children's Health Study. This well-designed prospective cohort study examined a range of air pollutants and adverse outcomes over a wide range of concentrations. While fine PM (and elemental carbon) was associated with most outcomes examined (e.g., lung growth, symptoms, asthma), so too were other co-occurring pollutants such as acid vapour and nitrogen dioxide. For these pollutants, lung function, growth and development were all adversely impacted, though there was no clear relationship with one pollutant versus the others. Lung growth and development in these cases were affected through age 18, when lungs are mature and a point at which any damage is not likely to be repaired, thus compromising lung health later in life. It was also found that children who moved at earlier ages from areas of high PM levels to areas of low PM levels exhibited significant lung growth recovery, and vice versa. Most of the outcomes were not associated with ambient O_3 levels. However, two particular adverse effects, school absenteeism for respiratory complaints and development of asthma, were associated with exposure to higher levels of O_3, but not any other pollutant. In these analyses, asthma development was manifested only in groups of children in the

The California Children's Health Study, which began in 1992, is a large, long-term study of the effects of chronic air pollution exposures on the health of children living in Southern California. Children may be more affected by air pollution than adults because their lungs and bodies are still developing. Children are also exposed to more air pollution than adults since they breathe faster and spend more time outdoors in strenuous activities.

About 5500 children in 12 communities were enrolled in the study; two-thirds of them fourth-graders. Data on the children's health, their exposures to air pollution and many factors that affected their responses to air pollution were gathered annually until they graduated from high school. Research results first appeared in 2000, with new publications continuing to be made available as data is analyzed.

highest-O_3 (greater than levels experienced in Canada) communities and engaged in a high level of outdoor activity. School absenteeism was manifested across a broad range of concentrations, including ones relevant to Canadian air quality.

Several studies in Europe and North America (including the lower mainland of British Columbia) have found significant relationships between exposure to PM (and in some cases specific PM sources) and adverse effects on the auditory system of children. While often characterized as being of less importance than some other outcomes, middle ear infections (the major reason for the prescription of antibiotics in younger age groups) and related outcomes were demonstrated to be associated with existing levels of PM. These are significant for population health because they are, in some cases, the most common cause of hospitalization for younger age groups.

Acute Exposure Morbidity—Hospital Admissions and Emergency Room Visits

Previous studies of hospital admissions and emergency room visits (ERVs) indicated a relatively robust relationship to ambient O_3 and PM levels (independent of each other), with most studies reaching statistical significance.

There are fewer examples of this type of study than for premature mortality, largely because centralized databases of hospital admissions are far less common and there are challenges in using the data. Overall, newer results from these types of studies are consistent with earlier ones. In general, the findings indicate significant relationships between unplanned hospital admissions and both particulate matter (generally PM_{10}) and O_3. Because the statistical power (i.e., sample size) in these studies tends to be less than for mortality, there is less opportunity to segregate specific causes; however, in studies large enough for specific analysis, the risk and level of statistical significance was greater for specific disease categories (especially cardio-respiratory), and for the elderly age-class (cardio-respiratory disease is far more prevalent after the age of 55). In addition, those suffering from asthma, especially children, appeared to be more susceptible to the effects of PM and O_3 than the general population.

As was the case in mortality studies, there is some evidence of seasonal patterns in the relationship between hospital visits and O_3 exposure. While there is, in general, a consistent relationship between O_3 levels and adverse health effects, the relationship becomes especially strong during the warm season (possibly due to difficulties dealing with between-season variance or the greater likelihood of exposure

during the warm season). This relationship holds true for ERV studies, especially for asthma. Associations between PM exposure and hospital visits do not show seasonality, with seasonally restricted analyses providing results similar to those for the full year.

Examination of the effect of co-pollutants provides evidence that the effects of O_3 on ERVs and hospital admissions are not significantly affected by the inclusion of other pollutants potentially associated with acute exposure morbidity. The case was not as clear for PM, with some studies (including Canadian ones) finding that results for PM were sometimes sensitive to the inclusion of other pollutants (especially NO_2). These studies often used varying indicators of PM (e.g., PM_{10} or $PM_{2.5}$), which can make drawing exact conclusions difficult, but at the same time illustrating the fact that no one study provides the "best" fit for the outcomes.

Acute Exposure Morbidity—Panel Studies

Panel studies involve the examination of small groups of individuals as they undertake normal daily routines or specific activities. In these studies, personal data regarding activity and exposure is collected and examined for relationships to health outcomes. Panel studies can serve as a bridge between large-scale epidemiological analyses and studies conducted under controlled conditions in the laboratory, but are increasingly used to find important relationships on their own. Up to the date of this review, most panel studies had examined only respiratory endpoints. Newer work is focusing on cardiovascular disease and a range of findings in this area are expected in the near future.

Asthmatics have been a key target of air pollution research given their greater potential sensitivities and vulnerabilities to pollutants, with most studies examining relationships in children. Findings generally indicate increased respiratory symptoms and medication use accompanied by reduced lung function following exposure to air pollution. In the small number of studies that examined the severity of asthma response, it appears that those with the worst cases of asthma are vulnerable to greater adverse effects. For non-asthmatics, there is evidence of increased respiratory symptoms, but an effect on lung function is not as evident. Conversely, people with chronic obstructive pulmonary disease (COPD) show evidence of reduced lung function but little impact on medication use or symptoms, though the challenges of studying this often heavily medically-compromised group makes it difficult to draw firm conclusions.

Panel studies of acute morbidity found associations for both PM_{10} and $PM_{2.5}$, with various respiratory symptoms and medication use. Though the studies are limited in number, there are some clear indications that PM_{10} is more likely to be associated with upper respiratory symptoms, while $PM_{2.5}$ is more clearly associated with lower respiratory symptoms, corresponding to the regions of the lung where the particles preferentially deposit. In a small number of studies, ultra-fine particles were also associated with both respiratory and cardiac outcomes, though results were not entirely consistent.

A newer area of research on biomarkers of effect (a physiological or biological measure that is not in itself adverse, but is indicative of a process that can or will lead to one) is less invasive than most laboratory and panel measurements of effect and therefore can be applied to a wider range of people, including those whose health is more compromised. The major areas of investigation for these studies are inflammatory and cardiovascular processes which are known to lead or suspected of leading to both acute and chronic

effects. One of the primary findings in these studies is that the production of pulmonary inflammatory markers is greater in asthmatics versus non-asthmatics, possibly explaining some of the sensitivity of this sub-population to both O_3 and PM.

Also examined in detail is a variety of markers of cardiac and circulatory inflammation which are possibly indicative of acute and chronic cardiovascular stress. Statistically significant findings have been found in elderly populations, but of note is the appearance of stress markers in healthy young subjects. These results provide evidence of pathways to both acute coronary events (for those with pre-existing disease) and long-term cardiac and circulatory tissue damage and dysfunction/disease. The most recent work has focused on indicators of coronary function (such as heart rate variability) and supports both direct and indirect impacts of inhaled PM on cardiac function. Overall, findings from studies using biomarkers are highly supportive of epidemiologic relationships with premature mortality and hospital admissions.

While there are relatively few recent O_3 studies in this area, work characterizing O_3 exposure at childrens' summer camps and of hikers provides confirmation of findings from earlier studies, and some additional insights. These results indicate that lung function declines and respiratory symptoms increase with increasing exposure to O_3, and that these effects may be more pronounced in asthmatics. These effects appear to be independent of PM and other co-pollutants and are observed at levels below those used in laboratory settings.

Conclusions

Long-term exposure studies on respiratory morbidity reported positive and statistically significant associations between fine particles and respiratory effects such as lung function decrements and chronic respiratory diseases, such as chronic bronchitis. For ozone, existing work indicated the potential for chronic respiratory effects, however, it was unclear if such effects were confined to the high exposures in such studies, and newer studies have not shed any light on this issue. However, the California Children's Health Study does provide evidence of effects for both PM and ozone (as well as other pollutants) on a range of adverse respiratory outcomes, most notably the effect on lung growth.

Studies on cardiac morbidity have also provided some insights into the possible mechanisms (e.g., inflammation, cardiac arrhythmias and other cardiac output parameters) by which PM could cause long-term disease and create conditions leading to sudden cardiac events and resultant hospital admission or mortality.

The epidemiologic evidence includes associations between short-term exposure to $PM_{2.5}$ and ozone and cardiorespiratory, hospitalization and emergency department visits. Studies examining specific causes (heart attacks, pneumonia) also indicate the involvement of both pollutants, however these studies are often limited by sample size and are less consistent than is the case for general classifications of disease.

Overall, the evidence base continues to indicate that both PM and ozone are associated with a range of respiratory effects; that $PM_{2.5}$ is capable of eliciting a range of adverse cardiovascular effects, and that both pollutants are associated with increases in medical and hospital visits.

Supporting Evidence from Other Types of Studies

Most epidemiologic studies of the health effects of PM and O_3 rely on measured air concentrations determined at central monitoring sites as a surrogate measure of exposure. This approach has often been questioned due to its potential to introduce a level of uncertainty to the studies, and reducing the validity of their findings. Recent work has addressed the question of whether central monitoring sites are good surrogates for individual exposures, and has largely found that these monitors are quite good at representing population exposure to ambient PM and O_3. As such, studies using ambient measures from central sites monitors are appropriate for use in air pollution epidemiology studies. Other important findings include:

- $PM_{2.5}$ appears to penetrate indoors with considerable efficiency, with the result that ambient $PM_{2.5}$ levels are highly correlated with personal exposure to $PM_{2.5}$ of ambient origin i.e., ambient monitors reflect exposure to particles of ambient origin, even in indoor environments;
- While there can be major sources of PM indoors, these sources do not confound outdoor sources indicating that outdoor monitors can be used to represent human exposure to outdoor PM, especially in the case of $PM_{2.5}$ i.e., levels of indoor and outdoor particles are not correlated. Therefore, while there may be effects of indoor-sourced PM, such effects are not captured in the epidemiological studies of outdoor air;
- Ambient monitors are less representative of exposure to coarse PM and ultra-fine PM, limiting the ability of studies using such monitors to identify effects;
- While there is great variability in individuals' exposure to O_3, studies indicate that this is strongly correlated to levels at ambient monitors;
- Ozone exposure is strongly influenced by building ventilation and time spent outdoors, and is much greater in the warm season. Hence, levels measured at ambient monitors are more representative of personal exposure in the warm season, and indeed O_3 related health effects are often restricted to the warm season; and
- The relationship between ambient concentrations and personal exposure to PM and O_3 will vary as a result of individual-, city-, or region-specific differences, resulting in measurement error and potential bias in risk estimates. The bias can be either upward or downward, but is expected to most often underestimate risks and to make it more difficult to detect a health effect.

Work with human volunteers in controlled laboratory settings (sometimes known as clinical studies) has provided several findings supporting observations made in epidemiological studies. While ethical and logistical considerations limit the extent of controlled human exposure investigations, they have provided evidence of the presence of sensitive groups within the general population, the time-course of damage to the cardio-pulmonary systems, and the mechanisms by which O_3 and PM could elicit effects (though the evidence is much less for PM). Results include:

- Additional evidence of the presence in the population of individuals who respond to even very low levels of air pollution (specifically O_3) who are otherwise healthy;
- The ability of O_3 and possibly PM to enhance airway hyper responsiveness (AHR) in combination with common allergens. AHR is a hallmark of asthma, and these results provide a possible explanation for their greater risk sensitivity in hospital visit and other studies;

- Based on evidence from O_3 exposure studies, lung inflammation, respiratory symptoms, lung function and cellular damage resolve (post-exposure) at very different rates, with cellular damage continuing after other measured responses return to normal;
- For all of the foregoing outcomes, there are very different levels of response in individuals indicating that there is a very wide range of variability and susceptibility in the general population; and
- Exercise, by increasing the effective dose, enhances the sensitivity of individuals to the adverse effects of O_3.

Work in laboratory settings with experimental animals has been increasingly illuminating, even though there are some limitations imposed on interpretation due to issues related to cross-species extrapolation. Earlier work, albeit at high exposure concentrations, had provided a set of mechanisms by which O_3 may exert significant health effects. This research has been enhanced to some degree, with extension to more complex mechanisms for the adverse effects of O_3, and the delineation of an even more complex series of mechanisms by which PM can exert its adverse pulmonary and cardiac outcomes. These include:

- Pulmonary inflammation and modified immune responses with associated increased tendency of infection;
- AHR in animal models of asthma, mirroring such results in human volunteers and giving further weight to the findings for asthmatics in epidemiologic studies;
- The use of different genetic strains of laboratory animals indicates that there are genetically determined sensitivities to PM and O_3, sometimes encompassing several orders of magnitude;
- Inter-individual variability in protective anti-oxidant regimes supports the findings in human studies of sensitive individuals and sub-populations;
- PM (and possibly O_3) causes inflammation not only in the lung, but also and potentially more importantly, in the cardiovascular system. This inflammation, along with oxidative stress, has been shown to lead to a series of biological changes including increased clotting factors, altered vasoconstriction, and alterations in heart rate and heart rate variability, all of which are known risk factors for subsequent adverse cardiovascular events;
- A limited number of studies have found evidence that PM at near-ambient concentrations has the ability to cause the progression of atherosclerotic plaques, providing a specific mechanistic pathway for the observations obtained from both acute and chronic epidemiologic studies of PM-mortality relationships; and
- While different properties (both chemical and physical) of PM have been shown to elicit different effects in animal models, it cannot be yet stated that there are constituents or forms of PM that are without effect.

While much of the work on smog-related pollutants continues to focus on respiratory and cardiovascular outcomes, newer work has indicated that effects may be seen for other biological systems. Of particular note is work indicating effects on the reproductive system (e.g., low birth weights and other pregnancy related outcomes) and the possibility of neurologically-mediated effects. For both these and other outcomes, additional work is required before causal inferences can be made.

Effects on Ecosystem Health

In Canada, quantification of the effects of smog on ecosystems is presently limited to impacts of ozone (O_3) on selected plant species. O_3 is taken up by plants through pores, or stomates, in their leaves. Once inside the plant, O_3 can cause direct physical damage leading to premature senescence (aging), reduced uptake of carbon dioxide (CO_2) and reduced primary productivity. O_3 can also affect a plant indirectly by diverting energy use from important physiological processes to the detoxification of O_3.These effects on individual plant health can lead to ecosystem changes, as plant species that are more resistant to O_3 can become more dominant than those that are less resistant. A plant's response to O_3 is a function of both duration and concentration of exposure, varies with species and plant development stages, and is modified by environmental factors, such as soil moisture content and humidity.

Currently, exposure-based metrics are the most appropriate tools for quantifying concentration exposure-response relationships and are well correlated with the metric used to calculate the Canada-wide Standard for O_3. This indicates that measures aimed at reducing the 8-hour average O_3 concentration would also reduce vegetation exposure to O_3.

The impacts of particulate matter (PM) on vegetation are dependent upon the chemical constituents and the particle size, as these define the plant-specific PM phytotoxicity levels. Plant response to PM is largely due to the resultant changes in soil chemistry (i.e., acidic PM deposition can leach out soil nutrients into surface waters) rather than direct deposition on the plant. Various PM constituents taken up by the plant from the soil can interfere with photosynthesis, thereby reducing plant growth and productivity. PM can also cause physical damage to plant surfaces via abrasion. Although beyond the scope of this assessment, PM deposition has also been shown to affect surface water chemistry and species diversity[10].

The impacts of PM and O_3 on wildlife is an area of emerging concern in understanding impacts to ecosystems. Research has focused on indirect effects of PM and O_3 on wildlife species through influences on vegetation, soil conditions and changes to habitat on which they depend. O_3 animal exposure studies (e.g., mice, rats, guinea pigs, rhesus monkeys), which are designed to extrapolate results for assessing exposure effects on humans, indicate species are variable in their responses and in their sensitivity to exposure; thus, the same can be reasonably assumed for wildlife species. At this time, however, there is no research available to extend the results of these laboratory studies to assess the impacts on wildlife species or to identify which species are more sensitive to exposure.

Effects on Social and Economic Wellbeing

Smog also has wide-ranging socio-economic impacts on health-related and non-health related outcomes valued by Canadians. Among the socio-economic impacts of smog on human health that can be quantified are medical treatment costs and lost worker productivity associated with hospital admissions and emergency department visits due to respiratory and cardiac illnesses.

10. Environment Canada, 2005. *The 2004 Canadian Acid Deposition Science Assessment*. Environment Canada, Gatineau, Quebec.

Non-health related impacts on social and economic wellbeing are also significant. O_3 can impact the output of Canadian farms and forestry operations by decreasing the primary productivity of crops and forests. The Province of Ontario estimated that O_3 causes $201 million in damages annually for select crops. Smog can also lower the enjoyment that Canadians and tourists experience because the light scattering and absorbing properties of PM can impair visibility of their surroundings and scenic locations. Visibility is based on visual perception and is one of the most obvious indicators of air quality to the public. Following a pattern similar to PM concentrations, visibility in Canada is best on the coasts and worst in the more urbanized central areas of the country. There are economic costs associated with impaired visibility. For example, the impact of a single extreme visibility degradation event on tourist recreational revenue is estimated at a loss of about $7.45 million in Vancouver and $1.32 million in Fraser Valley. The Canada-wide Standard for $PM_{2.5}$ does not preserve visual air quality as it can be impaired through a range of $PM_{2.5}$ levels, even at concentrations below 30 µg m^{-3}.

Smog also impacts our built environment by increasing the rate at which materials break down. PM impacts materials through accelerating naturally-occurring processes such as discolouration, fading, tarnishing, which could increase the rates at which materials (e.g., rubbers, textiles, surface coatings) need to be replaced or cleaned.

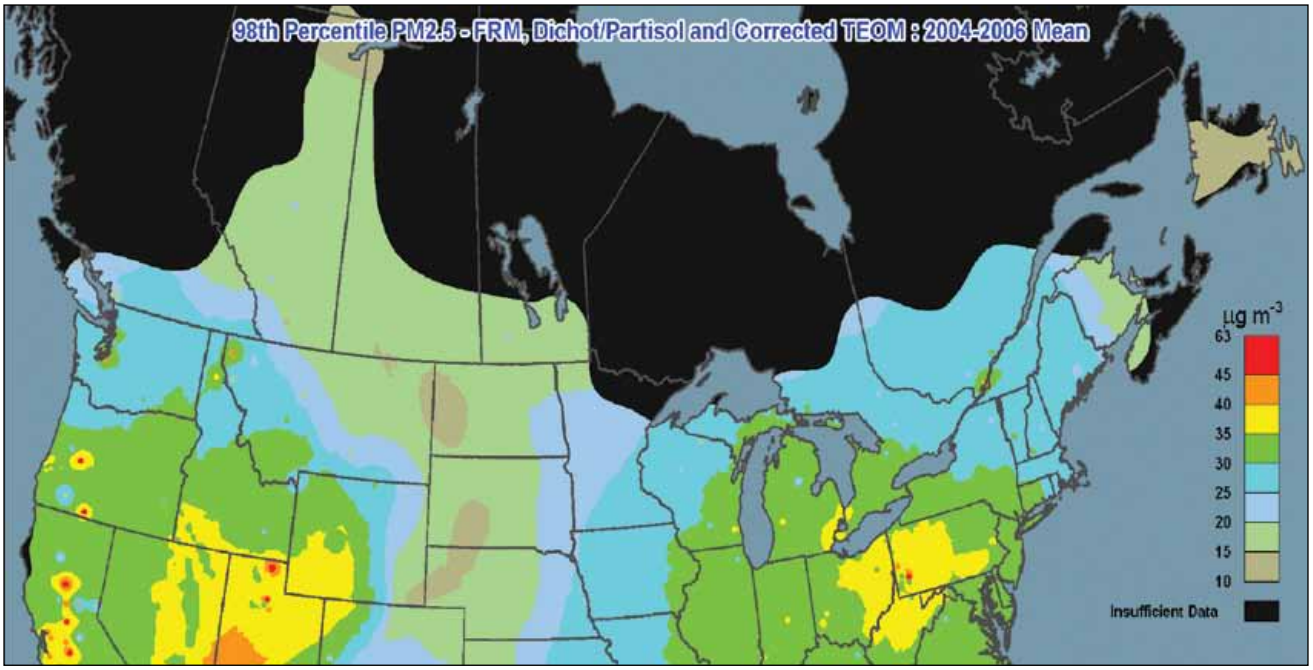
Levels of Smog in the Atmosphere

Recent Levels of Fine Particulate Matter ($PM_{2.5}$)

As mentioned under “Effects on Human Health”, fine particulate matter ($PM_{2.5}$) is especially (although not exclusively) harmful to human health and, therefore, is the size fraction emphasized in this discussion.

The spatial pattern of ambient $PM_{2.5}$ levels across the country in 2004–2006 (Figure 1), as defined by the Canada-wide Standard metric, shows southern Ontario and southern Quebec having the highest concentrations (>25 µg m^{-3}). This area is also part of a larger high concentration airshed that encompasses all of the eastern U.S. The highest levels in Canada over the 2004–2006 period occurred in the Great Lakes region, particularly in southwestern Ontario where densely populated urban sites experienced levels above the Canada-wide Standard. Across southern Quebec and eastern Ontario, $PM_{2.5}$ concentrations were generally below the Canada-wide Standard target, with the exception of some specific communities influenced by local industries and large urban centres, highlighting the potential for emissions to lead to localized high levels. There are uncertainties in the exact levels and local details of the spatial distribution of $PM_{2.5}$ (Figure 1) over some areas of the country because of a lack of $PM_{2.5}$ measurement sites. However, on a broad scale, Figure 1 provides a general picture of the spatial pattern.

To track the occurrence of high $PM_{2.5}$ concentration events across the country, the number of days per month in which the daily 24-hour average of $PM_{2.5}$ concentrations exceeded 30 µg m^{-3} was counted over the period 2001–2005 at sites across Canada. Days with $PM_{2.5}$ concentrations >30 µg m^{-3} can occur any month of the year but sites in southern Ontario and southern Quebec experienced the greatest frequency of days >30 µg m^{-3} in summer followed by winter. At the western sites, the highest frequency of days occurred in the summer associated with the occurrence of forest fires.



Note: Areas in black indicate either an insufficient number of sites or incomplete data for mapping.

Figure 1 Spatial distribution of the 98th percentile 24-hour $PM_{2.5}$ concentrations (µg m^{-3}) across Canada and the U.S. for the period 2004–2006

The frequency of regional scale episodes, defined as days where 33% of air monitoring sites in a region record 24-hour average $PM_{2.5}$ concentrations above 30 µg m^{-3}, was also determined for 2001–2005. Regional scale episodes of $PM_{2.5}$ occurred in both winter and summer. The greatest frequency of regional scale episodes also occurred in Ontario followed by Quebec with high $PM_{2.5}$ values often persisting for several days. Summer $PM_{2.5}$ episodes in Ontario and Quebec were often associated with O_3 values greater than the Canada-wide Standard target of 65 ppb, whereas winter episodes were solely a result of high levels of $PM_{2.5}$. Regional episodes were infrequent in the Prairies and the Lower Fraser Valley of British Columbia, and the events that did occur were associated with forest fires. Although some areas may not experience frequent regional episodes, they may still experience days when $PM_{2.5}$ levels are considered high locally and for which there would be an increase in health effects and a decline in visibility relative to average conditions.

Levels of $PM_{2.5}$ vary considerably by season within a region as observed through the analysis of $PM_{2.5}$ daily peaks and averages measured at monitoring sites. In southern Atlantic Canada, the daily peaks and averages in $PM_{2.5}$ are higher in summer than winter. This seasonal difference is due to more intense sunlight leading to greater sulphate concentrations, and more frequent favourable wind patterns that carry pollutants from sources in the southwest. In winter $PM_{2.5}$ levels are influenced more by local sources such as residential wood combustion. Across southern Quebec and eastern Ontario daily peaks and averages are higher in winter than summer. In particular, over the southern Great Lakes region, the highest daily averages occur in winter as colder temperatures favour the formation and build up of ammonium nitrate on the particles (see "Composition of Secondary $PM_{2.5}$") and meteorological conditions result in less pollutant dispersion.

In Alberta, both $PM_{2.5}$ peaks and averages are higher in the winter when cold and calm conditions lead to some of the highest concentrations of primary pollutants in Canada. In the Lower Fraser Valley, daily average levels are highest in late summer and early fall with peaks in fall and winter. This is due to favourable meteorological conditions in the late summer and changes in local activities, including wood burning and space heating in the fall and winter. In the interior of British Columbia, levels also peak in the fall and winter where emissions from residential wood combustion play a role along with strong inversions due to cold air pooling in the valleys and/or the trapping of air by the mountains. In Whitehorse, Yukon, daily averages are highest during the summer months, and annual variability at this site appears to be influenced by forest fires. In the winter high $PM_{2.5}$ levels occur due to woodsmoke.

The spatial pattern of the annual mean of $PM_{2.5}$ concentrations in Canada is very similar to the spatial pattern of the Canada-wide Standard metric (Figure 1), with the highest concentrations (>8 µg m^{-3}) occurring in southern Ontario and southern Quebec.

Composition of Secondary $PM_{2.5}$

The major components of $PM_{2.5}$ as measured at sites across the country are ammonium sulphate, ammonium nitrate and organic matter (Figure 2), although the composition varies by location and by season. The first two components are products of nitrogen oxides (NO_x) and sulphur dioxide (SO_2) oxidation in the presence of ammonia (NH_3). Organic particulate matter can be directly emitted, along with elemental or black carbon, from many combustion sources including fossil fuels, wood and cooking. Organic matter (OM) can also be formed in the atmosphere (secondary formation), largely from reactions involving both anthropogenic and biogenic volatile organic compounds (VOC).

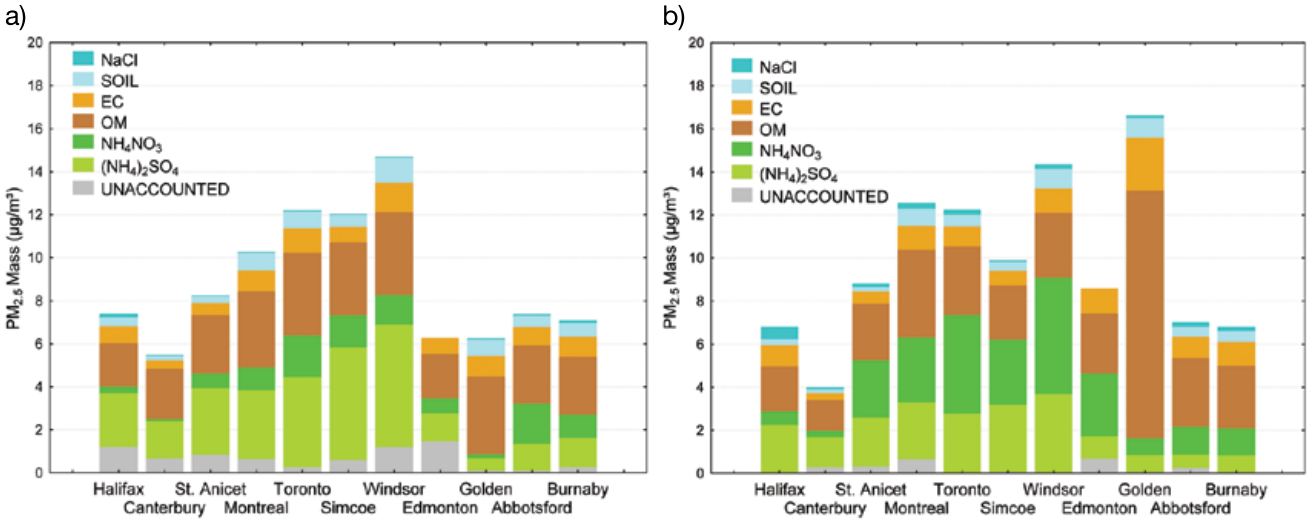


Figure 2 Mass fractions of $PM_{2.5}$ component species in the warm season (a) and in the cold season (b) sampled in 2003–2006 at selected National Air Pollution Surveillance (NAPS) network sites

In the warm season, $PM_{2.5}$ at speciation sites in Ontario, Quebec and the Atlantic region contains a relatively large amount of ammonium sulphate $(NH_4)_2SO_4$ followed by organic matter (OM) (Figure 2a) on average days and during peak episodes. Ammonium sulphate reaches its maximum during this season when photochemistry and transport from Canadian and U.S. emission sources are greatest. The warm season data from the western sites (with the exception of Golden, British Columbia) indicate that contributions from the main chemical constituents tend to be more equal, though organics have greater importance during peak episodes.

In the cold season, ammonium nitrate (NH_4NO_3) becomes a more important contributor (Figure 2b). In locations where wood burning is common, exemplified by Golden, British Columbia, organic matter is dominant and $PM_{2.5}$ attains some of the highest levels observed in the country. For the highest concentration days in winter at Ontario and Quebec sites, ammonium nitrate was the primary contributor. Ammonium sulphate remains an important contributor at eastern sites and the primary contributor at Atlantic sites. At sites in British Columbia, the highest $PM_{2.5}$ days in winter are dominated by organic matter.

Baseline Levels of $PM_{2.5}$

In the context of this assessment, background levels of $PM_{2.5}$ are defined as the ambient concentrations resulting from natural emission sources within North America, and from the long-range transport of anthropogenic and natural emissions from outside North America. Background levels are important in air pollution management as the responsible sources cannot be controlled by domestic or continental emission reduction strategies, although international negotiations on anthropogenic emission reductions may provide some benefit. Background levels explain some of the spatial variation in ambient concentrations, though they are influenced by atmospheric conditions and can be highly variable over space and time making them particularly difficult to quantify.

However, since background $PM_{2.5}$ levels are not directly observable, measurement data have been used to define a baseline level, the $PM_{2.5}$ level at a given site in the absence of strong local influence. Estimates of baseline annual median $PM_{2.5}$ levels are available for only a few regions across the country and range from 1–4 $\mu g\ m^{-3}$. Due to an insufficient number of long-term and regionally-representative sites, it is not currently possible to assess the temporal trends of baseline $PM_{2.5}$. These baseline levels reflect average atmospheric concentrations upon which $PM_{2.5}$ associated with North American anthropogenic activities is superimposed. Thus, neither background nor baseline is associated with the peak concentrations observed in time or space that are of considerable importance to human exposure. This conceptual difference makes it difficult to incorporate background or baseline values into discussions of the Canada-wide Standard targets which are indicators of high concentrations alone.

Trends in Ambient $PM_{2.5}$

In general, ambient annual mean and 98th percentile $PM_{2.5}$ concentrations at urban sites across Canada have declined by approximately 40% from 1985 to 2006 (Figure 3). The largest declines occurred prior to 1996 as a result of SO_2 emissions reductions in eastern Canada and the eastern U.S. with little change since then. Some urban locations still experience high ambient SO_2 levels due to large point sources and the same applies for $PM_{2.5}$.

At rural and remote sites, the lack of long-term data precludes analysis of long-term trends for ambient $PM_{2.5}$ concentrations. However, between 1999 and 2006, SO_2 and NO_x emissions reductions in eastern Canada and the eastern U.S. resulted in overall reduced ambient concentrations of precursors (i.e., ambient SO_2 and nitric acid (HNO_3)) and $PM_{2.5}$ constituents (i.e., ambient particle sulphate, particle nitrate and particle ammonium).

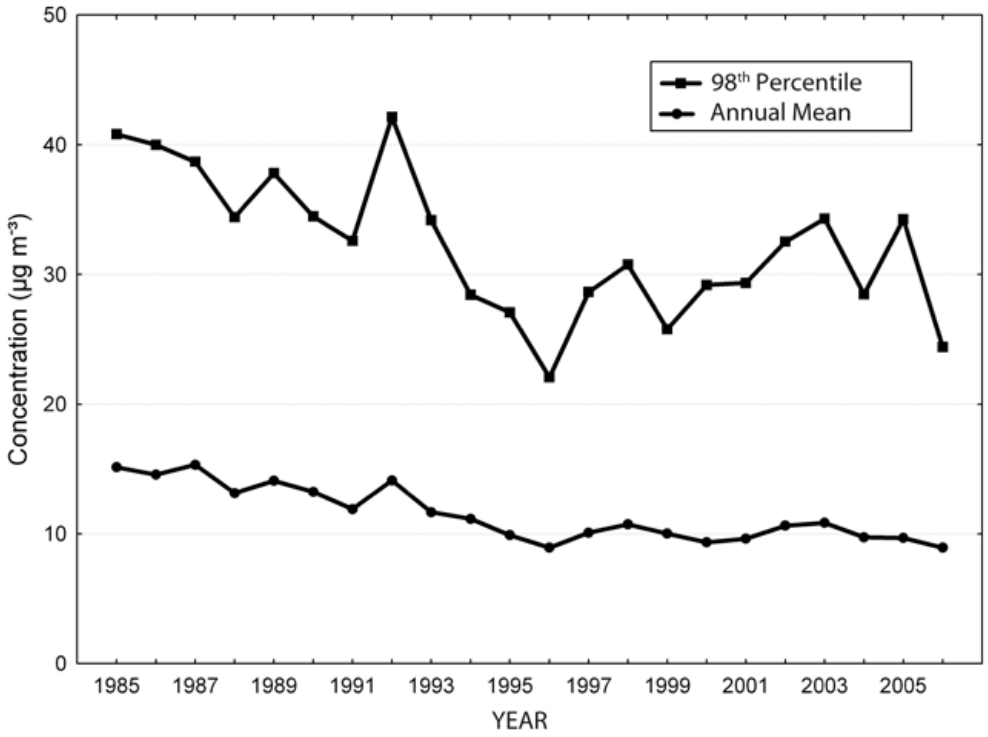


Figure 3 Trend in composite annual mean and 98th percentile $PM_{2.5}$ concentrations from dichotomous sampler sites

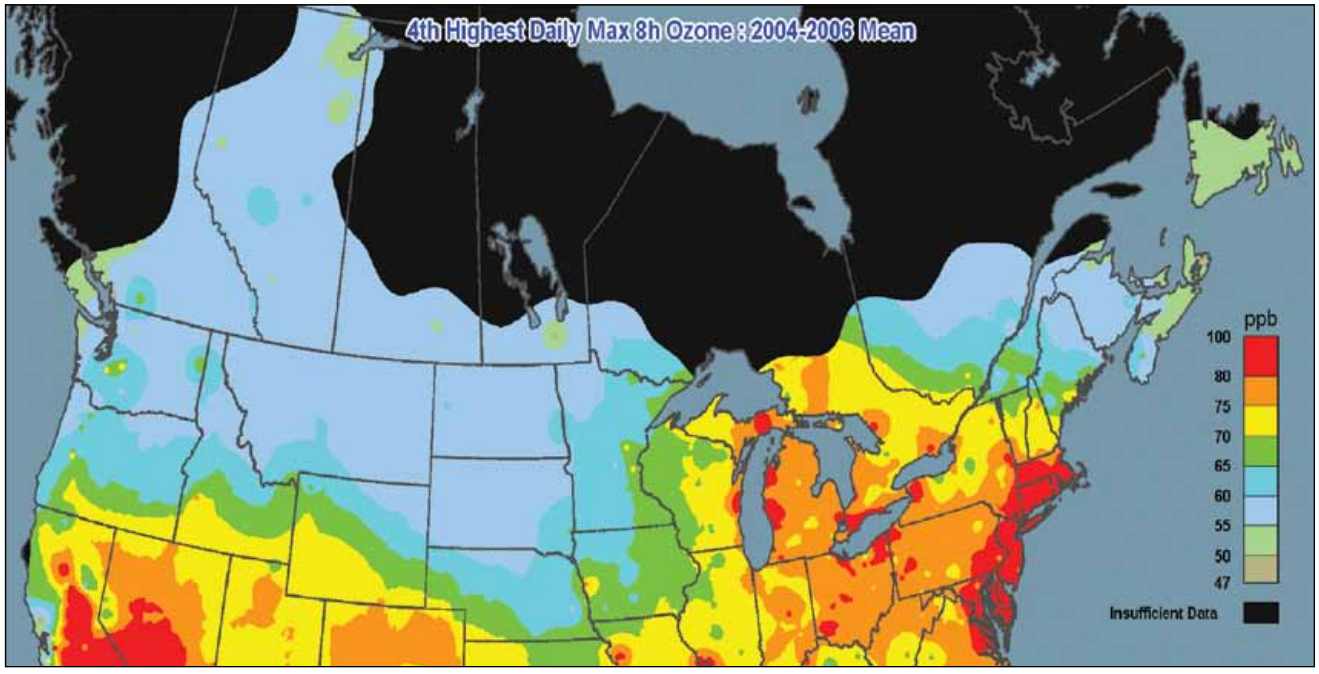
In light of decreasing SO_2 and NO_X emissions in eastern North America, emissions and ambient concentrations of NH_3 are expected to become more important in determining levels and trends of $PM_{2.5}$ (see section on "Factors Influencing Levels of Smog Across Canada"). $PM_{2.5}$ levels in various regions of the country exhibit different sensitivities to ambient NH_3 concentrations depending on the season and local chemical regime[11], increasing the complexity in reducing ambient $PM_{2.5}$.

11. For additional detail on the impacts of agricultural ammonia on local $PM_{2.5}$, see Li, S.-M., R. Vet, J. Liggio, P. Makar, K. Hayden, R. Staebler, E. Chan, and M. Shaw, 2010. Chapter 7: Sensitivity of particulate matter to NH_3 in major agricultural regions of Canada, in: *The 2008 Canadian Atmospheric Assessment of Agricultural Ammonia.* Environment Canada, Ottawa.

Recent Levels of O_3

The spatial pattern of O_3 levels across the country in 2004–2006, expressed according to the Canada-wide Standard metric (Figure 4), shows that the highest concentrations in Canada continue to occur over southern Ontario and southern Quebec. This area is within a large high concentration region (>65 ppb), encompassing the entire northeastern U.S. The highest concentrations in Canada occurred in the Great Lakes region, particularly in southwestern Ontario coincident with the highest $PM_{2.5}$ concentrations. Virtually all sites in the southern Great Lakes region recorded levels above the Canada-wide Standard for O_3. In southern Atlantic Canada, O_3 levels were considerably lower than southern Ontario and extreme southern Quebec, but there was noticeable spatial variability and some areas just surpassed the Canada-wide Standard. Similarly, levels observed in most of western Canada are lower (40–60 ppb) except around Edmonton, Alberta, and the eastern edge of the Lower Fraser Valley, British Columbia. Air quality in the Yukon and in the Northwest Territories (not shown in Figure 4 due to the limited number of sites) is generally better than in southern Canadian cities because of fewer local industrial sources and smaller, more dispersed populations. While the spatial and temporal coverage of monitoring sites for O_3 is greater than for $PM_{2.5}$ there are still uncertainties in the exact levels and local details of the spatial distribution of O_3; however, on a broad scale this map provides a general picture of the spatial pattern across Canada.

To provide an indication of the occurrence of high concentration events across the country, the number of days in which the daily maximum 8-hr average O_3 concentration exceeded 65 ppb over the period 2004–2006 was counted at sites across the country. The greatest number of days was recorded in southern Ontario along the north shore of Lake Erie (30–50 days) followed by the rest of southern Ontario



Note: Areas in black indicate either an insufficient number of sites or incomplete data for mapping purposes.

Figure 4 **Spatial distribution of the three year average (2004–2006) of the fourth highest daily maximum 8-hr O_3 average concentration (ppb) across Canada and the U.S.**

and southern Quebec (5–30 days). In contrast, almost all sites in Western and Atlantic Canada had zero to five days with daily maximum 8-hr O_3 levels of 65 ppb or greater.

The greatest frequency of regional scale episodes over the 2000–2005 period, defined as days when 33% of monitoring sites in a given geographic area record daily maximum 8-hour average O_3 levels above 65 ppb, occurred in Ontario and Quebec, with high levels persisting for several days and often associated with $PM_{2.5}$ levels above the Canada-wide Standard target of 30 µg m^{-3}. Regional scale O_3 episodes were almost non-existent in the Prairies and in the Lower Fraser Valley, with a couple of notable exceptions associated with forest fires.

Temporal Variations in Ambient O_3 Levels

There are pronounced seasonal variations in ambient O_3 levels, regionally and at individual sites throughout Canada. Many sites in Canada, especially outside of Ontario and Quebec, record the highest average daily mean and average daily maximum O_3 levels in the spring (Figure 5). Remote sites most commonly experience a predominant maximum O_3 concentration in the spring due to several mechanisms,

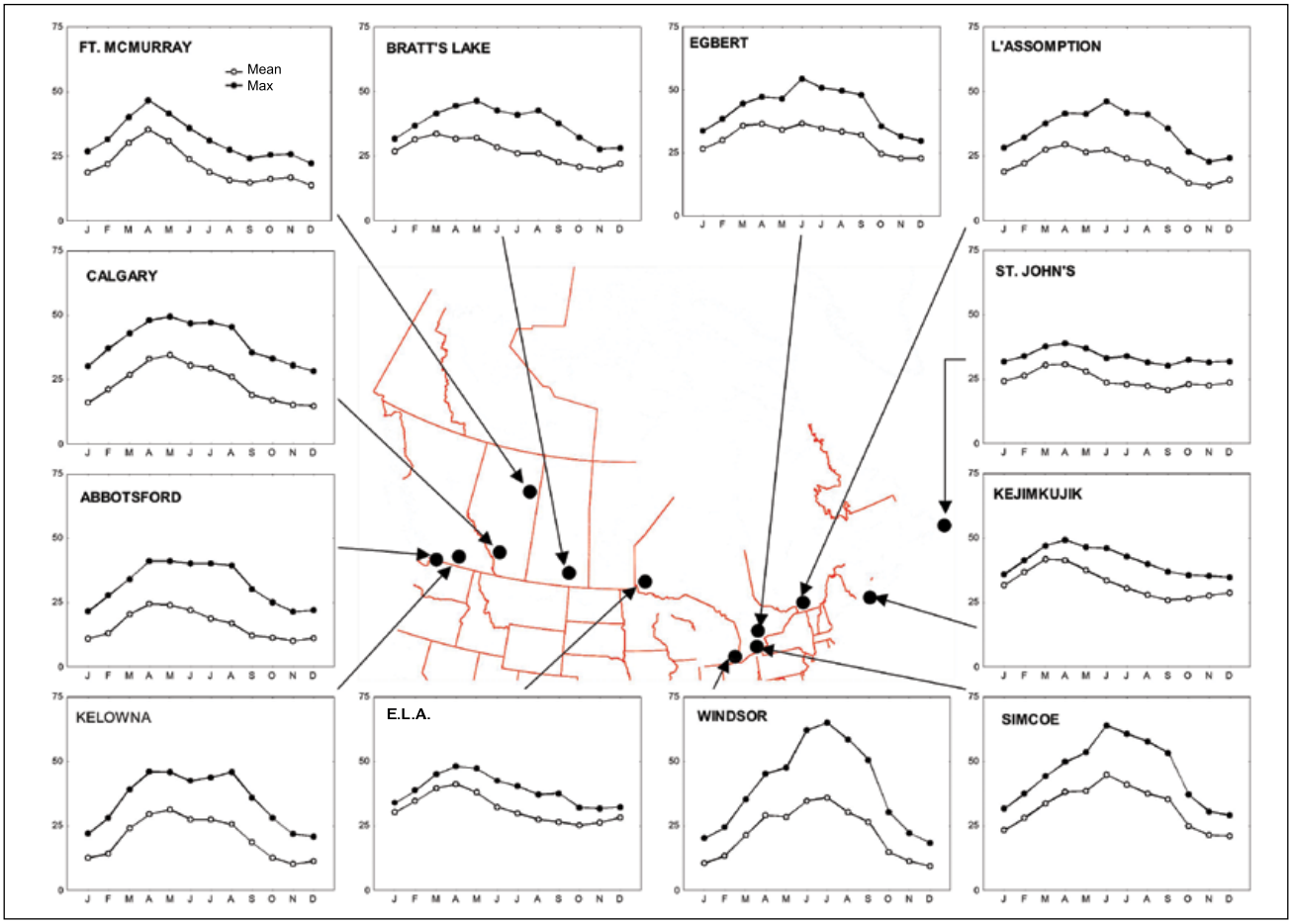


Figure 5 Seasonal variation in monthly averages of daily mean (o) and maximum (●) O_3 concentrations (in ppb) at selected sites (both urban and rural) across Canada averaged over the period 2001–2005

including enhanced photochemistry in the troposphere, downward exchange of O_3 from the stratosphere and enhanced hemispheric transport. Some of these sites with a spring maximum also have a less-pronounced secondary maximum in the summer.

In southern Ontario, southern Quebec and the eastern Lower Fraser Valley, the highest daily mean and maximum ambient O_3 levels occur in the late-spring and/or summer, which generally indicates the impact by anthropogenic emissions and subsequent local and regional scale photochemical production. In some regions closer to the border, long-range transport of pollutants from the U.S. also plays a large role in these peak O_3 events.

On average, ambient O_3 levels are low in the winter due to less intense sunlight, which reduces photochemical production of O_3, and different meteorological patterns, including a greater frequency of strong, northerly winds.

Baseline Levels of O_3

As earlier described under "Baseline Levels of $PM_{2.5}$", background is the ambient concentration resulting from natural emissions within North America and from the long-range transport of anthropogenic and natural emissions from outside North America. On the other hand, baseline O_3 is the O_3 level measured at a given site in the absence of strong local influence. In Canada, estimated baseline seasonal average O_3 levels are 19 ± 10 ppb in Pacific Canada, 28 ± 10 ppb in continental western Canada, 30 ± 9 ppb in continental eastern Canada, and 27 ± 9 ppb in Atlantic Canada. For broad comparison purposes only, because direct comparison of different metrics is not possible, the fourth highest daily maximum 8-hr concentration value at the most remote sites in Canada ranged from 44–53 ppb in 2006.

Baseline O_3 levels have been increasing in a number of areas of Canada; namely the Georgia Basin (coast of British Columbia), the Atlantic coast, and continental western Canada. The increasing trend in western Canada is consistent with evidence of increasing transport of O_3 into North America from Asia and a general increase in hemispheric O_3. In contrast, the data indicate that the baseline has been decreasing in Ontario and Quebec; however, the baseline level in these areas is less important to air quality management considerations because anthropogenic sources within the region and transboundary transport from the U.S. continue to be a dominant contributor to O_3. As O_3 precursor emissions continue to decline in North America, any increase in levels of baseline O_3 could play an increasingly important role in the achievement of ambient air quality targets.

Trends in O_3 Levels

Although a downward trend was observed in national O_3 levels in the form of the Canada-wide Standard from 2003–2006, no overall significant trend has been observed from 1990–2006. However, trends vary markedly depending on site location (urban, rural, background), period of record analyzed and the metric being examined.

The impact of Canada and U.S. NO_X emission reductions[12] on O_3 can be seen by examining meteorologically-adjusted trends at non-urban sites. Comparing the two periods of 1997–2000 and 2003–2006 (Figure 6) reveals that the four year average daily maximum 8-hr O_3 concentrations in the summer declined by 3.2% at sites in Quebec and by 4.1% at sites within an international airshed encompassing Ontario and U.S. locations in the Great Lakes/Upper Ohio region. On the other hand, ambient concentrations increased by 2.1% at sites in Atlantic Canada, remained constant at sites in Alberta, and increased by 5.2% at sites in the Georgia Basin area of the Pacific coast.

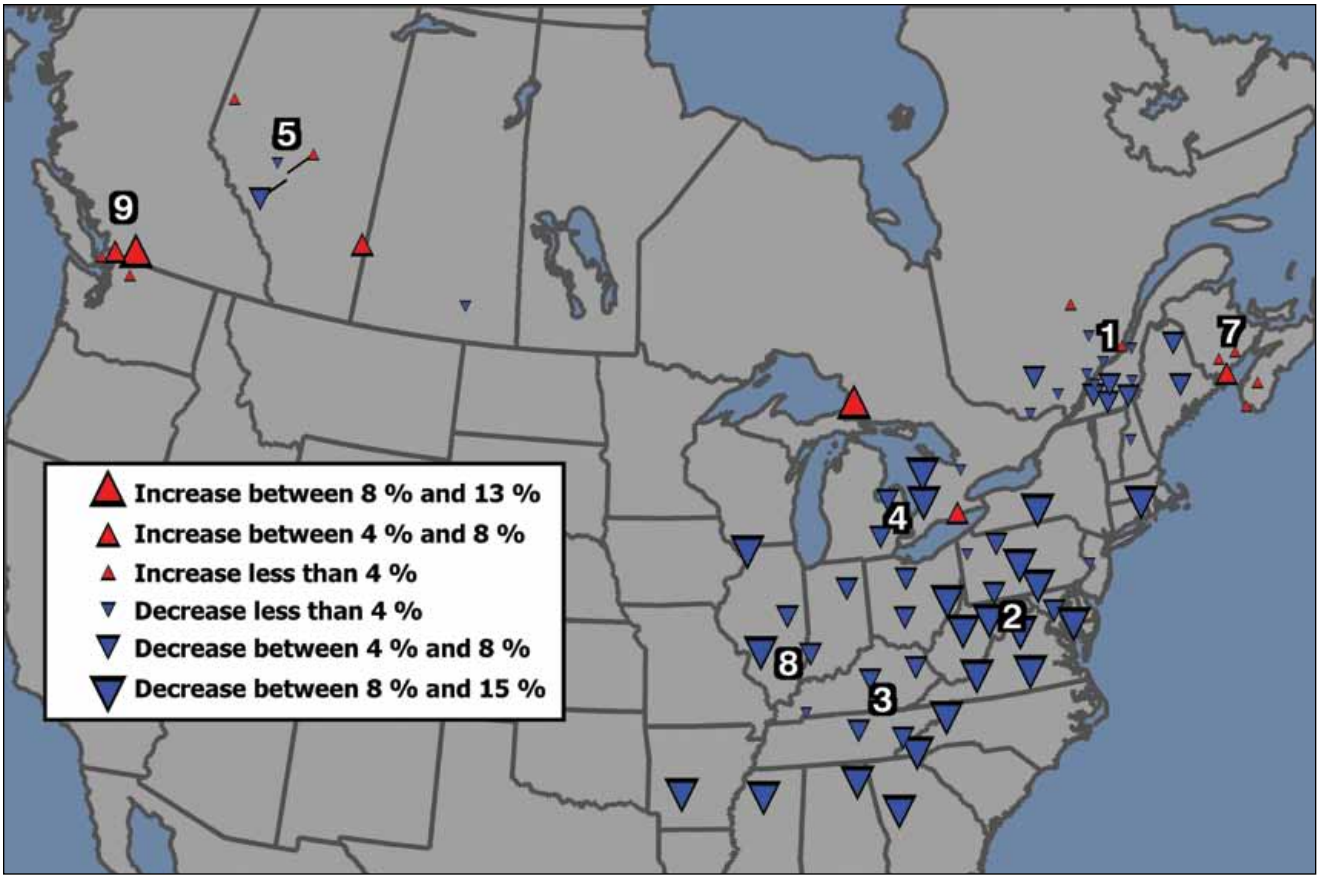


Figure 6 **Differences in the meteorologically-adjusted four-year average summer O_3 levels from 1997–2000 to 2003–2006 based on daily maximum 8-hr values**

At urban sites across Canada, there has been an increasing trend (not meteorologically-adjusted) in median and lower percentile ambient 8-hr O_3 concentrations between 1990 and 2006. This upward trend in the annual median and lower percentile ambient O_3 observations at urban sites is because there is less O_3 scavenging (where O_3 is removed when it reacts with NO to form NO_2) as a result of reductions in NO_X emissions. Over this period, annual mean ambient concentrations of NO, NO_2 and VOCs at urban

12. In Canada, NO_X emissions reductions have been achieved via the On-Road Vehicle and Engine Emission Regulations and annual caps on emissions from fossil-fuelled power plants in central and southern Ontario and southern Quebec. NO_X emissions reductions in the U.S. have been achieved under the NO_X SIP (State Implementation Plan) Call (NO_X Budget Trading Program) and Title IV of the *Clean Air Act* which requires NO_X reductions from certain coal-fired electrical generation units.

sites decreased by 55%, 34% and 46% respectively, with similar decreases at all sites. O_3 levels in urban areas have typically been lower than the surrounding rural areas as a result of O_3 scavenging. Thus, with less NO urban O_3 can increase depending upon other factors such as the relationship between ambient concentrations of NO_x and VOCs.

Consistent with the meteorologically-adjusted analysis, rural sites in Ontario and Quebec experienced a decreasing trend for all percentile O_3 concentrations from 1990–2006. This decrease was most significant in the upper part of the data distribution (i.e., higher percentiles, maximum), in response to regional scale reductions in precursor emissions. This includes a possible accelerating reduction in eastern Canada from 2004–2007 due to NO_X emission reductions in eastern Canada and in the northeastern and midwestern U.S.

Projections of Future Levels of $PM_{2.5}$ and O_3

Chemical transport models are amongst current state-of-the-science models capable of simulating atmospheric chemical conditions in response to anticipated changes in emissions relative to a base or reference year. One such model, A Unified Regional Air Quality Modelling System (AURAMS), was used to estimate the levels of $PM_{2.5}$ and O_3 across Canada and the U.S. for the year 2015 relative to the reference year 2002. A comparison of model predictions to observations of annual O_3 and $PM_{2.5}$ for the year 2002 shows they are well correlated geographically, providing a positive indication of model performance.

The emissions underlying the 2015 AURAMS $PM_{2.5}$ and O_3 projections (Table 1) were based on the implementation of existing Canadian and U.S. legislation, or business as usual (BAU), including the NO_X State Implementation Plan (SIP) Call and the Clean Air Interstate Rule (CAIR) as proposed prior to 2008 and now replaced by the Transport Rule.

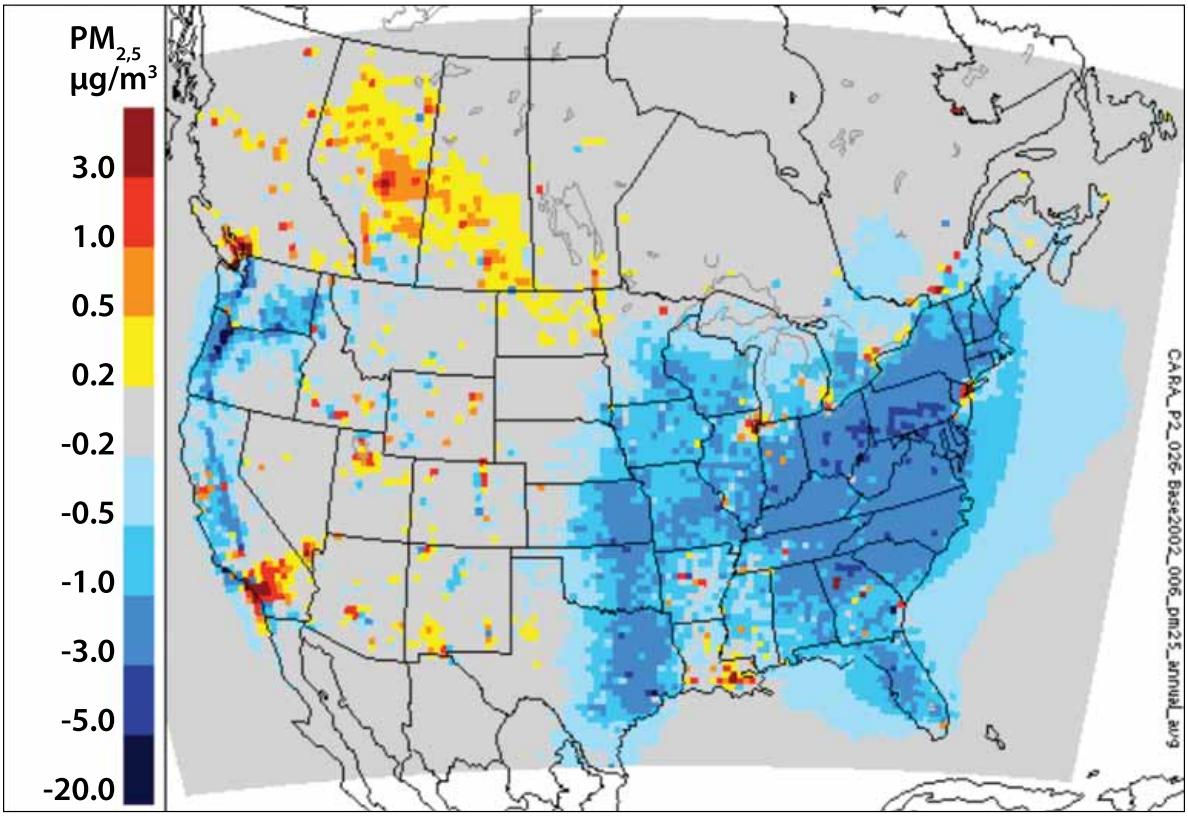
Table 1 Emissions changes between 2002 and 2015 in Canada and the U.S. included in the 2015 AURAMS business as usual scenario

Substance	Canada*	Eastern U.S.	Western U.S.
NO_X	-24.5%	-26.0%	-6.3%
SO_X	-22.0%	-24.8%	-0.4%
VOCs	-14.0%	-18.0%	-17.9%
Primary $PM_{2.5}$	+19.6%	-21.1%	-23.7%

Note: Emissions from forest fires are excluded and a 0.75 discount factor is assumed for primary $PM_{2.5}$ emissions from open sources.

*The 2015 BAU scenario was based on an earlier version of the 2015 Canadian and U.S. combined inventory and does not reflect recent updates in emissions projections presented in Chapter Four of the Assessment. As a result, the projected changes in Canadian $PM_{2.5}$, NO_X and SO_2 emissions based upon the latest information and presented in Figure 9 of this document are smaller than the projections on this table, and the direction of the Canadian VOC emissions trend is actually reversed (+12% versus -14%). The modelling results in Figures 7 and 8, which correspond to the emissions changes in this table are, nonetheless, indicative of the directional response of $PM_{2.5}$ and O_3 that can be expected across Canada relative to such emissions changes.

In 2015, annual $PM_{2.5}$ levels (Figure 7) are projected to be 0.2–3 µg m^{-3} (10–30%) lower than the 2002 levels in most of southern Ontario, responding primarily to emissions reductions in the U.S.; however, only marginal improvements are projected east of Toronto, including Quebec and most of the Atlantic Provinces. Increases of 0.2–3 µg m^{-3} (15–30%) are projected for a number of localized areas along the Windsor–Quebec City corridor, including the major cities. These estimates are due to increases in primary $PM_{2.5}$ emissions in the Windsor–Quebec City corridor and other upwind regions. Urban centres in Manitoba, Saskatchewan and British Columbia are also projected to experience $PM_{2.5}$ increases of 1–3 µg m^{-3} (20–50%), while the surrounding areas are projected to remain unchanged. These results show that projected $PM_{2.5}$ levels are sensitive to changes in primary $PM_{2.5}$ emissions; therefore, the magnitude of the change in ambient $PM_{2.5}$ needs ongoing re-evaluation based on the latest projections of Canadian primary $PM_{2.5}$ emissions.



Note: Blue regions correspond to areas of projected decreases in $PM_{2.5}$ concentrations while yellow to red regions correspond to areas of projected increases in $PM_{2.5}$ concentrations.

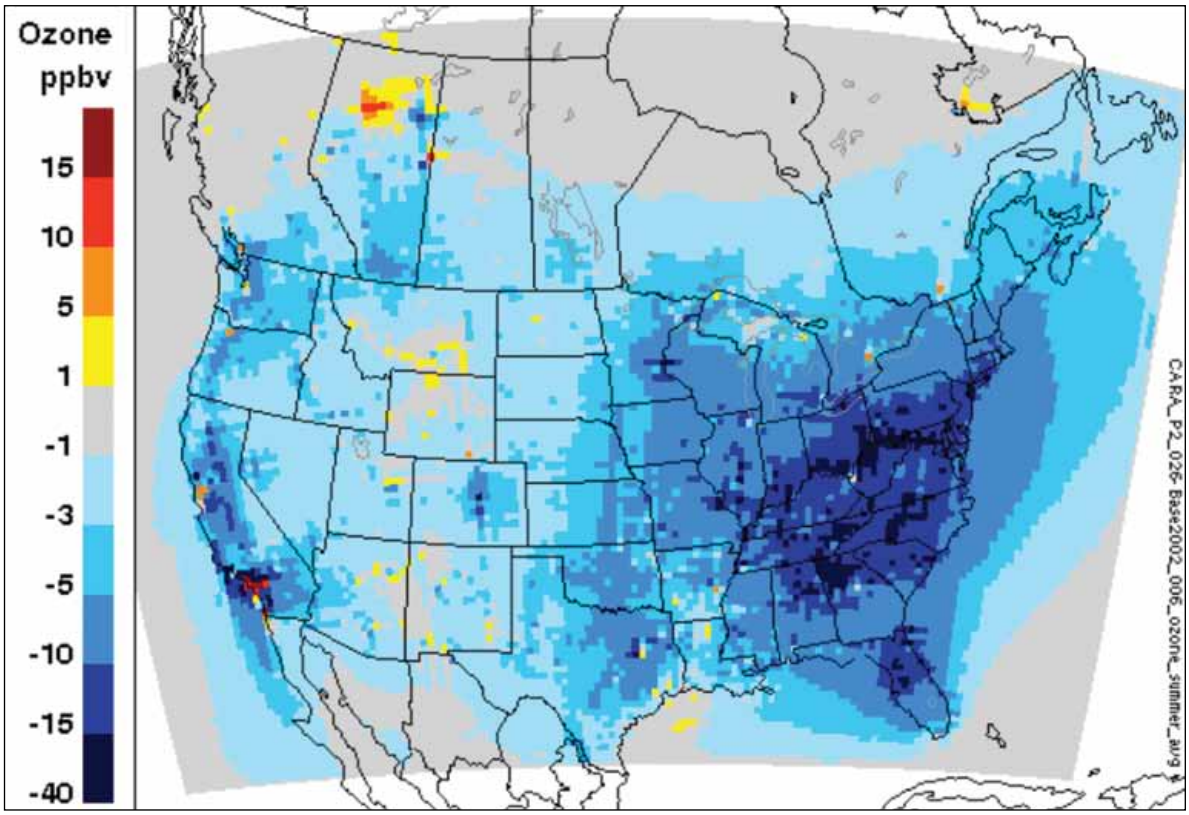
Figure 7 Absolute difference in the annual $PM_{2.5}$ 24-hour average between the 2015 BAU simulation and the 2002 reference case

Widespread increases in $PM_{2.5}$ are projected to occur in the Prairies, with the largest increases exceeding 3.0 µg m^{-3} (20–40%) in the vicinity of Edmonton, Alberta and southern Saskatchewan. While these are the greatest projected Canadian increases in annual ambient $PM_{2.5}$ concentrations, they do not result in average annual levels above 10 µg m^{-3} anywhere in the Prairies except over the main urban centres. As in the Windsor–Quebec City corridor, these changes are due to increases in primary $PM_{2.5}$ emissions. However,

changes in the local chemical regime also appear to be playing a role given that increases in the ambient concentrations of inorganic secondary PM components (sulphate, nitrate and ammonium) are predicted despite an overall decrease in emissions of NO_X, SO_2 and VOCs over the area.

The 2015 BAU AURAMS simulation did not predict additional excedances of the Canada-wide Standard for $PM_{2.5}$ in new locations compared to the 2002 reference year. Rather, it showed an increase in the frequency of exceedances in places that were already exhibiting levels above the $PM_{2.5}$ Canada-wide Standard numerical value of 30 µg m^{-3}.

Relative to 2002, average summer time 8-hour daily maximum O_3 levels in rural or remote southern Ontario, southern Quebec and the Atlantic provinces are projected to decrease by approximately 3–10 ppb (or 10–30%) by 2015 (Figure 8). These changes are in response to projected decreases in NO_X and VOC emissions (Table 1), especially along the Windsor–Quebec City corridor. Furthermore, these regions are significantly influenced by transboundary transport of O_3 and its precursors from the U.S., where NO_X and VOC emissions are also projected to decrease.



Note: Blue regions correspond to areas of projected decreases in O_3 concentrations while yellow to red regions correspond to areas of projected increases in O_3 concentrations. Note that the Fort McMurray increase is displaced to the west due to a geographic reporting error in the 2015 projected emissions inventory.

Figure 8 Absolute difference in the average summertime (June–August) 8-hour daily maximum O_3 between the 2015 BAU simulation and the 2002 reference case

Ambient O_3 concentrations in urban centres in eastern Canada are projected to increase by 5–10 ppb from 2002–2015, particularly in Toronto and Montréal, due to reduced NO titration associated with decreases in local NO_X emissions. This is a drawback associated with the complexity of O_3 chemistry whereby reductions in precursor emissions do not always translate into anticipated reductions in O_3 concentrations. However, these urban areas will benefit directly from the local NO_X emission reductions and also from reductions in O_3 levels and NO_X emissions transported from neighbouring cities.

In southwestern British Columbia, southern Alberta and the vicinity of Edmonton, O_3 levels are projected to decrease at about the same magnitude as the eastern provinces (3–10 ppb). In Saskatchewan and Manitoba, improvements are projected to be more moderate at 1–3 ppb. In contrast, O_3 levels in the vicinity of Fort McMurray are projected to increase by 1–15 ppb as a result of oil sands development and associated increases in local NO_X and VOC emissions.

Projected changes in ambient O_3 levels largely correspond to projected changes in the number of exceedance days of the Canada-wide Standard numeric value for O_3 (65 ppb). Despite an overall decrease in the number of exceedance days, it was predicted that the vicinity of Vancouver, most of Alberta, southern Ontario and Quebec, and part of Atlantic Canada would still experience exceedance days under the 2015 AURAMS BAU scenario.

The AURAMS predictions highlight the fact that changes in $PM_{2.5}$ and O_3 through to 2015 are not expected to be uniform across the country, at least partially due to geographic differences in the changes in emissions. It is worth noting that projected increases in primary $PM_{2.5}$ emissions in Canada have the potential to offset some of the improvements associated with lower levels of transboundary transport. In addition, an increase in $PM_{2.5}$ levels is projected in large urbanized areas especially in winter, when the formation of nitrate particles dominates. Further analyses are needed to confirm these findings based on the most up-to-date emission projections for Canada.

Additional reductions in emissions, beyond the BAU scenario discussed above, have also been simulated by AURAMS in order to provide an opportunity to consider how $PM_{2.5}$ and O_3 can be expected to respond if additional Canadian measures were implemented. In this scenario, Canadian SO_2, NO_X, VOC and primary $PM_{2.5}$ emissions from 15 industrial sectors have been substantially reduced relative to the 2015 BAU changes, by 41%, 23%, 16% and 6%, respectively. Based on absolute emission levels, the main sectors driving these emission reductions were smelting and electricity generation for SO_2; upstream oil and gas, oil sands, and electricity generation for NO_X; upstream oil and gas and oil sands for VOCs; and wood and electricity generation for primary $PM_{2.5}$.

With these additional emission reductions, the largest declines in $PM_{2.5}$ levels would occur in the Prairies and in the Windsor–Quebec City corridor, but only lead to marginal improvements in the Maritimes and the Vancouver area. Annual decreases in $PM_{2.5}$ levels across the country corresponded largely to decreases in secondary $PM_{2.5}$ formation primarily driven by reductions in sulphate levels, particularly in the summer. More moderate reductions in $PM_{2.5}$ levels were projected in the winter as these were exclusively driven by expected reductions in nitrate, which are smaller in comparison to those expected for sulphate.

Widespread declines were projected in summer daily O_3 in the Prairies, particularly over large portions of Alberta (>5 ppb) where the largest reductions in NO_X and VOC emissions were considered in AURAMS. However, projected O_3 declines would be marginal in densely populated areas, such as Vancouver, the Windsor–Quebec City corridor and the Maritimes.

From a Canada-wide Standards perspective, the additional emission reductions discussed above would be beneficial for reducing ambient $PM_{2.5}$ and O_3 levels, including a potential decline in the number of exceedance days in densely populated areas. Nonetheless, the reductions investigated would be insufficient to bring all of Canada below the numerical Canada-wide Standards threshold for both pollutants.

Overall, the model runs indicate that the seasonality in the chemical formation of particles means that the potential for winter chemistry to dampen or even offset summer improvements resulting from emission reductions needs to be carefully considered. In addition, as SO_2 and NO_X emissions continue to decline, primary (directly emitted) particles will become relatively more important. Consequently, to realize further reductions in ambient $PM_{2.5}$, a different approach with more emphasis on primary PM emissions would need to be explored.

Relative Influence of Key Economic Sectors

Air quality model scenario analyses were performed using an ensemble of chemical transport models similar to and including AURAMS, to examine the influence of changes in emissions from six major Canadian economic sectors on ambient $PM_{2.5}$ and O_3 levels. The sectors studied were: agriculture, marine transportation, oil and gas (including oil sands), refinery and chemical, electricity generation, and residential wood combustion.

Simulated reductions in NH_3 emissions from the agricultural sector at both the continental and regional scale (i.e., Lower Fraser Valley) were estimated to have little to no effect on O_3 levels, but a moderate effect on average and median $PM_{2.5}$ levels. There are also indications that locally, the influence of NH_3 emissions reductions on short-term episodic $PM_{2.5}$ can be substantial. Initial studies on the relative influence of the marine transportation sector on $PM_{2.5}$ and O_3 ambient levels in coastal areas and along seaways were indicative of a potentially large impact. Additional studies investigating the role of marine emissions in North America[13], released after the Canadian Smog Science Assessment review period, supported the establishment of a Canada–U.S. Emission Control Area sanctioned by the International Marine Organization.

Upstream oil and gas was shown to be the major sector influencing ambient O_3 levels in Alberta where this sector's activities are most prevalent. This includes oil sands activities from which simulated increases in NO_X and VOC emissions seemed to substantially increase O_3 levels in the surrounding area. From the limited information available for $PM_{2.5}$, the upstream oil and gas sector also appeared to have a large role in determining ambient $PM_{2.5}$ levels in Alberta. A relatively lower sensitivity of O_3 to NO_X and VOCs emissions from the refinery and chemical sector in comparison to the oil and gas sector was detected in Alberta, with influences constrained to the vicinity of the source. The electricity generation sector is also a major influence on both $PM_{2.5}$ and O_3 levels in Alberta and southern Ontario.

13. ECA proposal (http://www.epa.gov/otaq/regs/nonroad/marine/ci/mepc-59-eca-proposal.pdf) available from U.S. Environmental Protection Agency's website for Oceangoing vessel regulation (http://www.epa.gov/otaq/oceanvessels.htm).

The residential wood combustion sector, studied over eastern Canada, did not exhibit any influence on O_3 levels since emissions are confined to winter periods and the scenario reviewed did not involve any changes in NO_X and VOC emissions. The influence of reductions in primary $PM_{2.5}$ emissions from this source sector on ambient $PM_{2.5}$ levels in eastern Canada was also assessed. Initial results point to a significant influence, although the magnitude of the atmospheric response needs to be further investigated.

While the role of other Canadian transportation sources was not investigated as part of the model scenario runs for the *Canadian Smog Science Assessment*, modelling studies were conducted under the Program of Energy Research and Development (PERD)[14]. The results showed that in 2002, Canadian mobile transportation emissions contributed to approximately 7% of ambient $PM_{2.5}$ in Canada. The study also showed that Canadian mobile sources showed the largest influence on ambient $PM_{2.5}$ levels in the western provinces, while the combined estimated contribution from both Canadian and U.S. mobile sources exhibited a larger influence in eastern Canada. The results highlight the importance of transboundary transport from the U.S. in defining air pollution in eastern Canada.

Factors Influencing Levels of Smog Across Canada

One of the difficulties in evaluating the impact of emission reductions on air quality is the complexity of the numerous chemical and physical processes that lead to the observed ambient levels. Aside from local emissions of precursors, smog levels are influenced by meteorology, coastal and urban effects, the local chemical mixture, natural emissions of precursors, and transboundary transport. Such factors, which vary regionally across the country, explain much of the variation seen in ambient levels of fine particulate matter ($PM_{2.5}$) and ozone (O_3). These are important to understand in order to determine the causes of and the main emission sources contributing to elevated pollutant concentrations in specific areas. Improvements in air quality models also depend upon gaining a better understanding of the factors influencing concentrations.

Meteorology

Once air pollutants have been introduced into the atmosphere their transport, transformation and deposition are largely controlled by meteorology. In the summer, there is a greater frequency of air stagnation periods along with more intense sunlight. This meteorological situation leads to the build up of local emissions and greater formation of secondary pollutants such as O_3 and secondary organic aerosol, the latter of which can include more toxic particulate species. Warmer temperatures and higher humidity can also increase the demand on electricity generation (e.g., for cooling) and enhance natural emissions of volatile organic compounds (VOC). Summer smog episodes in urban locations, especially in Ontario and Quebec, are characterized by these conditions and O_3 is most enhanced downwind of the high emission areas. The occurrence of winter stagnation episodes which are associated with even less vertical mixing and high pollution (especially $PM_{2.5}$) levels have increasingly become of interest. Several of the highest $PM_{2.5}$ levels over Ontario and Quebec have occurred during these winter stagnation periods, including the highest event during the past 10 years.

14. Program of Energy Research and Development (PERD) (http://www.nrcan.gc.ca/eneene/science/perdprde-eng.php), Spring 2009 Cycle Update, PERD Clean Transportation Systems Portfolio Program 2.1.1.

In the Prairies, the wide open spaces and relatively windy conditions keep concentrations of pollutants low despite areas of high emissions. In the winter time, higher $PM_{2.5}$ concentration events are generally linked to periods of temperature inversions (colder air below and warmer air above) that trap and concentrate pollutants close to the ground. The latter is often the case in the interior valleys of British Columbia, the Rocky Mountains, and in the Quebec valleys. Furthermore, during cold and calm periods in the winter in the Prairies, urban and industrial areas experience some of the highest ground-level concentrations of primary pollutants in Canada.

Coastal and Urban Effects

Along coastal areas, the land-water temperature contrasts have an important influence on ambient levels of $PM_{2.5}$ and O_3. In southern Ontario, the Great Lakes influence local pollution levels by restricting dispersion and deposition of primary and secondary pollutants over the water. Lake breezes then move the accumulated pollutants onshore, even to more distant coastal locations. In Atlantic Canada, the cool ocean waters can lead to air temperature inversions, particularly in spring and summer. This slows down dispersion of onshore air masses, in turn increasing the impact of local emissions on ambient smog levels in coastal cities. In this region the high frequency of coastal fog has an added influence on the formation of PM and the deposition rates as fog moves inland. Along the coast of British Columbia, land-sea wind circulation patterns, along with restricted air mass movement because of the mountainous topography, can lead to air stagnation in the summer and elevated $PM_{2.5}$ and O_3 levels. The levels can increase as the air propagates inland up the Lower Fraser Valley due to the valley becoming narrower and the increased time for photochemical formation of secondary pollutants. Pollutant transport in urban environments is subject to the density, orientation and geometry of buildings in addition to wind speed and direction and atmospheric stability.

Local Chemical Mixture

The local atmospheric chemical mixture plays a role in the formation of O_3 and can contribute to secondary $PM_{2.5}$ and thus is critical to understand in order to explain spatial and temporal variations. In urban centres, where there are high nitrogen oxides (NO_X) emissions, O_3 levels are suppressed because O_3 reacts with nitric oxide (NO) (referred to as NO titration). On a temporal scale, fewer people go to work on weekends and the number of heavy duty diesel vehicles is substantially less, therefore O_3 concentrations exhibit differences between weekends and weekdays. In many instances, O_3 formation in urban areas is VOC-limited, thus the rate of O_3 production is dependent primarily on ambient VOC concentrations. In these areas, reducing NO_X levels alone will not be as effective at reducing local ambient O_3 concentrations. Furthermore, reductions in NO_x emissions that have been occurring in Canadian cities have resulted in increases in O_3 concentrations since less of the regional O_3 that is transported into the city is removed by titration. Weighing this downside with the benefit of the lower NO_X concentrations and the other pollutants that might be affected in the urban air pollutant mixture represents an ongoing challenge to air quality management.

Interactions between ammonia (NH_3) and the reaction products of primary NO_X or sulphur dioxide (SO_2) emissions (i.e., nitrate and sulphate) affect the formation and/or composition of secondary $PM_{2.5}$. $PM_{2.5}$ levels can respond non-linearly or negatively ($PM_{2.5}$ increase with sulphate decrease) to changes in

sulphate and/or NH_3 particularly in winter and depending on the local chemical regime. The potential for this $PM_{2.5}$ non-linear or negative response was estimated to be highest in winter over southern Ontario. Examination of the monitoring data suggests this possible drawback to SO_2 emissions reductions, but more research and data are needed. For example, in rural and remote sites in southern Quebec and Ontario, SO_2 and NO_X emissions remained fairly stable from 1995 to 1999 following a decrease in the early 1990s. However, ambient particle nitrate increased from 1995 peaking between 1998 and 2001 while particle sulphate saw a continual decrease over that same time period.

Natural Emissions of Precursors

Natural sources of gaseous precursors, including wild fires, biomass burning and sea salt are important in many regions. In the summer, the sea salt (NaCl) component of $PM_{2.5}$ is significantly higher at sites in Atlantic Canada (Halifax) and the Lower Fraser Valley (Abbotsford and Burnaby) than at other sites shown on Figure 2 due to their proximity to the coast. Forest fires can be a significant contributor to ambient $PM_{2.5}$ levels. For example, it was estimated that the 2002 Quebec fires were responsible for releasing an amount equivalent to 50% of the total annual Canadian anthropogenic $PM_{2.5}$ emissions, and in British Columbia, many $PM_{2.5}$ episodes can be traced to high emissions from forest fires. Natural emissions of VOCs are important in the formation of secondary pollutants in the warm months. The overall contribution of these emissions to secondary $PM_{2.5}$ is just beginning to be characterized.

Transboundary Transport

Transboundary transport is an important source of pollution in many regions of Canada, especially Atlantic Canada, Quebec and southern Ontario. Along with high local emissions, this results in the higher-on-average $PM_{2.5}$ and O_3 levels and peak episodes seen in these areas. Weather systems typically move from west to east, carrying pollutants from the Midwest U.S., southern Ontario and the U.S. eastern seaboard to Quebec and Atlantic Canada. In the Prairies and on the west coast, while air from the south typically brings some higher levels of $PM_{2.5}$ and O_3 into the region, transboundary transport is less of an important contributor in comparison to eastern Canada. High elevation sites (above 2000 metres in altitude) have often been associated with long-range flows of air pollutants. The transport of particles and gases from other continents also influences smog levels in Canada, as is discussed in more detail in the “Emerging Issues” section of this report.

Sources of Smog Pollutants

Emissions of most smog precursors have decreased over the 1985–2006 period, except for ammonia (NH_3) (Figure 9). Smog precursor emissions projections for the year 2015 indicate continued reductions into the future, with the exception of NH_3 and volatile organic compounds (VOC).

Emissions estimates contain uncertainties which vary considerably from one sector to another and from one chemical to the next. Diffuse or open sources in particular are difficult to quantify, as the emissions can vary quite significantly both spatially and temporally. Sector-specific emissions are often estimates based

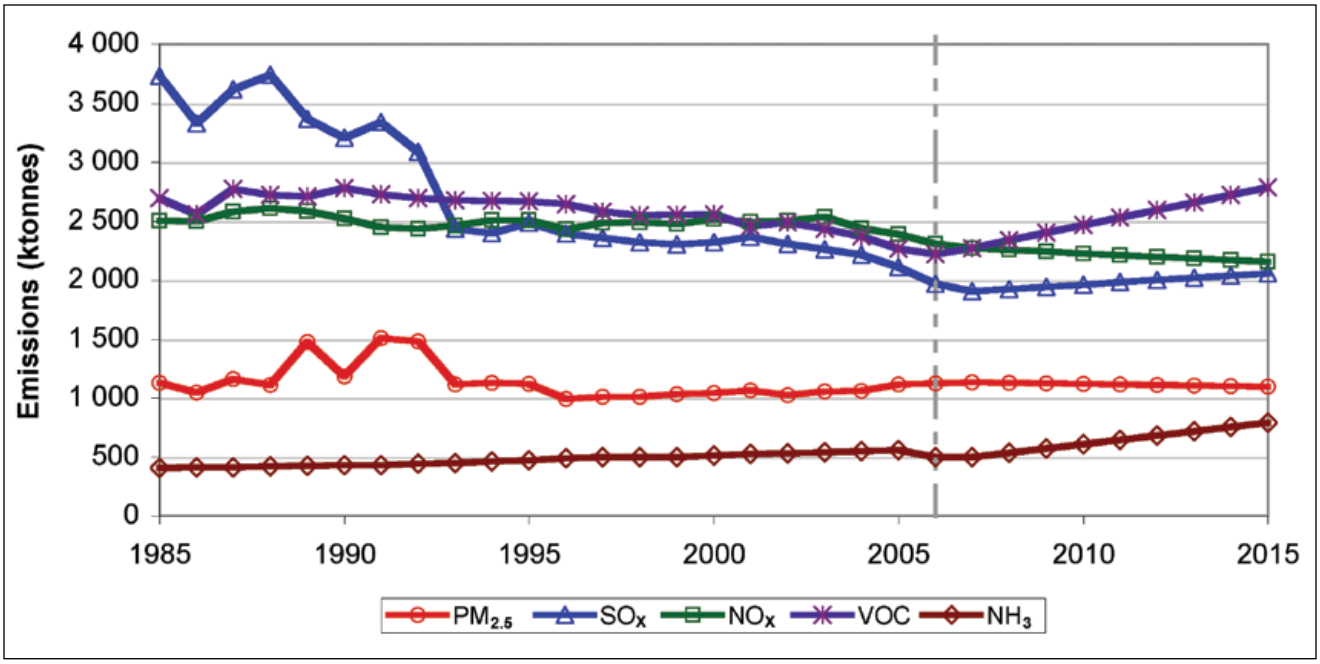


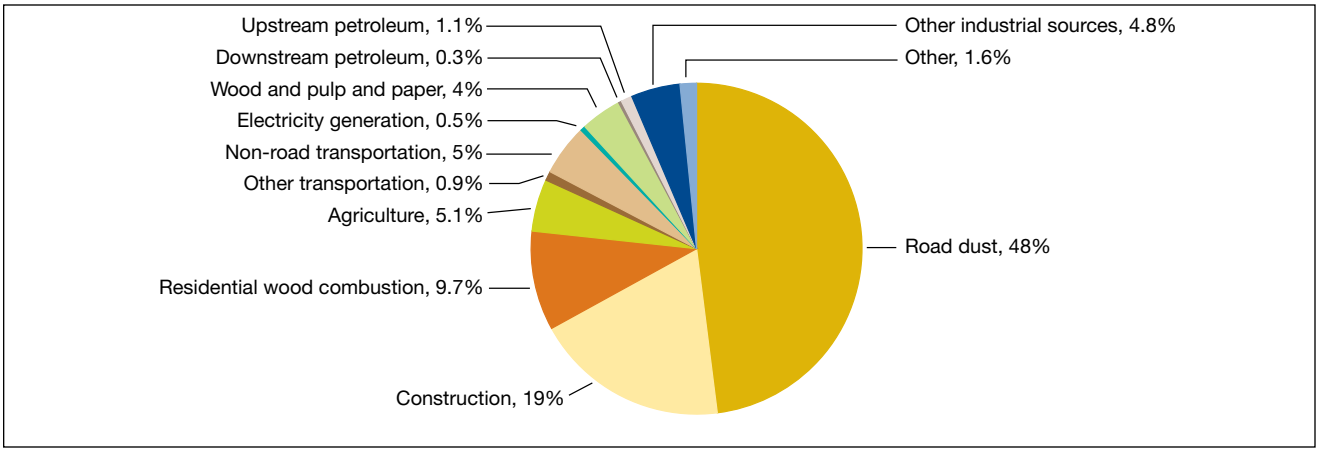
Figure 9 **Historical (1985 to 2006) and projected (2007 to 2015) anthropogenic emissions (including open sources) of smog-forming pollutants (Environment Canada, 2010)**

on an emission factors. Efforts are in place to reduce uncertainties through continuous monitoring and comparison to observations, testing and collaboration with industry. Low to high confidence rankings of emissions estimates are listed in a NARSTO (a cooperative public-private sector organization of Canada, Mexico and the United States) PM Science Assessment[15].

Primary Particulate Matter (PM)

The largest sources of primary (directly emitted) fine particulate matter ($PM_{2.5}$) are road dust and construction/demolition activity, both characterized as open sources, amounting to approximately 67% of the national total. Other important anthropogenic sources are residential wood combustion, transportation and some industrial activities such as wood processing and pulp and paper plants (Figure 10). One area of high $PM_{2.5}$ emissions density is the Windsor–Quebec City corridor resulting mainly from industrial activities and from the transportation, and residential wood combustion sectors (Figure 11). Major urban centres in western Canada and along the Edmonton–Calgary corridor are also shown as areas of high $PM_{2.5}$ emissions density, again likely the result of emissions from the transportation sector. Figure 11 includes the emissions from open anthropogenic sources, illustrating the impact of these sectors such as in the interior of British Columbia. In this area, primary $PM_{2.5}$ is a major issue of concern associated with residential woodstoves, agricultural and controlled burning, and road dust.

15. Hidy, G., Niemi, D. Pace, T., 2003. Chapter 4: Emission characterization, in: McMurry, P., M. Shepherd, J. Vickery (Eds.) *Particulate Matter Science for Policy Makers: A NARSTO Assessment*, Cambridge: Cambridge University Press. p. 147.



Note: Total 2006 national emissions of 1123 kt, not including natural sources.

Figure 10 Key sectors contributing to the 2006 $PM_{2.5}$ emissions inventory

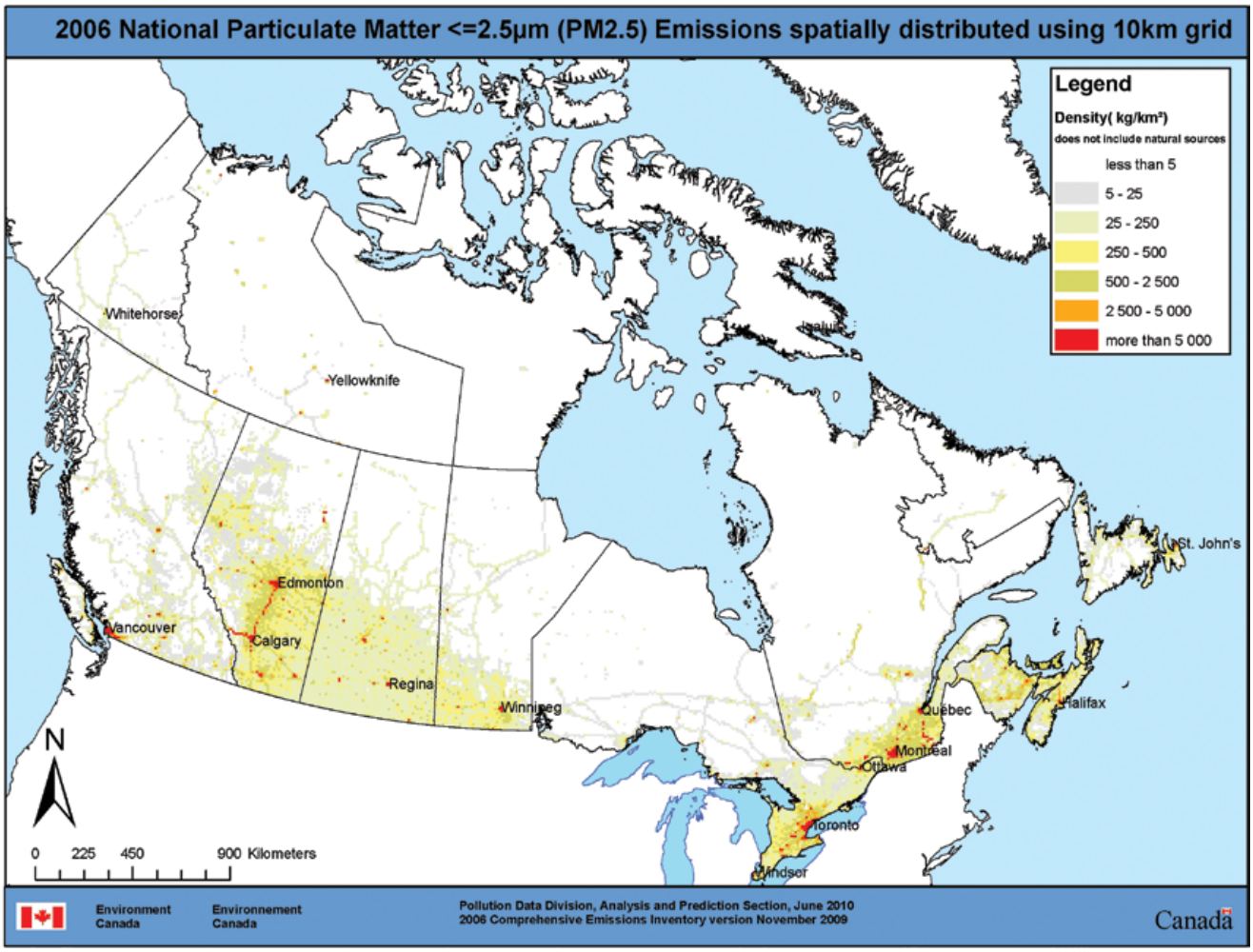


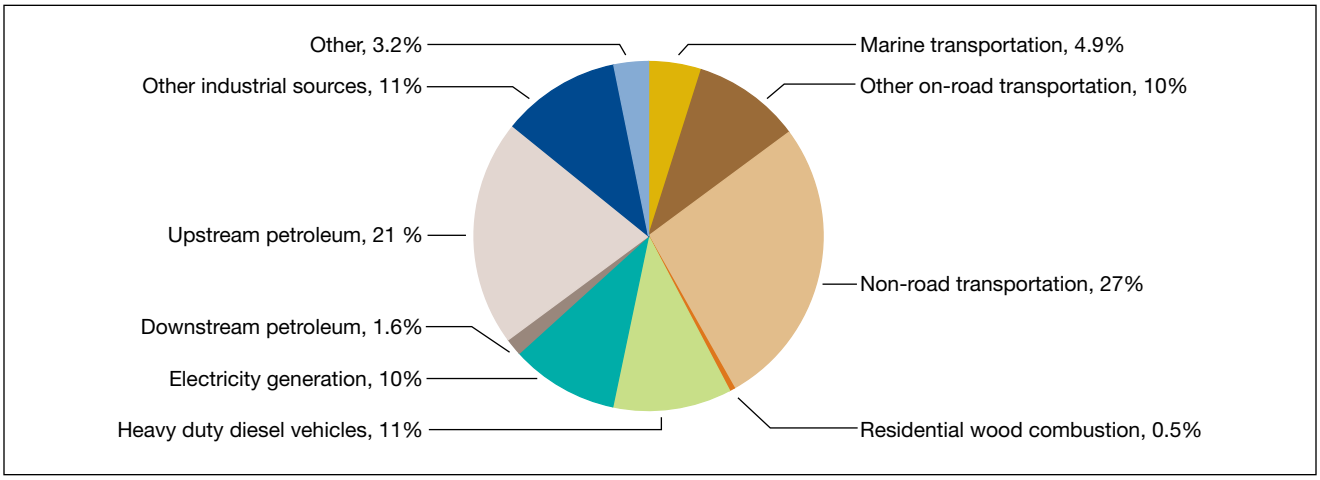
Figure 11 Density map of $PM_{2.5}$ emissions (kg km^{-2}) including open sources, in Canada for 2006

Total anthropogenic $PM_{2.5}$ emissions have remained fairly stable from 1985–2006 (Figure 9). Excluding open sources, $PM_{2.5}$ emissions have decreased by approximately 50% over that period due to reductions in emissions from the wood, pulp and paper and electricity generation sectors. Overall, anthropogenic $PM_{2.5}$ emissions (including open sources) are projected to slightly increase to 2015 due to the high proportion of road and construction dust and residential wood combustion.

Natural sources are also important contributors to primary $PM_{2.5}$ emissions. They include forest fires, windblown soil, sea salt spray, and volcanic dust. Forest fires can contribute to primary $PM_{2.5}$ emissions in the boreal forest and sea salt is an important influence along the coast.

Nitrogen Oxides (NO_x)

In Canada, transportation accounts for approximately half of national nitrogen oxides (NO_x) emissions. Upstream oil and gas and electric power generation are also important source sectors, collectively accounting for 31% of the national total (Figure 12). The highest density of NO_x emissions is in Alberta and the Windsor–Quebec City corridor (Figure 13), where the oil and gas and transportation sectors are the most prominent sources, respectively.



Note: Total 2006 national emissions of 2307 kt, not including natural sources.

Figure 12 Key sectors contributing to the 2006 NO_x emissions inventory

NO_x emissions have decreased by approximately 8% from 1985–2006 (Figure 9). This decrease is attributable to more stringent emissions regulations on the transportation and electric power generation sectors. Some of the decreases are currently being offset by greater emissions from the upstream and downstream petroleum sectors; however, an overall decreasing trend is projected to continue to 2015 (Figure 9).

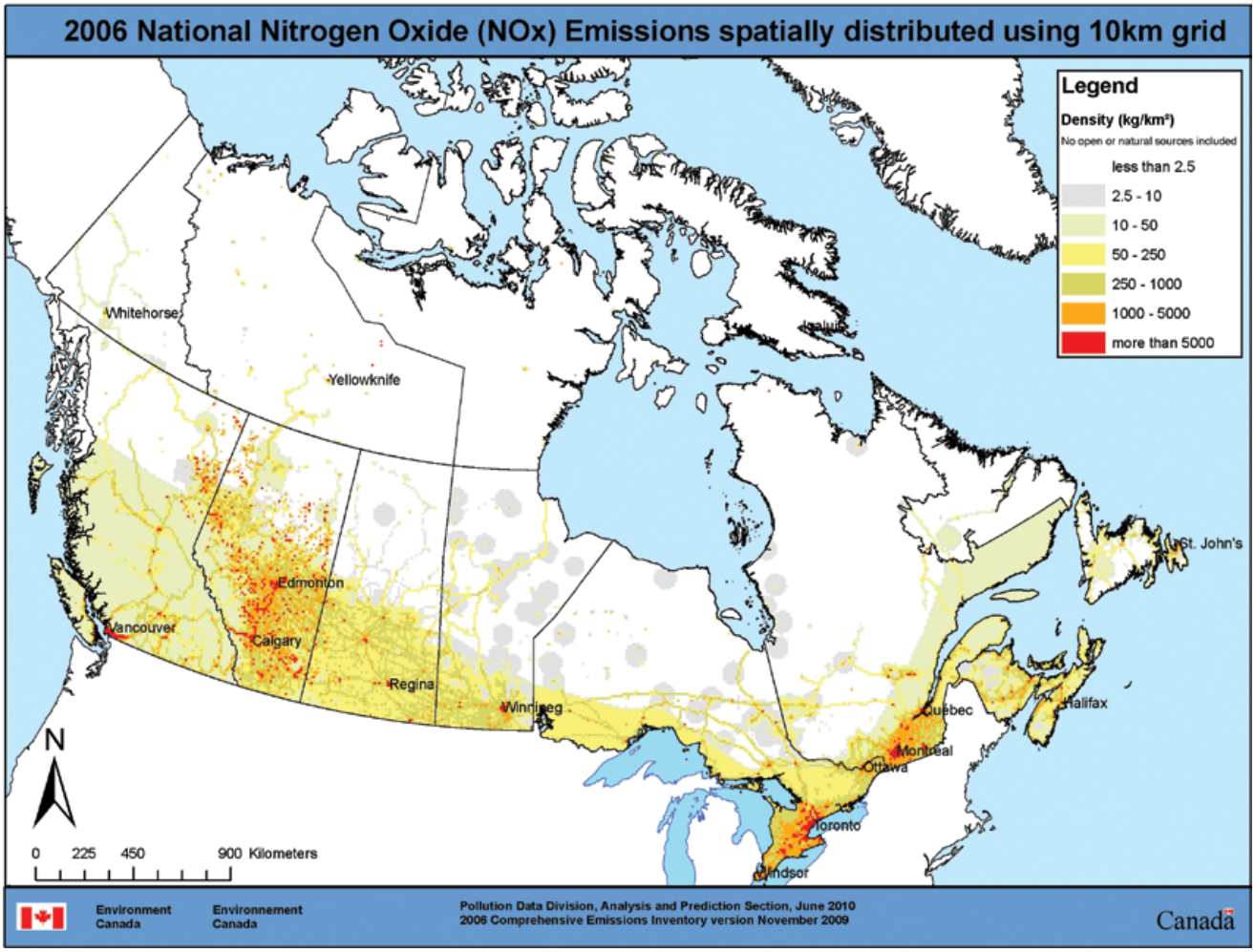
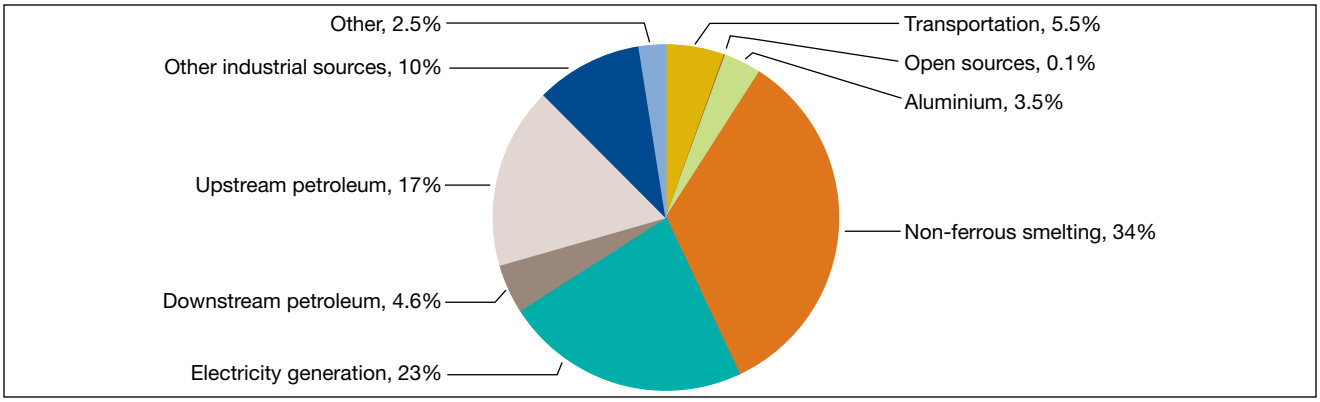


Figure 13 **Density map of NO_x emissions (kg km^{-2}) in Canada for 2006, not including open or natural sources**

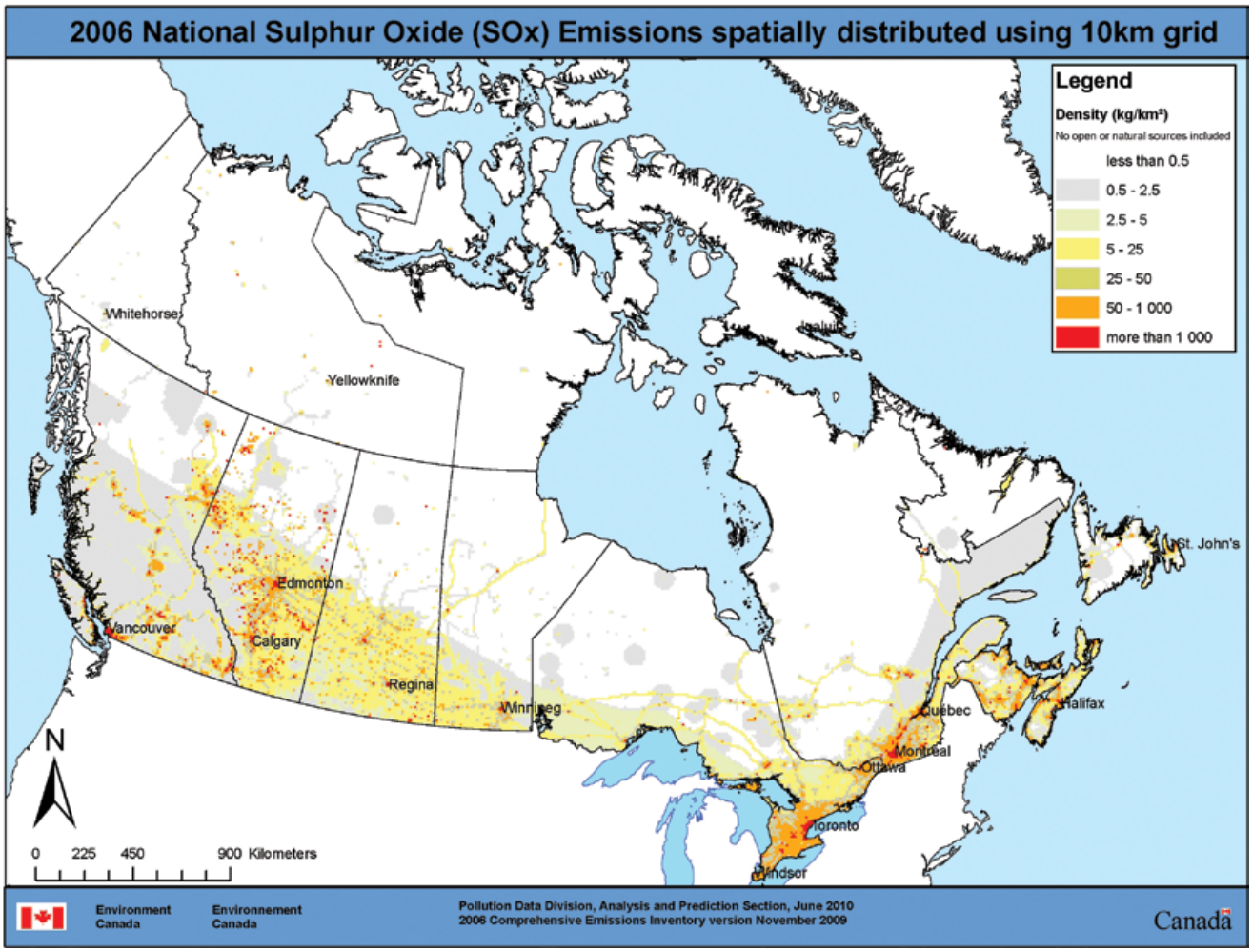
Sulphur Dioxide (SO_2)

Non-ferrous smelting is the largest source sector of national sulphur dioxide (SO_2) emissions followed by electricity generation and the upstream and downstream petroleum sectors. Collectively they account for approximately 80% of Canada's total SO_2 emissions (Figure 14). As with NO_X, the highest density of emissions occurs in the Prairie Provinces and the Windsor–Quebec City corridor (Figure 15). Sulphur dioxide emissions have decreased by over 47% from 1985–2006, with the largest decline (30%) occurring prior to 1995 (Figure 9) under the 1985 Eastern Canadian Acid Rain Program. Past reductions in SO_2 emissions are due to the regulation of the sulphur content of fuels, the phase out of coal-fired electricity generation units, and changes in industrial processes such as smelting. Since 1995, while some sectors have continued to see decreases in emissions, some of these improvements have been offset at the national level by increasing emissions from the petroleum sectors in recent years. This trend is expected to continue to 2015 (Figure 9), resulting in a gradual increase from 2006 levels although decreasing overall over the 1985–2015 period.



Note: Total 2006 national emissions of 1972 kt, not including natural sources.

Figure 14 Key sectors contributing to the 2006 SO_2 emissions inventory

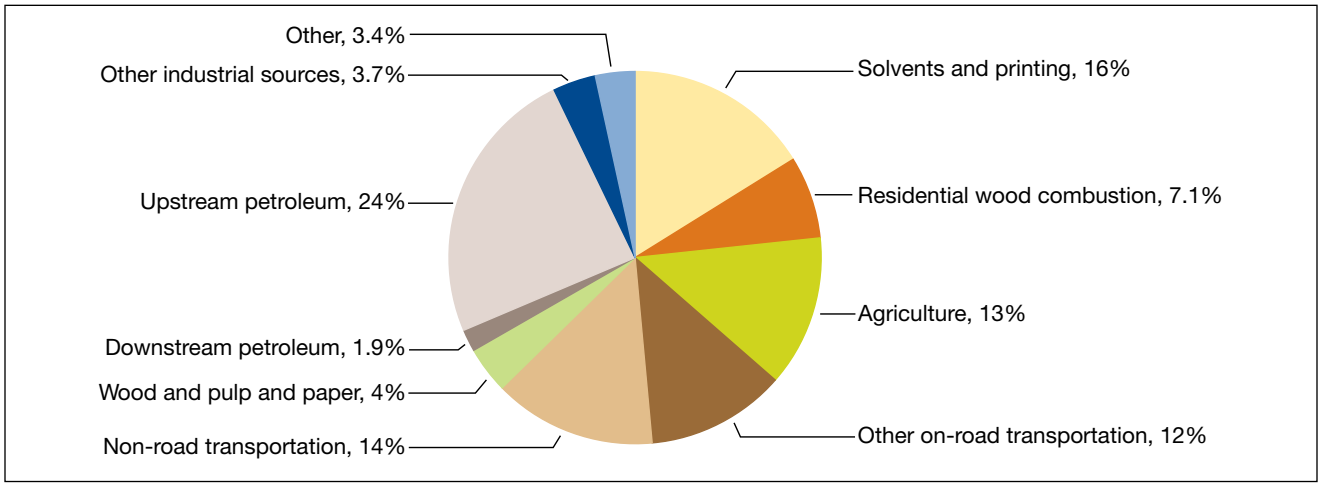


Note: The importance of H_2SO_3 emissions is relatively minimal therefore not discussed in the text.

Figure 15 Density map of sulphur oxide (primarily SO_2 and minimal contributions of H_2SO_3) emissions (kg km^{-2}) in Canada for 2006, not including open or natural sources

Volatile Organic Compounds (VOC)

Anthropogenic volatile organic compounds (VOC) emissions in Canada are primarily emitted by the upstream petroleum sector and the transportation sector, accounting for approximately half of national emissions (Figure 16). Emissions sources are concentrated in Alberta, southern Saskatchewan (primarily due to the petroleum industry and transportation sources) and in major urban areas such as along the Windsor–Quebec City corridor (Figure 17). VOC emissions have decreased by about 18% from 1995 to 2006, mainly due to reductions in the transportation and solvents and printing sectors (Figure 9). However, as is the case with NO_x emissions, some of the more recent decreases have been offset by increases in other sectors, particularly upstream petroleum. This trend is projected to continue leading to an overall increase in VOC emissions through to 2015 (Figure 9).



Note: Total 2006 national emissions of 2210 kt, not including natural sources.

Figure 16 Key sectors contributing to the 2006 VOC emissions inventory

Natural sources, including vegetation and forest fires, also contribute significantly to ambient VOC levels. This is especially the case in rural or forested areas, where VOC of natural origin are many times higher than those of anthropogenic sources. However, the importance of natural versus anthropogenic VOC as an O_3 precursor, even in rural areas, is dependent upon overall reactivity, in other words how efficiently the individual VOC reacts to form O_3. VOC reactivity may have implications for emissions reductions strategies, where individual compounds are considered rather than treating all VOC equally. While this approach has been considered, more research and monitoring of individual VOC species are required.

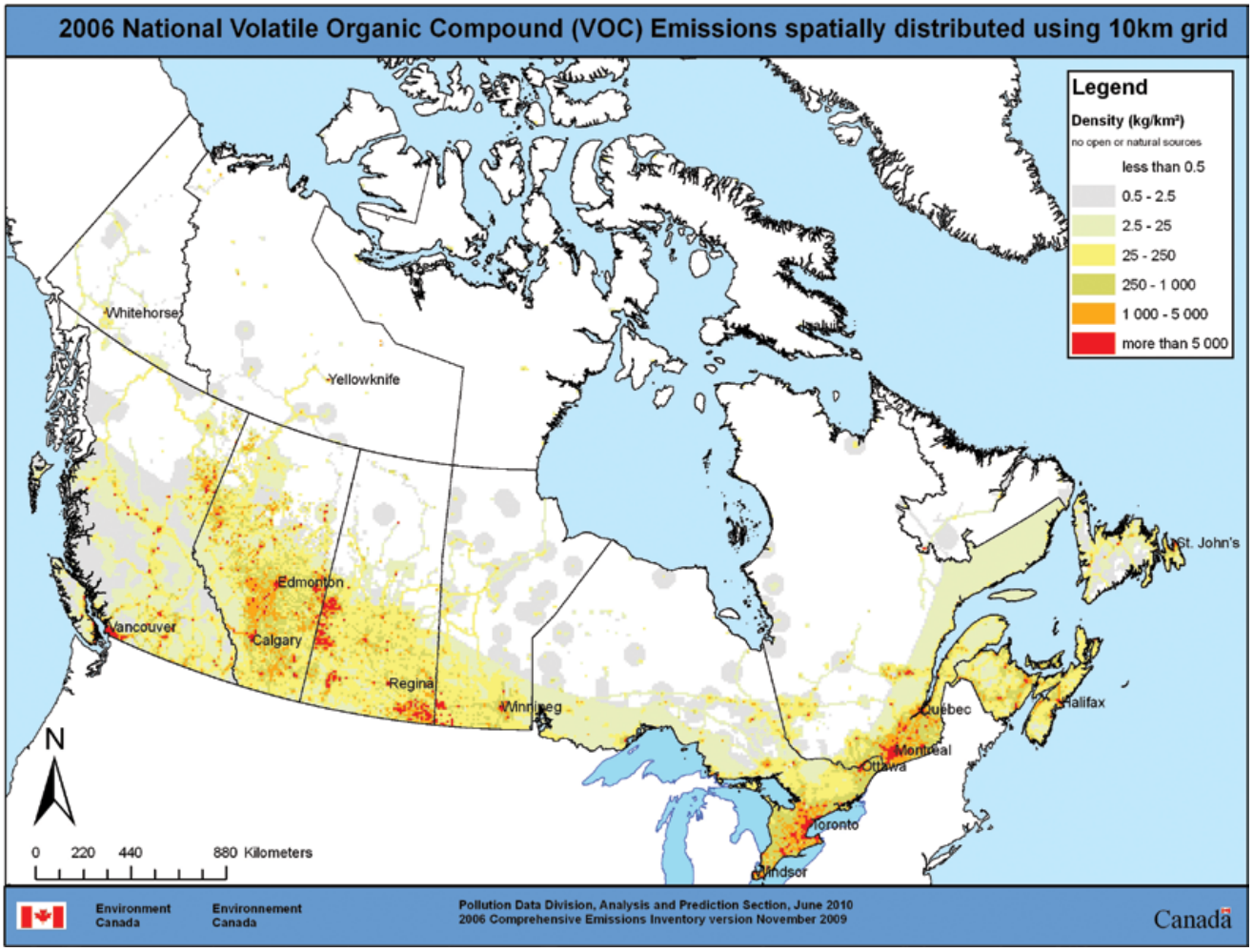
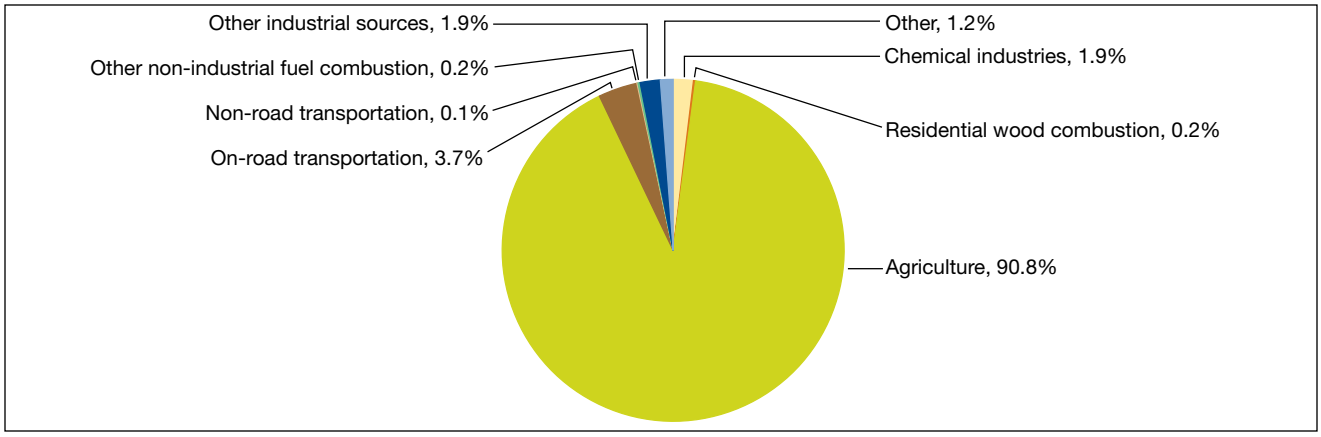


Figure 17 **Density map of VOC emissions (kg km^{-2}) in Canada for 2006, not including open or natural sources**

Ammonia (NH_3)

The agricultural sector is the most important contributor of ammonia (NH_3), accounting for 90% of national emissions (Figure 18). Areas of intense agricultural activity include southern Ontario and Quebec, southern British Columbia, Alberta and Saskatchewan (Figure 19). In Canada, NH_3 emissions have increased by about 22% over the period of 1985–2006 and are generally expected to increase through 2015 as a result of economic growth and the demand for agricultural outputs, including food and biomass for energy production and fuels (Figure 9).



Note: Total 2006 national emissions of 550 kt, not including natural sources.

Figure 18 Key sectors contributing to the 2006 NH_3 emissions inventory

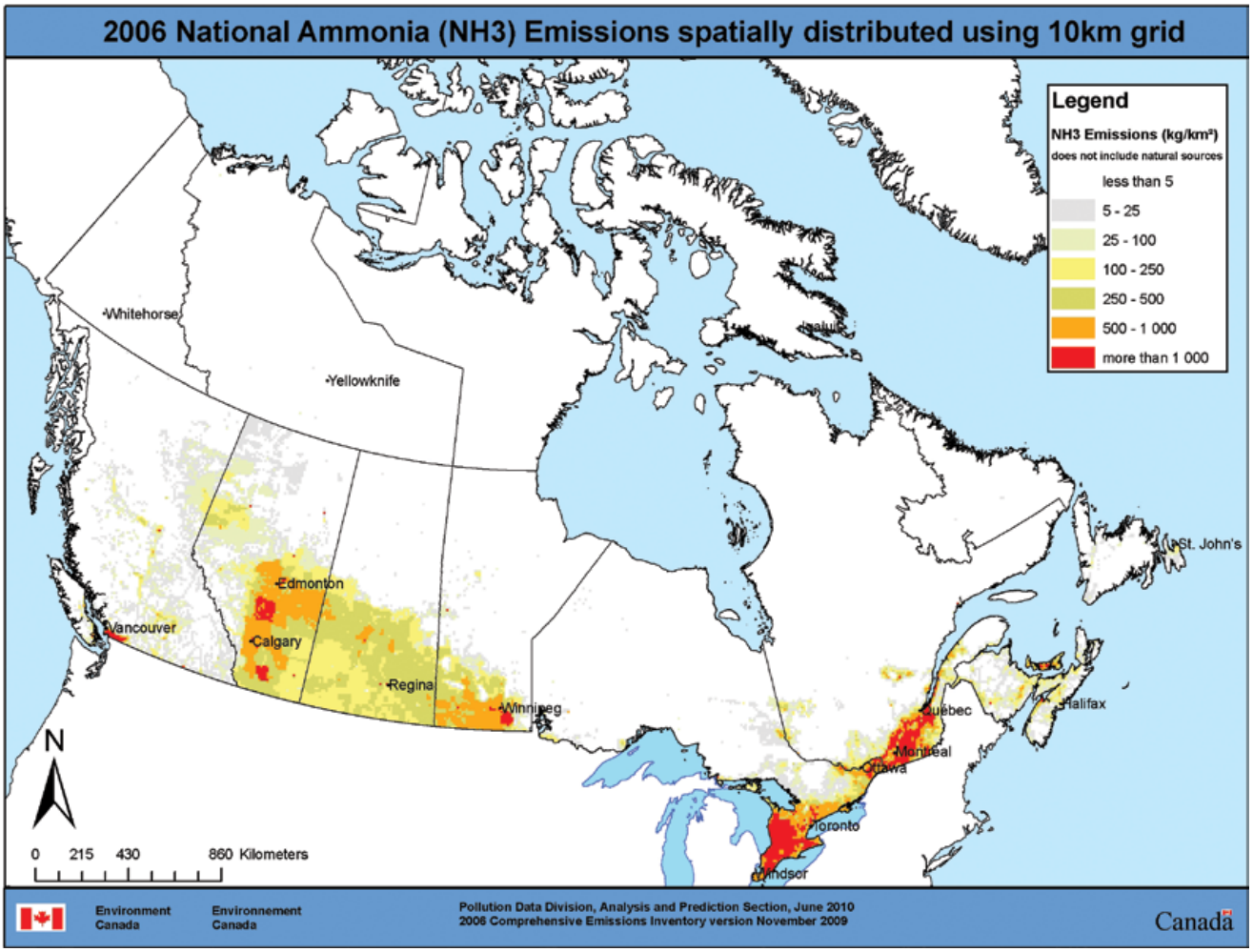


Figure 19 Density map of ammonia (NH_3) emissions (kg km^{-2}) in Canada for 2006 with open but no natural sources

Emissions Sources in the U.S.

Emissions sources in the U.S. are also important contributors to smog in Canada. In general, primary $PM_{2.5}$ and precursor emissions in the U.S. have decreased over the period of 1990–2006 and are projected to continue to decrease. U.S. emissions of primary $PM_{2.5}$ have decreased by about 65% since 1990 through reductions achieved in most source sectors. NO_X emissions have been reduced by 36%, primarily through reductions from transportation, electrical utility fuel and industrial fuel combustion sources. Much of the decrease can be attributed to the implementation in 1990 of the NO_X Budget Trading Program in the northeastern U.S. SO_2 emissions have decreased by approximately 50% from 1990–2006, primarily from electrical utility and industrial fuel combustion sources, which account for the majority of the emissions. VOC emissions have decreased by 34%, with the majority of the reductions from transportation and solvent sources.

Projections of U.S. emissions to 2015 also show a general decrease in gaseous precursor emissions in most regions of the country. In the northeastern states, there is a projected overall decrease in emissions densities in most of the areas, which would have a beneficial impact on ambient levels of smog and precursor gases in Canada.

Emerging Issues

Climate Change

A changing climate can impact air quality in a number of ways, including: changing chemical reaction rates because of an increase in temperature and water vapour; changing the distribution of meteorological conditions which affect transport of air pollutants and location of air pollution episodes; modifications in global circulation, again affecting distribution of air pollutants; changing emissions of natural precursor gases; decreasing cloudiness which enhances smog production; and changes to the frequency, seasonality and intensity of forest fires.

The impact of future climate change on regional air quality is not yet well quantified as there are many possible future emissions and climate scenarios, as well as many modelling methodologies. The complexity of the linkages between air quality and climate change, and interactions between the two, is an area of continuing research. Preliminary studies for Canada, however, indicate that climate change may increase O_3 levels with the greatest impact occurring in areas that already experience high levels, such as the Windsor–Quebec City corridor. The effect on O_3 would not affect all areas across the country consistently, as certain regions may experience local reductions in O_3, depending on changes to local emissions and meteorology.

Studies on climate change effects on PM show increases in both $PM_{2.5}$ concentration and number of peak days, but a smaller impact on the daily maximum 24-hour $PM_{2.5}$ concentration, and a smaller increase in the number of exceedance days in comparison to O_3. A future scenario involving both climate change and projected future emissions suggests an increase in PM concentration, as does a future scenario with climate change only (no changes in emissions), though to a lesser degree.

When the above scenarios were analysed for health impacts, there were clear indications that climate change impacts on air quality would result in increased adverse health effects. However, the scenarios used were relatively simplistic and more complex further analysis is required to determine overall impact and regional variation[16].

The combined impacts of O_3 and carbon dioxide (CO_2) on crop yields and ecosystems is also an emerging concern since there is uncertainty about the net effect on vegetation of elevated atmospheric CO_2, warmer temperatures, possibly elevated O_3 levels as well as other environmental changes such as changes in precipitation and nutrient availability. Short-term exposure to increased levels of O_3 can decrease plant species' ability to respond to increased levels of CO_2, decreasing net primary productivity. Recent studies on crop yields have also shown that the joint effects of both elevated O_3 and CO_2 can lead to crop yield reductions, with overall decreases in net biomass. Recent long-term studies also indicate that the differing sensitivity of species to various O_3 and CO_2 levels can alter species composition in plant communities and can either enhance or slow the conversion of communities towards less sensitive species.

Intercontinental Transport

Trans-Pacific transport of pollutants from Asia into North America can occur through the winter and early spring, mainly resulting in increased ambient levels from mid-latitudes to the Arctic where the contribution reaches its maximum.

The contribution of O_3 from Asian anthropogenic emissions to the annual average ambient concentration in North America has been estimated to be in the range of 2–5 ppbv[17] in 1997. Other similar studies estimate a 10% increase in Asian emissions would increase annual average surface O_3 in the U.S. by 0.1–0.2 ppbv, and more in regions of higher elevation. Short-term episodic influences could increase regional O_3 levels by more than 10 ppbv.

Transportation of dust aerosols across the Pacific has been observed through satellite data for many years and tends to be episodic in nature. However, the contribution of other transported species to local $PM_{2.5}$ levels is harder to quantify and observe. The annual average contribution of intercontinental transport to total PM levels in Canada has been estimated to be in the range of 0.1 μg m^{-3} in annual PM concentration in the form of sulphate, black carbon and organic carbon. Model simulations show that Asian sulphur emissions have little influence on surface sulphate levels in Canada, with the exception of the west coast; however, the contribution is significant at higher surface altitudes across the country. Ongoing research on the effects of intercontinental transport on domestic air quality includes reconciling the range of model results, and studying the climate change-air quality interactions resulting from the increased loadings of fine particles into the atmospheric column.

Satellite based measurements have emerged in the last decade as a new means of studying air pollution, and their use is expected to grow. Current generation satellites are capable of estimating surface air pollution concentrations, and have been used to track long-range transport events on the regional and

16. Health Canada, 2008. *Human Health in a Changing Climate: A Canadian Assessment of Vulnerabilities and Adaptive Capacity.* Health Canada, Ottawa.
17. ppbv refers to parts per billion by volume, a unit primarily used to describe concentrations in air as a volume fraction, and differs from ppb which describes concentrations on a weight to weight ratio.

global scales. These observations may also be used to provide estimates of emissions in parts of the world where surface observation data is unavailable. Additionally, this method is being investigated for its utility in representing population exposure in epidemiological studies of health effects.

Knowledge Gaps

The state of knowledge of smog science in Canada has greatly evolved over the last 10 years and is continuing to evolve. Knowledge gaps still remain, however, particularly with respect to the processing of emissions inventories, ambient monitoring, the understanding of some key smog formation processes, model sensitivity to all these factors, and the understanding of the dose-response relationships for ecosystem and human health impacts.

Sources and/or components—Ambient fine particulate matter (PM) concentrations continue to be largely characterized on a mass basis, even though it is known that the composition of PM varies between sources, regions and seasons. While to date there are indications that differential toxicity exists as a function of PM composition, nonetheless the evidence remains that PM of many types and from many sources exhibits toxicity, and at this point it is not possible to identify any component or source of PM as non-toxic.

Size fractions—Fine PM is currently the focus of regulatory programs, due to the preponderance of evidence and the depositional characteristics of this size fraction in the human lung. Evidence remains however that coarse particles 2.5–10 microns exert adverse effects. Although such effects of this larger size fraction are likely less than those seen in the 2.5 μm range, it is not clear if the risk management of $PM_{2.5}$ accounts for PM_{10} related effects.

Of much greater potential importance are the health effects of ultra-fine particles. Such particles are less than 0.1 μm in size, are highly reactive and, while accounting for relatively little mass, constitute a large number of particles and a large surface area (implying a corresponding potential for active chemistry). For these reasons, many in the health science community believe that a significant proportion of the health effects attributed to PM arise from ultra-fine particles. However, this size range can be challenging to measure. Additionally, since they are not distributed uniformly over large areas, fixed ambient monitors are not useful in estimating population exposure, and cannot therefore be used in epidemiological investigations.

Exposure—While the ambient monitoring system has been shown to adequately represent population exposure for the purposes of conducting epidemiological studies, more precise understanding of exposure for individuals (especially within susceptible groups) is lacking. As well, as a better understanding of the toxicity of components and sources evolves, specific analysis will be needed to understand the dynamics of exposure to these sources/components.

Concentration-response relationship—The shape of the concentration-response function, including investigation of potential population threshold levels, has important implications for risk management and for estimates of risk from air pollution. Most of the available studies and analyses continue to report

no clear threshold between ambient concentrations of $PM_{2.5}$ or ozone (O_3) and health endpoints such as premature mortality, hospital admissions and emergency department visits.

Role of co-pollutants—The extent to which other co-pollutants in ambient air may modify or contribute to the associations between ambient $PM_{2.5}$ or O_3 and morbidity or mortality continues to be important in the interpretation of the epidemiological findings.

Exposure durations of concern—Historically, most epidemiological research has investigated associations of health outcomes with ambient PM and O_3 concentrations measured over 24 hours and several hours, respectively, with lags of up to 2 days. However, there are indications that some health outcomes are more strongly associated with shorter durations of exposure (e.g., one hour), and that risk estimates are often increased by using the average concentration over several days as the exposure metric rather than a single day lag, though the available data are limited.

At the other end of the temporal spectrum, there is only limited information on the effects of exposure to PM and O_3 at longer time scales, including over entire seasons, years, or over multiple years.

Inflammation/oxidation and range of effects—While considerable knowledge has been developed concerning inflammation and oxidative stress caused by both PM and O_3 the degree to which these mechanisms underlie the various observed effects, and the implications of these mechanisms for other health outcomes, is not entirely clear. Since these mechanisms play a role in most diseases, it is theoretically possible that these pollutants could be implicated in a much wider range of adverse effects than is currently accepted. Indeed, there are indications in the literature of effects of these pollutants beyond the respiratory and cardiac systems, with a number of reports of increases in risk of reproductive outcomes and specific diseases such as diabetes, and isolated reports of increases in more novel inflammatory diseases (e.g., appendicitis, inflammatory bowl disease).

Exacerbation versus disease development or progression—It is widely considered that ambient $PM_{2.5}$ and O_3 can exacerbate pre-existing diseases (e.g., asthma). However, the epidemiological associations with chronic exposure imply that these pollutants can also contribute to the development or progression of disease, and toxicological evidence demonstrates the mechanisms which could be involved. The relative impact of exacerbation of existing disease versus the development and progression of new disease has not been established, and could have enormous public health implications if these pollutants were linked to the development of prevalent diseases with serious health outcomes.

Susceptible populations—Improving our understanding of subpopulations that are more susceptible to the adverse effects of ambient PM and O_3 is important to inform risk management to reduce public health risks from these pollutants. While persons with pre-existing cardiac and respiratory disease are recognized as susceptible populations, additional health conditions that may confer susceptibility to these pollutants continue to be identified. There are also important uncertainties with respect to the key windows of development during which PM and O_3 may cause respiratory-related effects in children, another susceptible group of the population. Finally, animal and human studies continue to reveal the presence of specific genotypes that are more affected by these pollutants than the general population, and thereby provide additional information on susceptibility and on pathways and mechanisms of action.

While emissions inventories are constantly being refined, they generally do not adequately represent the actual emissions at any given time, particularly as levels of uncertainty can be high depending on sources and methods used for estimating emissions. In turn, there are often discrepancies between concentrations estimated from emissions inventory and ambient observations. In particular, there is room for improvement in the quantification of non-point sources such as dust emissions, $PM_{2.5}$ and volatile organic compounds (VOC) species, ammonia (NH_3) and spatially-allocated mobile emissions in major urban areas. The contribution of expanding emissions sectors (e.g., offshore oil and gas, marine transportation) is uncertain, as is the magnitude of impact of emissions from local industries and/or residential wood combustion on smaller communities and rural areas.

Although there are considerable observations available from monitoring and special field studies across Canada, there are areas where more measurements are needed. One of the pressing needs is for more measurement sites required to draw conclusions about the chemical composition and temporal trends of PM across the country, such as during winter smog events in Alberta and changes in visibility in high interest areas such as national parks. In addition, it is currently difficult to assess the baseline $PM_{2.5}$ temporal trend due to insufficient long-term $PM_{2.5}$ measurements at regionally representative sites.

While the spatial coverage of O_3 monitoring sites is more extensive than other pollutants, there are still limitations particularly where spatial patterns are complex (i.e., over and near the Great Lakes, over the southern Atlantic region, and above the layer of air adjacent to the ground). There are insufficient O_3 measurements in rural and remote areas that are potentially impacted by human activities such as downwind of Edmonton and Calgary and in the eastern parts of the Lower Fraser Valley. There are even greater limitations on the ability to characterize ambient levels of individual VOC species and NO_2 due to gaps in the existing measurement network.

There is a lack of in-depth understanding of many PM and O_3 processes and mechanisms as reductions in precursor emissions do not always translate into the anticipated reductions in ambient concentrations of $PM_{2.5}$ and O_3 (referred to as non-linearity in the processes). For example, reducing sulphur dioxide (SO_2) emissions may in certain circumstances (e.g., in the presence of high concentrations of NH_3) result in an increase in $PM_{2.5}$ and as precursor emissions continue to decline in North America, the role of NH_3 in $PM_{2.5}$ formation may increasingly become more important. It is also often very difficult to predict ambient O_3 concentrations due to the complexities of O_3 formation associated with ambient VOC/NO_X concentration ratios and varying reactivity of dominant VOC in an airshed.

Air quality model performance is greatly influenced by uncertainties in emissions inventories and the state of understanding of atmospheric physical and chemical processes. In general, there is less confidence in modelling secondary pollutants and predictions at finer spatial and temporal scales. Some of the key model inputs that need clarification include the understanding in the sources, characteristics and processes of various organic components of $PM_{2.5}$. A major gap is also the effect of climate change on the formation of $PM_{2.5}$ and O_3 since climate change has the potential to affect conclusions from modelling analyses of the efficacy of proposed emissions regulations.

Finally, there are also important knowledge gaps in the understanding of the effects of PM and O_3 on human and ecosystem health and how these affect the socio-economic welfare of Canadians. There

is a lack of understanding in linking multi-pollutant concentrations (i.e., pollutants beyond $PM_{2.5}$ and O_3) and potential changes in toxicity to population exposure. Vegetation exposure-response functions, of the greatest use to regulators, are still in short supply in the published literature. There is a clear paucity of research available on the effects of PM and O_3 on an ecosystem level, especially with the direct effects of exposure on wildlife species.

Recommendations for Future Research

Health Effects

Sources/components—Research directed at source characterization, exposure, epidemiology, and toxicology of the different fine particulate matter (PM) components, properties and sources that may be more strongly related to health effects would reduce the uncertainties in estimating risks associated with exposure to PM.

Size fractions—Although effects of the coarse PM_{10} size fraction are likely less than those seen in the 2.5 µm range, it is not clear if the risk management of $PM_{2.5}$ would account for PM_{10} related effects, and thus continued investigation of such effects is warranted. For the ultra-fine fraction, more precise exposure-response studies are revealing significant effects but more work is needed to determine the relative toxicity and health impacts of ultra-fine particles versus other particles. As well, because the standard monitoring system does not represent this fraction well, specific studies to understand source-exposure relationships are also necessary.

Exposure—Additional research into personal and population exposures to PM and ozone (O_3), especially under conditions relevant to Canada, would enhance our understanding of exposure and the interpretation of the epidemiological evidence for these pollutants. Work could be directed at investigating the extent of exposures to various PM components and size fractions, as well as the determinants of these exposures, including source contributions, personal attributes and activities, and building-related factors. Further work to address the size of the measurement error introduced by the difference between concentrations at a central monitor and actual pollutant exposures would help in evaluating the associated error in the risk estimates of pollutant effects in epidemiological studies.

Concentration-response relationship—Further research into concentration-response functions for air pollution related morbidity and mortality, including studies at the relatively low ambient levels of $PM_{2.5}$ and O_3 measured in Canada, with a wider range of health endpoints, and (as more information becomes available) fine PM components and sources would serve to reduce uncertainties in characterizing risk.

Role of co-pollutants in health effects—Research focused on furthering our understanding in this area would inform our ability to discern the effects of these substances in the broad mix of ambient pollutants and help to direct risk management of ambient air pollution sources.

Exposure durations of concern—While there are a few studies which specifically address this subject, they do not cover most endpoints of concern, nor do they cover all the possible time frames of concern. Since

toxicological studies indicate potential effects running the full gamut of ultra acute to chronic timeframes, further research to examine human responses in the field and in controlled settings is warranted and could provide additional time frames relevant for risk management. Research examining the very short time course events related to inflammatory mechanisms and especially as relates to cardiac events would be highly useful. Longitudinal studies of health outcomes in relation to chronic exposures to these pollutants have the potential to yield particularly valuable information on the health effects of air pollution. Such studies should also attempt to incorporate design features which would provide information on the degree to which air pollution plays a role in the instigation and progression of disease.

Inflammation/oxidation and range of effects—Since these mechanisms appear to be one of the fundamental aspects of the health effects of both PM and O_3, research to better understand the processes, especially but not solely related to those with pre-existing disease or susceptibilities is warranted. Given the recent appearance of more novel disease endpoints in the air pollution literature (e.g., appendicitis, bowel diseases), examination of existing cohorts for inflammatory diseases could provide new and important insights as to the impact of air pollution.

Susceptible populations—As we move towards the development of risk management tools for individuals [i.e., such tools as the Air Quality Health Index (AQHI)], the understanding of the susceptibility of specific sub-groups of the population is warranted in order to provide more targeted messaging, and to provide better estimates of how air pollution affects quality of life. As well, the appearance of information indicating that those with specific genetic makeup are more susceptible to air pollution (but are otherwise perfectly healthy) warrants considerable attention in order to better understand how such genetic factors influence susceptibility and overall population health impact.

Environmental Effects

Further research on exposure-response relationships for Canada-relevant plant species is recommended. This could entail a meta-analysis (combination of data from several studies to produce a single estimate) of existing information and the development of indices predicting dose-response functions for plant species that have not been directly investigated based on previously established plant O_3 sensitivities. The use of an O_3 flux approach to estimate plant uptake, in particular the quantification of plant defences to O_3 damage, is recommended for North American species and environments.

Continued studies of the linkages between O_3 and other pollutants (e.g., atmospheric carbon dioxide) in the context of impacts to forest growth and productivity are recommended. In addition, further research on the effects of PM and O_3 on an ecosystem level would better evaluate the broader ecosystem risks associated with PM and O_3 exposure to sensitive or endangered species, as well as the impacts on wildlife species.

Moving ahead, there is a need for more Canadian-based primary valuation research on the negative effects of smog on ecosystems. Further research into both the scientific basis underlying physical drivers of the impact of smog and how these impacts are valued by Canadians will help formulate effective smog abatement policies. There is a need to develop a framework capable of capturing non-linear relationships among emissions, air quality and impacts on humans and the environment through an integrated assessment approach with greater coordination among the various fields of expertise.

Emissions

Improving the spatial and temporal quantification of precursor emissions would serve to resolve some of the discrepancies between measured ambient levels of smog and precursors and emissions inventories. Areas that would benefit from additional work include the identification and quantification of source regions of intercontinental transport (e.g., Asia), of sources and species of natural and anthropogenic primary $PM_{2.5}$ and volatile organic compounds (VOC) emissions, of sources of mobile emissions (e.g., aviation, rail, biodiesel) such as through the use of $PM_{2.5}$ compounds that mark specific emission sources. Historical emissions inventories and current non-point source emissions estimates need refinement. The latter is important in air quality model applications as it is often difficult to reconcile observed values with current emissions estimates (e.g., open sources of PM from road dust, agriculture, construction and mining).

Monitoring and Trend Analysis

As PM and O_3 precursor emissions continue to decline in North America, the influence of background PM and O_3 concentrations on local air quality may become increasingly important. Additional measurements, particularly for $PM_{2.5}$ at remote (or background), high-elevation, and Arctic sites are recommended. In addition, further research is recommended to understand the contribution of intercontinentally transported pollutants to ambient concentrations in Canada. The establishment of more $PM_{2.5}$ speciation sites across Canada would facilitate tracking the fraction of chemicals transported from other continents to determine which play the largest role in contributing to local $PM_{2.5}$ concentrations.

Measurement sites are recommended near roads to better track motor vehicle emissions as these appear to dominate observed ambient PM and O_3 precursors and can provide a platform to carry out special studies. A sampling shortcoming is the bias in nitrogen oxides (NO_X) measurements, leading to over-prediction of NO_X in rural areas, and which needs to be quantified through urban/rural sampling of NO_X and other nitrogen species.

Carefully designed long- and short-term studies, which may include measurements of compounds that mark specific emissions sources, are required to minimize some of the discrepancies between emission inventory estimates and observations of ambient PM and VOC (both anthropogenic and natural). Considerable opportunity exists to extend remote sensing applications currently being developed elsewhere to Canadian-specific objectives to improve observations of surface air quality, provide constraints on emissions and track long range transport.

To understand the role of ammonia (NH_3) in PM formation, especially as emissions of other precursors are expected to decline, more measurements of NH_3 and PM are recommended in areas downwind of agriculturally intensive areas both in Canada and the U.S. This is required in order to investigate the sensitivity of $PM_{2.5}$ to changing NH_3 emissions as well as to quantify the transboundary flow of these emissions.

Understanding of Processes

There is a continued need to increase the understanding of processes affecting smog formation, as these feed into air quality models used to assess the impacts of proposed risk management measures. A combination of both field and laboratory studies along with integrated analyses of existing monitoring data are required to elucidate these non-linear chemical, physical and meteorological processes. In particular,

these include the role of different VOC species, sea salt, local meteorology, local atmospheric chemical composition, cloud, urban, and coastal influence on smog formation. As well, there is a need to better characterize local/regional scale processes for inclusion into higher spatial resolution air quality models for human exposure assessments.

Further research is needed to better understand the relationship between O_3, sulphur, dust and organic matter during intercontinental transport for better integration of global results into a regional modelling system to assess the intercontinental influence in more detail.

Atmospheric turbulence plays a key role in influencing air pollutant concentrations, yet the mechanisms of turbulence in urban settings are still being studied. Further observations and evaluation of models of turbulence are required to improve accuracy in air pollution model simulation, particularly in urban and shoreline environments.

Finally, systematic studies tailored to specific smog formation processes of importance at a regional-scale at adequate resolution are needed, while simultaneously accounting for both climate change and changing regional emissions.

Modelling

As scenario analyses become increasingly relied upon, continued efforts to develop and evaluate air quality models will help improve their accuracy and increase confidence in the guidance provided by scenario studies. In particular, the improvement in model representations of emissions sources and the processes affecting the production of secondary aerosols is needed. Model development and evaluations are dependent on the availability of pertinent observational data of O_3, $PM_{2.5}$ mass, and the chemical composition of $PM_{2.5}$ and other important precursors from monitoring and field studies. These data, along with satellite based observations, can also be assimilated into models for improving forecasts and correspondingly, improve emissions estimates through inverse modelling. Continued research into these methods is recommended.

The influence of individual emissions source sectors on ambient PM and O_3 levels on a national scale, as estimated by modelling scenarios, needs to be verified and in some cases further investigated through more comprehensive studies. Aside from sources already routinely monitored and for which emissions data are available, approaches to include intermittent sources (e.g., wildfires, wind, dust) in model analyses should also be considered. The significance of these sources will likely increase as precursor emissions are reduced and climate change influences become more pronounced.

Studies where models can exchange information interactively between global, regional and local scales are also recommended to obtain complete spatial information on air quality, including the background influence and better quantification of the influence of point sources, local sources and smog formation processes.

A number of model evaluation approaches are available, including operational evaluation (comparison of model prediction against routine monitoring data), dynamic evaluation (model response to changes in meteorology or emissions) and diagnostic evaluation (simulation of atmospheric processes), all of which

require continued development. The latter two are of particular importance in ensuring model credibility when evaluating emission control scenarios. Probabilistic modelling approaches based on uncertainties in model inputs and formulations should also be considered for assessing model uncertainties in forecasting and policy applications.

Models are currently the best available means of estimating the effects of climate change on air quality. There have been relatively few studies simulating both climate change and air quality, and additional work is needed in this area. An integral part of characterizing the impact of climate change will be initially to understand the inherent variability induced by year-to-year changes in meteorology. The development of multi-year air quality simulations to characterize inter-annual variability is especially important in the investigation of the potential influence of climate change on the conclusions drawn from modelling analyses evaluating the efficacy of proposed emissions regulations. Furthermore, multi-year simulations would enable the assessment of incremental emissions reductions and associated effects over a number of years rather than all at once as currently simulated. Multi-year simulations are also needed to characterize the inter-annual variability of trans-Pacific transport.

Conclusions

Epidemiologic evidence has been published in recent years confirming earlier observations of significant harm from particulate matter (PM), especially, but not confined to, the fine fraction. Of note is the confirmation of mortality from long-term exposure to PM, and the linkage to adverse cardiac outcomes, both from acute and chronic exposures. Additionally, the finding of a robust relationship with lung cancer mortality in the most influential chronic exposure cohort study (the American Cancer Society study) has added to the understanding of specific adverse outcomes associated with fine PM. The emergence of animal toxicology studies using ambient particles has provided evidence of a broad range of mechanisms and toxicological pathways by which adverse outcomes could manifest themselves. These studies also implicate the importance of the source of PM (and hence the chemical composition of PM) in these outcomes. Overall, the database provides sufficient evidence to conclude that there are causal relationships between PM and a range of adverse effects including restricted activity days, respiratory symptoms, bronchitis (both acute and chronic), asthma exacerbation, as well a range of respiratory and especially cardiac impacts which result in increased emergency room visits, hospital admission and premature mortality.

Significant uncertainty remains as to the relative toxicity of sources and components of PM, the role of co-occurring pollutants for some endpoints, the importance of reproductive outcomes seen in an increasing number of studies, and the role of PM in eliciting effects beyond the cardio-respiratory systems. The lack of significant insights from studies with human volunteers in controlled exposure situations (i.e., clinical settings) limits understanding of some important mechanistic pathways.

The database on O_3 health effects confirms its role in mortality (especially acute exposure-related) and a variety of morbidity effects. The major effect is on respiratory outcomes, though there are indications of possible adverse effects beyond the lung. The most recent evidence on the health effects of ambient ozone is consistent with that reported in earlier evaluations, and provides additional details on outcomes, modifying factors, and the presence of significant susceptible groups in the population.

Evidence is very strong for an effect of O_3 on a range of adverse respiratory outcomes including lung function, respiratory symptoms, inflammation, and immunological defenses. Overall, this evidence lends considerable support to the epidemiological literature, which has reported significant associations with respiratory emergency room and hospital visits (especially asthma-related), and premature mortality associated with short-term exposure to ozone.

Observations from the PM and O_3 literature are sufficient to conclude that certain groups within the population can be characterized as particularly susceptible to adverse effects following exposure to these pollutants. Susceptible sub-populations include children (both healthy and asthmatic), the elderly (especially those with a pre-existing respiratory or cardiac condition), individuals who hyper-respond to respiratory irritants, and those who are more active outdoors (e.g., exercising individuals, outdoor workers). There is also an increasing body of evidence to support susceptibility related to the specific genetic makeup of the individual.

Recent evidence also confirms that exposure to PM and O_3 can have negative effects on plant health which can lead to ecosystem changes. Plants take up O_3 via leaf stomates which can cause direct physical damage, while PM effects are largely due to changes in soil chemistry rather than direct deposition. Vegetation response to these pollutants is dependent upon the composition of PM, plant species and development stages (in the case of O_3), and can be modified when combined with other ecosystem stressors. Impacts from the exposure to PM and O_3 on wildlife is an area of emerging concern in understanding ecosystem impacts; however, no research has been conducted to asses direct impacts or to identify which species are more sensitive.

Smog has wide-ranging negative impacts on our social and economic wellbeing. Although most of the focus has been on the quantification of health-related socio-economic impacts (e.g., medical treatment costs and lost worker productivity), non-health related socio-economic impacts (e.g., decline in crop and tree output from farm and forestry operations, reduced enjoyment of vistas from poor visibility, and costs associated with material breakdown) are also significant.

Recent monitoring observations and analyses show that southern Ontario and southern Quebec continue to record the highest fine $PM_{2.5}$ and O_3 levels in Canada, according to the Canada-wide Standards metrics. Almost all sites over the Canada-wide Standards 2010 targets are located in southwestern Ontario and in large urban centres and communities influenced by local industries in eastern Ontario and southern Quebec, although $PM_{2.5}$ levels are closer to the target than is the case for O_3. Southern Ontario and southern Quebec also have the highest number of days and frequency of episodes with levels greater than the Canada-wide Standards numeric targets for $PM_{2.5}$ and O_3 (30 µg m^{-3} and 65 ppb, respectively). These conclusions are based on studies and data covering the period 2001–2006.

As discussed throughout this summary document, the spatial patterns of $PM_{2.5}$ and O_3 across the country vary regionally and seasonally. Local ambient levels of $PM_{2.5}$ and O_3 are influenced by a variety of factors that can lead to substantial differences in ambient concentrations from one region of the country to another and over time. These include meteorology, coastal and urban effects, the local chemical mixture, natural emissions, transboundary transport and background concentrations. High $PM_{2.5}$ levels occur both in the summer and winter, with the cold season receiving more attention recently due to the significant

role of local emission sources leading to the build up of primary pollutants (i.e., primary $PM_{2.5}$). High O_3 levels occur in the spring or summer due to enhanced photochemical production, precursor emissions and favourable meteorological conditions.

Air quality observations show that between 1985 and 2006, ambient $PM_{2.5}$ levels at urban sites across Canada declined by approximately 40%; though most of the decline occurred before 1996 as a result of SO_2 emission controls in eastern U.S. and Canada. No long-term data are available to examine regional-scale changes in ambient $PM_{2.5}$; however, since 2001, emission reductions of sulphur dioxide (SO_2) and nitrogen oxides (NO_X) in eastern Canada and the northeastern U.S. have resulted in declines in $PM_{2.5}$ constituents (namely ammonium sulphate and ammonium nitrate) and ambient SO_2 and nitric acid (HNO_3) precursor gases. In light of decreasing SO_2 and NO_x emissions in eastern North America, the influence of ammonia (NH_3) on $PM_{2.5}$ levels is expected to increase.

Changes in O_3 levels have been observed at the regional and local scale. At urban sites across Canada, median and lower percentile ambient 8-hr O_3 levels increased between 1990–2006; attributed to less O_3 scavenging (the removal of O_3 from the atmosphere as it reacts with NO to form NO_2) as a result of NO_X emission reductions. Outside of these unique urban circumstances, rural sites in Ontario and Quebec experienced a decreasing trend from 1990–2006 across all percentiles in the distribution of O_3 concentrations. Regionally, ambient O_3 concentrations have been declining in Quebec and Ontario, remaining constant in Alberta, and increasing in Atlantic Canada and on the Pacific coast.

Monitoring and modelling studies show that transboundary transport of pollutants occurs all across Canada but has a greater impact on air quality over southern Ontario, Quebec and Atlantic Canada. In spite of the strong influence of emissions from the U.S., local emissions and local formation of O_3 and $PM_{2.5}$ in some areas of eastern Canada play a significant role in degrading air quality to levels close to or above the Canada-wide Standards.

Baseline O_3, the level measured at a given site in the absence of strong local influence is increasing in a number of areas in Canada (i.e., Georgia Basin in the Pacific coast, Atlantic coast, and continental western Canada). Changes in baseline O_3 are consistent with evidence of increasing intercontinental transport of O_3 into North America.

Air quality models project that with the implementation of existing North American legislation, $PM_{2.5}$ and O_3 levels are expected to decrease by 2015 (relative to 2002) across the country and regionally. Exceptions include large urban areas where emissions of primary $PM_{2.5}$ are projected to increase and changes in the atmospheric chemical regime may lead to increases in $PM_{2.5}$ formation, and parts of the Prairies where industrial emissions are increasing. The number of days exceeding the Canada-wide Standard numeric value for $PM_{2.5}$ is projected to increase in areas already showing exceedances, while the number of days exceeding the Canada-wide Standard numeric value for O_3 are projected to decrease overall. Model simulations of additional reductions in industrial emissions, if new Canadian legislation is implemented, show widespread improvements in summertime $PM_{2.5}$ and O_3 levels in the Prairies but only marginal improvements in other parts of the country.

National emissions of primary $PM_{2.5}$, and $PM_{2.5}$ and O_3 precursors have generally declined from 1985–2006, with the exception of NH_3. Some of this improvement has been offset in recent years by

increases in some source sectors. Emissions projections for the year 2015 show expected increases relative to 2006, namely for VOCs and NH_3.

The state of knowledge of smog science in Canada has greatly improved over the last 10 years and is continually evolving. Emerging areas of research include the impact of climate change on smog and the associated ecosystem and human health effects, the role of intercontinental transport (e.g., across the Pacific from Asia) on Canadian air quality, and the use of satellite-based monitoring. Modelling studies indicate that climate change may result in increases in both $PM_{2.5}$ and O_3; however, further investigation is needed to understand the complex interactions between air quality and climate change and to quantify the overall impact and regional variation. The role of intercontinental transport of smog-forming pollutants on Canadian air quality is an issue needing attention, especially with changing domestic precursor emissions. The use of satellite-based measurements to study air pollution is expected to grow, particularly in areas of the country where ground-level monitoring is limited.

The information contained within the *Canadian Smog Science Assessment* and synthesized in this document reflects the considerable amount of research, monitoring and modelling information available across Canada. It is also clear that large gaps still remain in our understanding of smog and its impacts on Canadians and their environment. Further advancements are needed to address these gaps and to improve our ability to track changes resulting from air quality management strategies (see the sections “Knowledge Gaps” and “Recommendations for Future Research”).

www.ec.gc.ca

Additional information can be obtained at:

Environment Canada
Inquiry Centre
351 St. Joseph Boulevard
Place Vincent Massey, 8th Floor
Gatineau, Quebec K1A 0H3
Telephone: 1-800-668-6767 (in Canada only)
or 819-997-2800
Fax: 819-994-1412
TTY: 819-994-0736
Email: enviroinfo@ec.gc.ca

www.hc-sc.gc.ca

Additional information can be obtained at:

Health Canada
Address Locator 0900C2
Ottawa, Ontario K1A 0K9
Telephone: 1-866-225-0709 (in Canada only)
or 613-957-2991
Fax: 613-941-5366
TTY: 1-800-267-1245
Email: Info@hc-sc.gc.ca