PROGRAM

NERAM Colloquium V

Strategic Policy Directions for Air Quality Risk Management

OCTOBER 16, 2006

A.M. Chair: **Glen Okrainetz**, *BC Ministry of Environment*

8:00 Registration & Coffee ICBC Concourse

8:30 Welcome – **John Shortreed**, *NERAM*

8:40 Opening Remarks – **Glen Okrainetz**, *BC Ministry of Environment*; TBA, *Environment Canada*

SESSION 1: Why is Air Pollution a Global Public Health Concern?

9:00 Keynote Address: Science and Policy for Global Air Quality Management
– **Michal Krzyzanowski**, *WHO Europe*

9:25 The Global Burden of Disease Due to Urban Air Pollution: Estimates and Uncertainties
– **Aaron Cohen**, *Health Effects Institute*

9:50 Discussants: The Value of GBD Estimates in Air Quality Management – **Daniel Krewski**, *McLaughlin Centre for Population Health Risk Assessment;* **Alan Krupnick**, *Resources for the Future*

10:20 *Coffee* ICBC Concourse

10:55 Impact of pollution on Public Health in Hong Kong and the Pearl River Delta
– **Anthony Hedley**, *University. of Hong Kong*

11:20 Impact of Air Pollution on Public Health: Transportability of Risk Estimates
– **Jonathan Samet**, *Johns Hopkins University*

11:45 Plenary Discussion

12: 00 LUNCH ICBC Concourse

P.M Chair: **Martin Williams**, *UK Department of Environment*

SESSION II: Inputs to Policy: Air Quality Measurement, Modeling and Monitoring

1:30 Ambient Air Quality Measurement for Policy Decisions – **Jeff Brook**, *Environment Canada*

1:55 Assessment of Emission Inventories in North America – **William Pennell**, *NARSTO*

2: 20 Plenary Discussion

2:40 *Coffee* ICBC Concourse

3:10 Source-receptor Relationships: Implications for Air Quality Policy
– **Philip Hopke**, *Environmental Quality Systems Center, Clarkson University*

3:35 Integrated Assessment Modeling
– **Hadi Dowlatabadi**, *Institute for Resources, Environment and Sustainability, University of British Columbia*

4:00 Matching the Metric to Need: Modelling Exposures to Traffic-Related Air Pollution for Policy Support
– **David Briggs**, *Imperial College London*

4:25 Plenary Discussion

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A.M. Chair: **Michael Brauer**, *University of British Columbia*

SESSION III: Evidence-Base for Effective Air Quality Management

8:00 Registration & Coffee ICBC Concourse

8:30 Mortality Risk Valuation for Air Quality Policy – **Alan Krupnick**, *Resources for the Future*

8:55 Policy Case Studies from North America – **Bart Croes**, *California Air Resources Board*

9:20 Air Quality Management Capability of Selected Asian Cities - 2006 Update – **Kong Ha**, *CAI-Asia*

9:45 Plenary Discussion

10:00 *Coffee* ICBC Concourse

10:20 Policy Case Studies from Europe – **Matti Vainio** *European Commission, Environment Directorate-General,* – **Martin Williams**, *UK Department of Environment*

SESSION IV: Global Strategies for Air Quality Management

10:45 Canada/US Transboundary Airshed Strategies – **Brian McLean**, *USEPA;* **Jane Barton**, *Environment Canada*

11:25 Joint Air Quality and Climate Change Strategies: Challenges and Opportunities – **Quentin Chiotti**, *Pollution Probe*

11:50 Plenary Discussion

12:15 LUNCH ICBC Concourse

P.M. Chair: **Jonathan Samet**, *Johns Hopkins University*

SESSION V: Directions for Global Air Quality Management

1:15 Break Out Session I: Identification of Initial Strategies – **Lorraine Craig**, *NERAM* Asia Pacific Hall & Room 320

2:15 *Coffee* ICBC Concourse

2:35 Rapporteur Reports

3:05 Emerging Issues and Opportunities for Air Quality Management – **Jonathan Samet**, *Johns Hopkins University*

3:15 Evaluating the Effectiveness of Interventions: Accountability – **Annemoon van Erp**, *Health Effects Institute*

3:35 Environmental Justice in Air Quality Policy Development – **Marie O'Neill**, *University of Michigan*

3:55 How do Montrealers with Chronic Obstructive Pulmonary Disease (COPD) and/or Congestive Heart failure (CHF) Behave on Smoggy Days? – **Tom Kosatsky**, *Montreal Department of Public Health*

4:15 Panel Perspectives: **Tony Clarke-Sturman**, *Shell International;* **Hugh Kellas** *Greater Vancouver Regional District;* **Quentin Chiotti**, *Pollution Probe;* **Kong Ha**, *CAI-Asia*

5:05 Plenary Discussion/Wrap up

7:00 **Conference Dinner –** Innes Thompson Room, Delta Suites

"Reflections on Air Quality and Health" **Dr. Ray Copes,** *Director, Environmental Health, BC Centre for Disease Control & Scientific Director, National Collaborating Center - Environmental Health*

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SESSION V (Directions for Global Air Quality Management Continued)

Chair: **Tony Clarke-Sturman**, *Shell International*

8:00 Coffee ICBC Concourse

8:30 KEYNOTE: Future Directions for Global Air Quality Management – **Martin Williams**, *UK Department of Environment*

8:50 Break Out Session II: Strategic Policy Directions for Air Quality Management – **Lorraine Craig**, *NERAM* Asia Pacific Hall & Room 320

10:15 *Coffee* ICBC Concourse

10:45 Rapporteur Reports

11:15 Statement on Strategic Policy Directions for Air Quality Management – **Daniel Krewski**, *McLaughlin Centre*

12:30 Closing Remarks

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ABSTRACTS

October 16, 2006 SESSION 1

Science and Policy for Global Air Quality Management - Michal Krzyzanowski, *WHO Europe*

The knowledge on health effects of air pollution has increased substantially in the recent decade. Results of hundreds of new studies are published yearly in scientific literature, showing consistently that the air pollution affects health in all parts of the world. The estimates of burden of disease due to air pollution show that both the outdoor and indoor air pollution remains an important public health burden, in particular in medium and low income countries. Evaluation of the scientific evidence conducted in the framework of the update of WHO Air Quality Guidelines, recommends globally relevant goals for air quality management, necessary to be achieved in order to reduce adverse health effects of the pollution.

Many countries of the world notice this new evidence and set ambitious policy objectives. The member states of the European Region of WHO agreed that the reduction of respiratory disease due to outdoor and indoor air pollution is one of (4) regional priority goals for Children Environment and Health Action Plan for Europe. The EU 6th Environmental Action Plan calls for "levels of air quality that do not give rise to significant negative impacts on, and risks to, human health". The EU Thematic Strategy on air pollution, announced in Sept. 2005, proposes ways to achieve these objectives.

However, the implementation of the actions to reduce air pollution are often challenged by growing transport and energy demands, and claims that the measures aimed at pollution reduction, or prevention, are costly or restrict development. Consideration of health issues is still not always integrated in environmental impact assessment or in cost-benefit analysis of various actions. In many cases, lacking expertise or financial resources of pollution control agencies affect implementation of actions. Many of these limitations may be reduced by a more effective international collaboration, based on a better understanding of the global nature of the risks and of transboundary nature of air pollution. National actions will be strengthened by effective transfer of information, exchange of expertise and technical assistance to the countries developing their capacities for air quality management.

The Global Burden of Disease Due to Urban Air Pollution: Estimates and Uncertainties - Aaron Cohen, *Health Effects Institute*

Estimates of the global burden of disease due to urban outdoor air pollution in 2000 were made as part of the World Health Organization's Comparative Quantification of Health Risks (CRA) project, and the results were presented in the 2002 World Health Report. The CRA project created a framework for estimating the burden of disease due to (24) major risk factors using a consistent methodology. The relative magnitude of the burdens estimated for different risk factors could then be compared in a context that minimized the influence of advocacy on behalf of specific risk factors.

In order to provide estimates of the burden of disease due to urban outdoor air pollution for all 14 world regions defined by WHO, models developed by the World Bank were used to estimate ambient concentrations of inhalable particles (PM_{10}) in 3211 national capitals and cities with populations of >100 000. These estimates were converted to estimates of fine particles ($PM_{2.5}$) using information on geographic variation in the ratio of $PM_{2.5}$ to PM_{10}. Population-weighted sub-regional annual average concentrations of $PM_{2.5}$ were estimated using the population of the cities in the year 2000. Burden of disease estimates were based on three health outcomes: mortality from cardiopulmonary disease in adults, mortality from lung cancer, and mortality from acute respiratory infections in children aged 0–4 years. Attributable deaths and years of life lost (YLL) for adults and children (aged <5 years) were estimated using risk coefficients from a large cohort study of adults in the United States and a meta-analytic summary of five time-series studies of mortality in children, respectively.

Outdoor air pollution in urban areas worldwide, in terms of concentrations of $PM_{2.5}$, was estimated to cause about 3% of mortality attributable to cardiopulmonary disease in adults, about 5% of mortality attributable to cancers of the trachea, bronchus and lung, and about 1% of mortality attributable to acute respiratory infections in children. This amounts to about 0.80 million deaths (1.2% of the global total premature deaths and 6.4 million YLL (0.5% of the global total). This burden is borne predominantly by the populations of developing countries, with 65% of the attributable burden occurring in the developing countries of Asia.

The estimates were subject to considerable uncertainty. Uncertainty about the shape of the concentration–response function and other factors was assessed in sensitivity analyses. These analyses indicated that the estimates were most sensitive to the choice of concentration–response function and theoretical minimum (counterfactual) level of exposure. The major effect of these uncertainties was to underestimate the burden of disease. Other major, and potentially greater, sources of uncertainty, however, have only become clearer as the scientific evidence has developed in the last 5 years.

These include the spatial resolution of current exposure estimates, and omission of health endpoints, such as low birth weight and diabetes. Some of these uncertainties can be addressed with the evidence currently at hand; others will require additional research. (The views expressed in this paper are those of the authors and do not necessarily reflect the views of the Health Effects Institute *(HEI) or its sponsors.)*

Impact of Pollution on Public Health in Hong Kong and the Pearl River Delta: Linking Pollution Awareness with Public Health Impacts - AJ Hedley[1], SM McGhee[1], B Barron[2], P Chau[1], J Chau[1], TQ Thach[1], TW Wong[3], C Loh[4], CM Wong[1]

[1] Community Medicine, School of Public Health, University of Hong Kong; [2] Institute for the Environment, University of Science & Technology; [3] Community and Family Medicine, Chinese University of Hong Kong; [4]Civic Exchange; Hong Kong SAR, China

Hong Kong, a territory of 1000 km^2 has a population of 6.8 million with about 30,000 deaths, 1.1 million hospital admissions and over 43 million primary care consultations for respiratory complaints a year. Air quality in Hong Kong and the Pearl River Delta is poor and compares unfavourably with other socio-economically developed cities such as Auckland, Berlin, London, New York, Paris and Vancouver. Particulate levels in Hong Kong are about 40% higher than in Los Angeles. Governmental policy on both the mainland and Hong Kong SAR sides of the boundary have been ineffective in controlling air quality. Our aim was to present analyses of poor and better visibility days, morbidity and mortality counts associated with these levels of pollution and the costs to the community for health care, lost productivity and intangible costs.

Public awareness of pollution in Hong Kong is linked to progressive reductions in visibility over 30 years. We obtained landscape photographs and data on associated pollutant levels on typical *poor* and *better* visibility days and found a marked difference in the 24 hour pollutant profiles. We calculated the annual proportions of days at those different pollutant levels and the annualized primary and secondary health care events and deaths and value of the costs which would be avoided if, instead of the higher pollution levels, we achieved the lower level throughout a year. The principal illustrations were based on improvement from average to *"good"* levels defined as the lowest levels observed on *better* visibility days. We used previously published risks for doctor consultations, cardiopulmonary disease and mortality associated with four criteria pollutants. The total number of avoidable health events associated with pollution was estimated using NO_2 + 75% SO_2 + 46% RSP + 100% O_3 to take account of correlation between pollutants at monitoring stations. Other sensitivity analyses were made based on the single pollutant with the biggest effect associated with morbidity (NO_2), mortality (SO_2) and with RSP + O_3.

We estimated costs in three main categories: direct costs of illness; productivity losses, and intangible costs using data from health care providers, other statistical sources and public surveys of willingness-to-pay to avoid illness and premature death. Based on visibility improvement from average to *"good"* we estimate that 1600 deaths, 64000 admissions and 6.8 million doctor consultations would be avoided annually. Avoidable costs total US$2.7B, including direct costs ($192M), lost productivity ($65M) and intangible costs ($2.4B).

The analysis is conservative and demonstrates the potential health gains and reduction in community costs which could be achieved by consistent steps in the use of cleaner fuels and power units, improved power generation efficiency, infrastructure and transportation.

Media and public responses to the demonstration of a link between health gains and improved visibility were strongly positive but that of the Government largely negative. The process of advocacy based on public health accountability is continuing but the outcome is uncertain.

Impact of Air Pollution on Public Health: Transportability of Risk Estimates - Jonathan M. Samet, *Johns Hopkins University*

The health effects of air pollution have been characterized in epidemiological and toxicological studies carried out throughout the world, although to date most of the research has been carried out in the North America and Europe. The sources of air pollution vary across the globe with the mixture being driven by patterns of industrialization and fuel use. Consequently, although not well documented, there is likely to be substantial variation in the air pollution mixture across the world's major cities; the differences are potentially relevant to public health and regulation.

This presentation addresses the possibility of heterogeneity in the effects of air pollution across the globe. Such heterogeneity would signal a need for local research on air pollution and locale-specific control strategies. It would be a source of uncertainty in extending risk estimates based in research in one location or country to others.

How would heterogeneity arise in estimates of the health effects of air pollution? The possibilities include: 1) true heterogeneity in the toxicity of the mixture; 2) differences in population susceptibility; and 3) methodologic differences

in research approaches or the play of chance. To address the third explanation, standardized protocols are needed, such as those developed and applied in multiple locations by the groups conducting the National Morbidity, Mortality and Air Pollution Study (NMMAPS) and Air Pollution and Health: A European Approach (APHEA) project.

The need for such multi-country protocols as the basis for characterizing and limiting heterogeneity is now well recognized. This presentation offers case studies on variation in risk estimates for PM and ozone and the basis for this variation. It explores potential implications for risk assessment and for the conduct of research.

SESSION II

Ambient Air Quality Measurement for Policy Decisions - Jeff Brook, *Environment Canada*
Not available

Assessment of Emission Inventories in North America - Bill Pennell, *NARSTO*

Emission inventories are the foundation of effective air quality management. Without accurate information on the sources of air pollutants – what they are, where they are located, what they are emitting, and how much – it is impossible to identify which sources are most important to control, to predict the air-quality consequences of these emissions, or to monitor the effectiveness of emission reduction programs. This presentation will discuss the basics of contemporary emissions inventory development, evaluation, and dissemination. It will address a) the development of emission inventories, b) methods for evaluating them and quantifying their uncertainties, c) typical strengths and weaknesses of mature emission inventories, d) actions for addressing the weaknesses, and e) the costs of emission inventory development and improvement.

The Use of Source Apportionment for Air Quality Management and Health Assessments
- Philip K. Hopke, *Center for Air Resources Engineering and Science, Clarkson University*,

The U.S. Environmental Protection Agency has deployed a network of urban airborne particulate matter samplers to provide $PM_{2.5}$ composition data for urban centers across the United States. These data show the differences in composition in different part of the country. These data can also be used to identify and apportion the particle sources. These results can be used to develop effective and efficient air quality management plans. They can help to refine emission inventories for input into deterministic models to predict changes in air quality as the result of the implementation of various management plans. The apportionments can also serve as exposure estimates for health effects models to identify those components of the particulate matter that are most closely related to observed adverse health effects. Although current regulations target total airborne mass concentrations, such health effects results could result in targeting those sources that are most likely linked to health effects and thus, would produce the maximum health benefit.

Systems Thinking But Not Practice: The Other Human Dimensions of Integrated Assessment
- Hadi Dowlatabadi, *University of British Columbia*

Systems thinking has come a long way from the Club of Rome. The human dimensions of issues can now be more explicitly characterised. This presentation will offer examples from such progress. However, there are many areas where such analysis offers insights that are yet to be implemented with examples from climate policy, air quality management and infectious diseases.

There is more than enough evidence that analysis alone will not win the day. So, what can be done to arrive at better solutions more expeditiously?

Matching the Metric to Need: Modelling Exposures to Traffic-Related Air Pollution for Policy Support
David Briggs, Kees de Hoogh and John Gulliver, *Imperial College London, UK*

With the development of GIS, a wide range of different methods have become available for exposure assessment in the field of air pollution. Taking advantage of these, research studies have tried to move away from the traditional approach of using data from the nearest monitoring site (for acute studies) or measuring distance from the nearest road (for chronic studies) to use other, more sophisticated exposure metrics. Many of these were recently reviewed by Jerrett (2005). These methods have greatly increased the capability for analysis, and encouraged innovation in study design, but they are also creating a dilemma of diversity: how can the results from these different studies, using different exposure metrics, be compared? what can we learn from the different methods and studies? And, above all, what are the implications for intervention or policy?

This paper is intended to start to address these questions. Based on a detailed analysis in London, UK, it first compares the results from a series of different methods and metrics of chronic exposure, including proximity to road, road density, traffic density, and modelled exposures to NO_2 (as a marker for traffic-related air pollution) using land use regression and dispersion modelling techniques. It also compares the various results against measured concentrations of particulates (the main agent of concern) at monitoring stations across the city. It then discusses the implications of the different pictures of exposure that each of these different methods and metrics provide (and their differing associations with monitored concentrations) for our interpretations of health risks from traffic-related air pollution. Finally, it considers the questions of how better to match the exposure metric used to research or policy need, and what needs to be done to improve the basis for air pollution exposure assessment in the future.

October 17 SESSION III

Mortality Risk Valuation for Air Quality Policy - Alan Krupnick, *Resources for the Future*

This paper will examine the conceptual foundation and use of mortality risk valuation estimates -- the Value of Statistical Life -- in air quality policy, with an emphasis on the U.S. experience. The conceptual foundation is in neoclassical welfare economics and one of its practical applications is in cost-benefit analysis conducted by governments to help design efficient air quality policy. Particular attention in the application of the VSL will be paid to the issue of the senior death discount, which is the theoretical expectation and the relatively unrobust empirical findings that the WTP of seniors for reducing their mortality risk is lower than that of younger adults. This issue also plays into the debate over the use of the Value of Statistical Life Year to replace the VSL and the use of quality adjusted life years as an effectiveness metric in cost-effectiveness analyses of government regulations.

Policy Case Studies from North America - Bart Croes, *California Air Resources Board*

In response to severe air quality problems, Canada, Mexico and the United States imposed comprehensive emission controls, but growth in population, vehicle miles traved and industrial activity prevented attainment of health-based ambient air quality standards. Emissions of VOC and CO (and to a lesser extent NO_X) from new passenger vehicles were reduced by a factor of a hundred in comparison to pre-control vehicles. The United States adopted emission standards for 2007 and subsequent model year heavy-duty engines that represent 90% reductions of NO_X and PM compared to 2004 model year emission standards. Implementation of reformulated gasoline and diesel fuels resulted in further reductions. Stationary source NO_X and SO_X emission standards were reduced by at least a factor of ten since 1980. Small off-road engines, architectural coatings, consumer products and solvents are also targeted for large emission reductions. While some air quality problems have been eliminated or greatly reduced (i.e., lead, NO_2, SO_2), particulate matter and ozone levels remain high in many North American cities, resulting in thousands of deaths per year and increased disease rates. In response, air quality management agencies are developing innovative approaches, including regulation of in-use emissions, reactivity-based VOC controls and exposure-based prioritization of PM controls. Several cooperative, multi-national efforts (e.g., NARSTO) have advanced the science and technology of air pollution control, and begun to address transboundary issues. Newly recognized challenges also need to be integrated into air quality management programs, ranging from the microscale (e.g., air pollution “hotspots”, ultrafine particles, indoor air quality) to global scales (e.g., climate change mitigation, international goods movement).

Urban Air Quality Management in Asian Cities (2006) - Kong Ha, *CAI-Asia*

In cooperation with the Air Pollution in the Megacities of Asia project of the Stockholm Environment Institute in York, and environmental agencies of participating cities, CAI-Asia has completed the benchmarking study on urban air quality management of selected Asian cities, which aims to raise the level of knowledge and awareness on the various policies and programs that Asian cities employ in order to mitigate air pollution. The study provides the most current and comprehensive assessment and comparison of the status and drivers of urban air pollution in 20 Asian cities and the Asian region, covering the effects on the environment, human health, agriculture and cultural heritage and the future implications for planning, transport and energy industries. National and local governments have begun to develop air quality management (AQM) strategies to address the deterioration in urban air quality, however the scope and effectiveness of such strategies varies widely. The study has qualitatively classified the capability of the cities to manage urban air quality into minimal, limited (I and II), moderate (I and II), good (I and II) and excellent (I and II) and identified challenges in Asian cities to effectively manage their air quality. Ambient air quality concentrations in these cities have shown a declining trend over the past decade. Annual ambient concentrations of sulfur dioxide were found to meet the prescribed guidelines of the WHO (WHO 2000) for the cities included. However, particulate matter remains to

exceed the WHO 2000 guidelines in these cities, while data on nitrogen oxides, ozone, and carbon monoxide widely varies across the different cities.

Policy Case Studies from Europe - Matti Vainio, *European Commission, Environment Directorate-General*
Martin Williams, *UK Department of Environment*
Not available

SESSION IV

Canada-U.S. Transboundary Airshed Strategies - Brian McLean, *USEPA*, Jane Barton, *Environment Canada*

The impetus for the Canada-U.S. Air Quality Agreement was transboundary acid rain in eastern North America. This problem drove us to develop a bilateral agreement that not only addressed this issue, but also set up a broad and flexible framework to address other air quality problems. In 2000, we negotiated the Ozone Annex to reduce smog and its precursor pollutants. A transboundary PM science assessment in 2004 is supporting discussions concerning a possible particulate matter annex, which continue.

Over the course of fifteen years, Canada and the United States have also developed innovative cooperative arrangements. Among these arrangements are two transboundary airshed dialogues that have become important sources of practical on-the-ground cooperation in the Georgia Basin-Puget Sound and the Great Lakes Basin. As well as providing the basis for ongoing international dialogue, these transboundary airshed projects are resulting in changes to administrative practices as the parties exchange information and learn from each other in ways that benefit the airshed community. The nature of the Air Quality Agreement has also enabled both Canada and the U.S. to address concerns each has had about specific pollutant sources and to address them in ways that have avoided confrontation and resulted in air quality improvements for people living in the airsheds. Case studies of three of the "informal consultations" that have occurred under the Agreement will be described – where discussions occurred around a power plant in Michigan, a power plant in Saskatchewan, and a steel mill in Ontario.

More than an agreement, this relationship has established trust, building a capacity to deal with problems rather than dealing with national origins. Fostering such a relationship with its implicit transfer of knowledge and experience has opened doors for discussions on a new Clean Air framework in Canada and joint analyses of cross-border SO_2 and NOx emissions caps and trading. U.S. experience with cap and trading will be highlighted for background and context. The flexibility inherent in the agreement provides a platform for future air quality issues and continued communication without borders.

Joint Air Quality and Climate Change Strategies: Challenges and Opportunities for Policy Makers
Quentin Chiotti[1], K. Oglivie[1], J. Drexhage[2], M. Pattenden[1]
[1] Pollution Probe, [2] International Institute for Sustainable Development

This presentation addresses the broad challenges and opportunities for developing and implementing joint air quality and climate change strategies. These atmospheric issues have largely been addressed as separate policy issues, but in recent years there has been growing awareness that they are inextricably linked and that integrated policies and actions need to be developed. It is not clear if it is better to improve air quality through climate change measures, or climate change through air quality measures, especially since they have different temporal and spatial dimensions in terms of atmospheric chemistry, health effects and political responsibility. As part of the presentation, the following framework is applied to air quality and climate change strategies for energy (electricity generation) and transportation, illustrating the complexity of the problem and the challenges facing decision-makers to develop and implement effective policies:

1. To what degree is there scientific certainty about the problem and the health effects?
2. How well are the main [emission] sources known?
3. Is there agreement on the solutions and their expected outcomes?
4. Which solutions are economically feasible, and are they politically acceptable?
5. What are the direct and indirect impacts of various solutions, and how are these being contested in the political arena?

While examples are drawn largely from the Canadian experience, the issues, challenges and opportunities also apply to other developed and developing countries.

SESSION V

Evaluating the Effectiveness of Interventions: Accountability - Annemoon van Erp, Robert O'Keefe, Aaron Cohen, and Jane Warren, *Health Effects Institute*

Evaluating the extent to which air quality regulations improve public health is part of an emerging effort—sometimes referred to as *accountability*—to assess the performance of environmental regulatory policies. Air quality has improved substantially in the United States and Western Europe in recent decades, with far less visible pollution and decreasing concentrations of several major pollutants. In large part, these gains have been achieved through increasingly stringent air quality regulations. The costs associated with compliance and, importantly, the need to ensure that the regulations are achieving the intended public health benefits, underscore the importance of accountability research. To date, accountability research has largely emphasized measuring the effects of actions already taken to improve air quality, but such research may also contribute to estimating the burden of disease that might be avoided in the future if certain actions are taken.

Several agencies have started to collect and assess evidence about the extent to which air pollution control measures have improved health, including the National Research Council, California Air Resources Board, Environmental Protection Agency, and others. In 2003, the Health Effects Institute (HEI) formally committed to a comprehensive program of new research in this area. As an initial step it issued a monograph on accountability: *Assessing Health Impact of Air Quality Regulations: Concepts and Methods for Accountability Research* (HEI Communication 11), which sets out a conceptual framework for accountability research and identifies types of evidence required and methods by which the evidence can be obtained. At the heart of this framework is the concept of a chain of accountability that links regulatory action to health responses via sequential changes that affect emissions, air quality, exposure and, ultimately, human health.

HEI has underway a substantial research program on accountability with 7 ongoing studies. These research projects involve measuring indicators along the entire chain from regulatory action to health outcomes. Projects cover near-term interventions that are typically implemented over relatively short periods of time, such as a ban on the sale of coal, replacing old wood stoves with cleaner ones, measures to reduce traffic, and more long-term, wide-ranging actions or events such as decreasing sulfur content in gasoline, and complex changes associated with the reunification of Germany, such as switching from brown coal to natural gas (power plants and home heating) and increased numbers of diesel-powered vehicles in eastern Germany. HEI is also supporting the development of methods and research to assess an especially challenging area, regulations that are implemented incrementally over extended periods of time, such as Title IV of the Clean Air Act, which reduces SO_2 emissions from power plants. The presentation will contain an overview of the accountability studies currently funded by HEI.

Environmental Justice in Air Quality Policy Development - Marie O'Neill, *University of Michigan*
Not available

How do Montrealers With Chronic Obstructive Pulmonary Disease (COPD) and/or Congestive Heart Failure (CHF) Behave on Smoggy Days - Tom Kosatsky, A. Renouf, J. Dufresne, L. Richard, N. Giannetti, J. Bourbeau, *Montreal Department of Public Health*

Ozone and fine particulate matter are important threats to the life and health of persons with COPD and CHF. Weather services and public health authorities advise patients with heart and lung conditions to reduce outdoor physical activity and to maintain close contact with at least one other person during smoggy days. The degree to which this target group responds to these recommendations has received little attention.

During the summer of 2005, we recruited 152 patients from specialized pulmonary clinics and 86 from cardiac clinics, all Montreal residents, into a two-phased survey of their knowledge, attitudes and practices concerning poor air quality and intense heat. A subset of 61 participants completed a telephone interview the day following a smog warning, or the day following warnings of both smog and excessive heat, as called by Environment Canada. The 61 were asked if, in their opinion, the preceding day was smoggy, if they had heard or seen an air quality warning and where, if symptoms had appeared or got worse, and which protective behaviors they practiced during the previous 24 hours. Results showed that 43/61 (71%) found the air polluted: asked what motivated their response, 13% mentioned high heat and humidity, and 7% noted pollen. Of 43 who described the air as polluted, 26 (60%) said their opinion was based on sight or smell. 75% of the 61 recalled hearing a smog warning, the majority via television. The principal symptoms reported related to the respiratory system (57%). As for protective measures taken, 77% said they had reduced their level of physical activity during the warning day, and 72% had checked in with a friend or relative. Oddly, some respondents also

reported turning on their air conditioners, and/or drinking extra fluids.

This vulnerable group does hear warnings, does report symptoms, and does take protective actions during smoggy days. However, their capacity to reduce an already low level of physical activity, and their ability to distinguish intense heat from smog, are doubtful.

Responses among 46 persons interviewed following excessive heat (but not smog) warnings will also be presented and compared to those for the 61 interviewed the day after air quality warnings. *(This work was supported by the Climate Change Action Funds)*

Future Directions for Global Air Quality Management - Martin Williams, *DEFRA, UK*

The paper will address the future pressures driving air quality management, including evidence on health and environmental effects and their significance for methods of air quality controls. Novel approaches to air quality management currently emerging in Europe will be considered. The importance of a coherent consideration of approaches to the management of greenhouse gases and air pollutants will be discussed. The related issue of the widening spatial scope of air quality problems and intercontinental transport of air pollutants will be discussed along with the future implications of these issues for air quality management on a global scale.

BIOGRAPHIES

Jane Catherine Barton, *Environment Canada*

Jane Barton has a Masters degree from London School of Economics and over 25 years experience in environmental management in the private and public sectors. Since the mid 1980's, Ms. Barton has been a Canadian federal member of negotiating teams that have put in place four international legal agreements to address transboundary issues of acid rain and smog, the most recent being the Ozone Annex to the 1991 Canada-U.S. Air Quality Agreement signed by Canada and the U.S. in 2000. She currently heads a unit that deals with North American Smog and Acid Rain programs in Environment Canada and is the Canadian Co-Chair of the Canada-U.S. Air Quality Committee Subcommittee on Program Monitoring and Reporting under the Canada-U.S. Air Quality Agreement.

David Briggs, *Imperial College London*

David Briggs is Professor of Environment and Health Science in the Dept. of Epidemiology and Public Health at Imperial College London. A geographer by background, his research focuses on the use of GIS methods for exposure modelling and spatial analysis of environmental health effects. As leader of the GIS Group in the Small Area Health Statistics Unit, he is involved in a wide range of national studies including investigations of landfill sites, multiple deprivation, and urban air pollution. He has been a principal investigator on many EU-funded projects, including several studies focusing on traffic-related pollution and health (SAVIAH 1 and 2, HEARTS), a study aimed at mapping air pollution and atmospheric emissions across the EU (APMoSPHERE), a study on data limitations for EU environmental policy support (BICEPS) and two studies on the use of satellite data for environmental monitoring and policy support (MANTLE, GEMS). He currently leads a 33-institution project on Integrated Assessment of Health Risks from Environmental Stressors (INTARESE), funded under the EU 6th Framework Programme. Other current projects include studies on mobile phone base stations, powerlines, traffic-related benzene, intake fractions and air pollution modelling. He has published over 100 research papers, and about 40 research reports and books, and has presented invited papers to more than 150 conferences and symposia, world-wide.

Jeffrey R. Brook, *Environment Canada*

Dr. Brook, BS, Dip Met, MS, PhD, is a senior research scientist at Environment Canada in Toronto, Ontario, and adjunct professors in the Depts. of Public Health Sciences and Chemical Engineering at the University of Toronto. Dr. Brook is an Associate Editor for the Journal of the Air & Waste Management Association. He serves on the editorial board of a number of other scientific journals and the research management committee for AllerGen, which is a Canadian National Centre of Excellence focusing on genes and the environment. Dr. Brook began his career as an operational meteorologist before undertaking graduate work at the University of Michigan. He conducts original research in acid deposition and urban/regional air quality, emphasizing fine particulate matter, ambient measurement, and exposure assessment in support of a wide range of health-effect study designs. This latter research involves the interface between air pollutant characterization, with source-receptor analysis, and toxicological, clinical and both retrospective and prospective epidemiccological studies. Dr. Brook is currently leading Environment Canada's effort in advanced air quality and exposure research related to the Border Air Quality Strategy. This new program has involved the development of one of the world's most advanced mobile air quality laboratories known as CRUISER (Canadian Regional and Urban Investigation System for Environmental Research), which recently returned from nearly a year of studies in western Canada and is now focusing on southern Ontario and Quebec

Quentin Chiotti, *Pollution Probe*

Dr. Chiotti became the Air Programme Director and Senior Scientist at Pollution Probe in June 2002. He has a PhD in Geography from the University of Western Ontario, and has worked extensively in the area of climate change since 1993, including the Adaptation and Impacts Research Group of the Meteorological Service of Canada, Environment Canada (1995-2002). From 1998 - 2002 he was the scientific authority for an Environment Canada led multi-stakeholder study on atmospheric change in the Toronto-Niagara Region. He has published over 40 articles in scholarly journals and books, including co-editing a book on agricultural restructuring and sustainability, and was a contributor to the Canada Country Study, the first national assessment on climate change impacts and adaptation. Currently he is the co-lead author for the Ontario chapter of the 2007 national assessment on climate change impacts and adaptation. Dr. Chiotti has taught at various Universities across Canada (University of Guelph, Carleton University, University of Lethbridge, University of Toronto), and currently represents Pollution Probe on over a dozen environment-related advisory boards and committees, including the Ad Hoc Panel of the Air Management Committee, the Base Metals Environmental Multi-stakeholder Advisory Group (BEMAG), the Clean Air Foundation, the Canadian Climate Impacts and Adaptation

Network (Ontario), and the Environment and Food Committee of the Laidlaw Foundation.

Tony Clarke-Sturnman, *Shell International Petroleum*

Currently Manager Product Stewardship - Fuels and Bitumen Supply Chain with Shell International Petroleum, leading a small global team supporting the Fuels and Bitumen businesses' management of product risks. Studied Chemistry at Oxford and then Microbial Biotechnology for his PhD. Ten years R&D in biotechnology in Shell, working on water-soluble polymers produced by bacteria, was followed by spells in Public Affairs and Quality Management. He then edited publications on Additives and the Environment for the European industry association, the Additives Technical Committee. Since 1994, he has held several roles in the Shell Fuels business, including advising on the phase-out of leaded gasoline. He is a member of the joint IPIECA/OGP Health Committee and represents the Petroleum Industry at the UN sub-committee of experts for the Global Harmonised System of classification and hazard communication of Chemicals (UNSCEGHS). He spent 3 years as an elected representative of his community on the local district council. He has 2 patents. That on environmentally acceptable, clear, high-density, polymer-friendly drilling fluids has spawned a multi-million pound business.

Aaron J. Cohen, *Health Effects Institute*

Aaron J Cohen is currently Principal Scientist at the Health Effects Institute (HEI) in Charlestown, MA, where he has been employed since 1990. At HEI he manages an international program of epidemiologic research on the health effects of air pollution, and is involved in scientific program development. Dr. Cohen received his DSc in Epidemiology (1991), and Masters in Public Health (1985) from the Boston University School of Public Health. He also is a Registered Respiratory Therapist (AS and BS, Northeastern University), and worked as a therapist in newborn intensive care, and subsequently as Research Associate in Perinatal Epidemiology in the Joint Program in Neonatology at Brigham and Women's Hospital in Boston, where he conducted epidemiologic and clinical research on neonatal respiratory disease, and the evaluation of related medical technologies. Since 1994 Dr. Cohen has been an Adjunct Assistant Professor of Environmental Health at Boston University School of Public Health, where he lectures on environmental epidemiology.

Bart Croes, *California Air Resources Board*

Bart Croes is the Chief of the Research Division for the California Air Resources Board, with responsibilities for California's ambient air quality standards; climate change science and mitigation; health, exposure, atmospheric processes, and emissions control research; economic analyses; and indoor air quality. He was the Public Sector Co-Chair for the NARSTO Executive Assembly and former member of the National Research Council Committee on Research Priorities for Airborne Particulate Matter. He has been a peer reviewer for the National Research Council, the U.S. EPA, and numerous journals, and received the Editors' Citation for Excellence in Refereeing from the Journal of Geophysical Research. Mr. Croes has published peer-reviewed articles on air quality simulation modeling, emission inventory evaluation, reactivity-based VOC controls, acid deposition, the weekend effect for ozone and PM, PM data analysis and trends, and diesel particle traps.

Hadi Dowlatabadi, *University of British Columbia*

Dowlatabadi is Canada Research Chair & Prof in Applied Mathematics and Global Change, University of British Columbia. He is Associate Director of the Institute for Resources Environment and Sustainability and the Bridge Scholarship Program. He is a University Fellow at Resources for the Future and an Adjunct Professor at the Department of Engineering & Public Policy, Carnegie Mellon University. He is co-founder and Editor of the Integrated Assessment Journal and serves on the editorial boards of four other periodical. He is co-founder of Offsetters (www.offsetters.org) and Cooldrivepass (www.cooldrivepass.com) and a Director of Canadian Bioenergy Corporation. As a Rockefeller Foundation Warren Weaver Fellow he co-created Lead (www.lead.org). His academic research has focused on the interface between humans and the environment and systems approaches to decision-making under uncertainty. He studies problems in technology choice, acid rain, air quality, infectious and vector-borne diseases, energy policy, equity, ethics and climate change. He received his BSc in physics from Edinburgh University (1980) and his PhD in Physics from Cambridge University (1984). He has had the pleasure of working with more than 2 dozen scholars in completing their PhDs.

Kong Ha, *CAI-Asia*

Mr. Ha graduated from Michigan Technological University at Houghton, MI, USA, with a Master of Science in Mechanical Engineering in 1981. Upon graduation, he joined Cummins Engine Company at Columbus, IN and worked as an emission engineer for 3 years. Mr. Ha then moved to Toronto, Canada and joined Ortech International as an emission consultant with main responsibility in diesel and alternative fuel engines emission testing and development of diesel exhaust treatment systems. During this period, Mr. Ha obtained a US patent on a diesel particulate trap system. In 1992, Mr. Ha who was born in Hong Kong, returned to Hong Kong and joined Hong Kong Environmental Protection Department as a vehicle emission specialist. He was a senior officer in the Motor Vehicle Emissions Group of EPD till recently as the Motor Vehicle Emissions Group is now Motor Source Control Group. Mr. Ha's main responsibility is providing technical support to policy formulation/development related to motor vehicle emissions area. In recent years he has actively participated in many workshops on air quality internationally and across Asia. This has given him considerable insight into the air quality problems faced by other countries and cities. In addition to his full time responsibility, Mr. Ha is now the Chairman of the Clean Air Initiatives – Asia.

Anthony .J. Hedley, *University of Hong Kong*

Professor Hedley was trained in the medical schools of Aberdeen and Edinburgh and formerly worked in endocrinology and internal medicine before moving to the field of public health medicine. He has been an active researcher in chronic disease epidemiology, health services research and tobacco control for nearly 40 years. In 1983 he was appointed to the chair of public health in the University of Glasgow and since 1988 has been Professor of Community Medicine in Hong Kong and honorary consultant to the Hong Kong Department of Health and Hospital Authority. He has worked on environmental health issues in Hong Kong since 1989. He was Chairman of the Hong Kong Council on Smoking and Health from 1997-2002. In 1999 he was awarded a World Health Organisation medal for outstanding contributions to public health.

Philip K. Hopke, *Clarkson University*

Philip K. Hopke received BS in Chemistry from Trinity College, Hartford, CT in 1965 and his PhD degree from Princeton University in 1969. He was a research associate at MIT until August 1970. After 4 years as an assistant professor of chemistry at the State University College at Fredonia, NY, he joined the University of Illinois as a visiting assistant professor of chemistry. Beginning in 1975, Dr. Hopke joined the Institute for Environmental Studies where, in 1978, he was promoted to Associate Professor and in 1982, he became Professor of Environmental Chemistry with joint appointments in the Departments of Civil Engineering and Nuclear Engineering. On July 1, 1989, he joined the faculty of Clarkson University as the first Robert A. Plane Professor. He holds a joint appointment in the Department of Civil and Environmental Engineering. From July 1997 to June 1999, he served as Dean of the Graduate School and from July 1999 to June 2000, he was Chair of the Department of Chemistry and Head of the Division of Physical and Chemical Sciences. In July 2000 his principal appointment was moved to the Department of Chemical Engineering while retaining an appointment in the Department of Chemistry. As of January 1, 2002, he becomes the Bayard D. Clarkson Distinguished Professor and Director of the Center for Air Resources Engineering and Science.

Hugh Kellas, *Greater Vancouver Regional District*

Hugh Kellas is Manager of the Policy and Planning Department for the Greater Vancouver Regional District. The Department's activities, overseen by Hugh and a co-manager, include planning and monitoring for regional water, liquid waste and solid waste utilities; regional growth management planning; air quality monitoring, planning and management; regulation of air emissions, liquid waste sources and solid waste disposal; social housing policy; programs to reduce demand for utility services by business and the public; and other activities. Hugh is a Fellow of the Canadian Institute of Planners, and is a former President of both the Canadian Institute of Planners and the Planning Institute of British Columbia.

Tom Kosatsky, *Montreal Department of Public Health*

Tom Kosatsky MD (Manitoba), CIH (Dundee, Scotland), MPH (Emory) is a Montreal-based community medicine specialist. He maintains a clinical practice in occupational and environmental medicine, conducts research in environmental health for the Montreal Public Health Department and is Associate Professor of Epidemiology and Occupational Health at McGill University. During 2004-5, Dr. Kosatsky was engaged as epidemiologist to the WHO European Centre for Environment and Health in Rome, with responsibilities in the area of climate change and health. He continues to be involved in several ongoing European studies in this field. Much of his Montreal research is community based: current projects include estimation of the association between residence nearby local sources of air pollution and asthma in children, the analysis of the effect of neighbourhood social and economic conditions on the type and

frequency of care provided for asthma in children, influences on the participation of exposed workers in screening for sensitivity to beryllium, and indoor fungal exposure and disease. He directs several projects on heat and health, including an assessment of knowledge, attitudes, and practices of persons with chronic heart and chest conditions when confronted with heat episodes and/or smog alerts, and a GIS-based system to portray joint vulnerabilities to heat (environmental, residential, social, and medical) on a spatial basis.

Alan Krupnick, *Resources for the Future*

Alan Krupnick is Director of the Quality of the Environment Division and Senior Fellow at Resources for the Future. He holds a PhD in Economics from the University of Maryland and was a Senior Economist at the President's Council of Economic Advisors during the Clinton Administration. He is an expert on cost-benefit analysis, the valuation of non-market goods and activities, on survey techniques and on policy issues associated with the Clean Air Act. His work on applying contingent valuation techniques to estimate the value of health risk reductions is used in cost-benefit analyses by governments around the world. His most recent effort at valuing improvements in the Adirondacks is already being used by the USEPA and stakeholders to inform policy initiatives to reduce air pollution. Krupnick has served as a consultant to state governments, federal agencies, private corporations, the Canadian government, the European Union, the World Health Organization, the Asian Development Bank and the World Bank. For the international institutions, he has focused on environment and development issues in China, including projects on the development of regional emissions trading programs and on surveys to estimate the willingness to pay for mortality risk reductions. He co-chaired an 80-person advisory committee that counseled the USEPA on new ozone and particulate standards. He is a regular member of expert committees from the National Academy of Sciences and the USEPA and has served on a Royal Society of Canada committee analyzing ambient air quality standard-setting in Canada.

Michal Krzyzanowski, *WHO Europe*

Dr. Michal Krzyzanowski ScD, is responsible for the Air Quality and Health programme of the European Regional Office of World Health Organization. The tasks of the programme include review and synthesis of scientific evidence on health impact of air pollution on health as well as assessment of health burden of air pollution. He lead the WHO project "Systematic review of health aspects of air quality" implemented to support the development of European strategy on air quality (CAFE) in 2001-2004. He also chairs the Joint LRTAP Convention/WHO Task Force on Health Aspects of Air Pollution. Before joining WHO in 1991, Dr. Krzyzanowski conducted epidemiological research on health aspects of air pollution and other environmental factors in Poland, United States and France.

Brian McLean, *US EPA*

Not available

Marie O'Neill, *University of Michigan*

Marie O'Neill is Assistant Professor of Epidemiology and Environmental Health at the University of Michigan School of Public Health. She has an MS in Environmental Health Sciences from Harvard University and a PhD in Epidemiology from the University of North Carolina at Chapel Hill. She has worked for the U.S. Environmental Protection Agency, the Pan American Health Organization, in Mexico at the National Institute of Public Health and the National Center for Environmental Health as a Fulbright Scholar, and as a Research Fellow in Environmental Epidemiology at the Harvard School of Public Health. Her research interests include health effects of air pollution and temperature extremes (mortality, asthma, and cardiovascular endpoints); climate change and health; international health; environmental exposure assessment; and socio-economic influences on health.

William Pennell, *NARSTO*

Dr. Pennell is Management Coordinator of NARSTO -- a public/private partnership among industry, government agencies, and academia in Canada, the United States, and Mexico -- which works to improve the scientific basis of air-quality management in North America. NARSTO performs scientific assessments, conducts workshops and special scientific studies, and it operates the NARSTO Quality Systems Science Center, which provides a central data archive for NARSTO-related field activities. Before becoming NARSTO Management Coordinator, Dr. Pennell was principal line manager for Pacific Northwest National Laboratory's (PNNL) research activities in atmospheric science and global change. The scope of these activities ranged from basic research into the processes responsible for the formation, transport, removal, and environmental impact of energy-related pollutants (and expression of these processes in numerical models) to basic economic and social science research that is focused on obtaining a predictive understanding of the effects of climate and other environmental change on human activities.

Jonathan M. Samet, *Johns Hopkins University*

Jonathan M. Samet, MD, MS, is Professor and Chairman of the Department of Epidemiology of the Johns Hopkins Bloomberg School of Public Health. Dr. Samet received a Bachelor's degree in Chemistry and Physics from Harvard College, an MD degree from the University of Rochester School of Medicine and Dentistry, and a Master of Science degree in epidemiology from the Harvard School of Public Health. He is trained as a clinician in the specialty of internal medicine and in the subspecialty of pulmonary diseases. From 1978 through 1994, he was a member of the Department of Medicine at the University of New Mexico. At the Johns Hopkins University Bloomberg School of Public Health, he is Director of the Institute for Global Tobacco Control, a WHO Collaborating Center, and Co-Director of the Risk Sciences and Public Policy Institute. His research has addressed the effects of inhaled pollutants in the general environment and in the workplace. He has written widely on the health effects of active and passive smoking and served as Consulting Editor and Senior Editor for Reports of the Surgeon General on Smoking and Health and the National Cancer Institute's Monographs on Tobacco Control. He testified against the tobacco industry in litigation brought by the State of Minnesota and the U.S. Department of Justice. He has edited books on the epidemiology of lung cancer and on indoor and outdoor air pollution. He has served on the Science Advisory Board for the U.S. Environmental Protection Agency and was Chairman of the Biological Effects of Ionizing Radiation Committee VI and the Committee on Research Priorities for Airborne Particulate Matter of the National Research Council. He presently chairs the Board on Environmental Studies and Toxicology of the National Research Council. For the Institute of Medicine, he was Chair of the Committee on Asbestos: Selected Health Effects, and is presently Chair of the Committee on Evaluation of the Presumptive Disability Decision-Making Process for Veterans. He was elected to the Institute of Medicine of the National Academy of Sciences in 1997. He received the Surgeon General's Medallion in 1990 and 2006 for his work on the Surgeon General's Reports and the Prince Mahidol Award, from the King of Thailand, in 2005 for his work on air pollution.

Annemoon van Erp, *Health Effects Institute*

Annemoon van Erp is a senior scientist at the Health Effects Institute, Charlestown,MA were she has been employed since 1999. She received a PhD in neurobiology (1993) from Leiden University, the Netherlands, and was Research Assistant Professor at Tufts University before joining HEI. Her research interests are the toxicology of fuel additives such as MTBE and manganese, evaluation of new diesel engine and aftertreatment technologies, nanotechnology, and accountability. In addition, she enjoys managing the Walter Rosenblith New Investigator program and HEI's website, and maintains an interest in sustainability, biodiversity, and climate change issues.

Martin Williams, *UK Department of Environment*

Martin Williams is currently Head of the Air and Environment Quality Division of the Department for Environment, Food and Rural Affairs in the UK. He graduated in Chemistry from the University of Wales, Cardiff, and took a PhD in Chemistry at the University of Bristol. After postdoctoral research at UBC in Vancouver, Canada, he joined the air pollution division of the Government's Warren Spring Laboratory, becoming its Head in 1982. After a period as technical assistant to the Chief Scientist of the Department of Trade and Industry, he joined the then Department of the Environment to head the Science Unit on Air Quality, becoming Head of the Division in 2002, where he has responsibility for policy on air quality and industrial air pollution control. He is currently chairman of the Executive Body of the UNECE Convention on Long Range Transboundary Air Pollution.

DELEGATES

Name	Organization	Break Out Group
Ryan Allen	*Simon Fraser University, Burnaby BC*	1
Jane Barton	*Environment Canada, Gatineau QC*	1
Alex Basiji	*Health Canada, Toronto ON*	1
Paul Baynham	*Northland Regional Council, Whangarei, New Zealand*	2
Richard Bennett	*Ministry of Environment, Victoria BC*	1
Carmelita Biagtan	*BC Lung Association, Vancouver BC*	2
Maryse Bouchard	*Institut national de santé publique du Québec, Montréal QC*	1
Michael Brauer	*University of British Columbia, Vancouver BC*	2
David Briggs	*Imperial College London, London, U.K.*	1
Jeffrey Brook	*Environment Canada, Toronto ON*	2
Christina Cheng	*Ontario Ministry of the Environment, Etobicoke ON*	1
Quentin Chiotti	*Pollution Probe, Toronto ON*	2
Anthony Clarke-Sturman	*Shell International Oil Products, London, U.K.*	2
Aaron Cohen	*Health Effects Institute, Boston MA*	2
Ray Copes	*BC Centre for Disease Control, Vancouver BC*	1
Lorraine Craig	*University of Waterloo, Waterloo ON*	1
Bart Croes	*California Air Resources Board, Sacramento CA*	2
Hadi Dowlatabadi	*University of British Columbia, Vancouver BC*	2
Louis Drouin	*Montreal Public Health, Montreal QC*	1
Ali Ergudenler	*Greater Vancouver Regional District, Burnaby BC*	2
Long Fu	*Alberta Environment, Edmonton AB*	1
Maria Furberg	*RWDI AIR Inc., Vancouver BC*	2
Larry Gephart	*ExxonMobile Biomedical Sciences, Annandale, NJ*	2
Scott Giffin	*Public Health Services New Brunswick, Saint John NB*	2
Bruce Gillies	*Environment Canada, Toronto ON*	1
Stephanie Gower	*University of Waterloo, Waterloo ON*	2
Andrew Green	*Environment Canada, Vancouver BC*	1
Kong Ha	*Government of the Hong Kong SAR, Hong Kong, China*	1
Jamal Harb	*Health Canada, Burnaby BC*	2
Anthony Hedley	*University of Hong Kong, Hong Kong, China*	2
Steven Hilts	*Teck Cominco Metals Ltd., Trail BC*	1
Philip Hopke	*Clarkson University, Postdam NY*	2
Tracey Inkpen	*Environment Canada, Dartmouth NS*	1
Rand Jackson	*Natural Resources Canada, Ottawa ON*	2
Matti Jantunen	*National Public Health Institute of Finland, Kuopio, Finland*	1
Derek Jennejohn	*Greater Vancouver Regional District, Burnaby BC*	2
Sam Kacew	*University of Ottawa, Ottawa ON*	1
Bruce Kay	*Environment Canada, Vancouver BC*	2
Roger Keefe	*Imperial Oil Ltd., Calgary AB*	2
Hugh Kellas	*Greater Vancouver Regional District, Burnaby BC*	1
Norman King	*Montreal Public Health, Montreal QC*	1

Graham Kissack	*Catalyst Paper, Vancouver BC*	1
Tom Kosatsky	*Montreal Public Health, Montreal QC*	2
Dan Krewski	*University of Ottawa, Ottawa ON*	1
Alan Krupnick	*Resources for the Future, Washington DC*	2
Michal Krzyzanowski	*World Health Organization, Bonn, Germany*	2
Brian McLean	*USEPA, Washington DC*	2
Addy Mitchell	*University of Waterloo, Waterloo ON*	1
Curtis Moore	*Health & Clean Air Newsletter, McLean VA*	2
Jack Nickel	*Health Canada, Burnaby BC*	1
Marie O'Neill	*University of Michigan, Ann Arbor MI*	1
Glen Okrainetz	*Ministry of Environment, Victoria BC*	1
William Pennell	*NARSTO Management, Pasco WA*	2
Karla Poplawski	*University of Victoria, Victoria BC*	2
Franck Portalupi	*Environment Canada, Gatineau QC*	2
Kathy Preston	*RWDI AIR Inc., Vancouver BC*	1
Gloria Rachamin	*Ontario Ministry of Health, Toronto ON*	2
Michael Rensing	*Ministry of Environment, Victoria BC*	1
Nigel Routh	*Department of Environment & Conservation (NSW), Sydney, Australia*	2
Jonathan Samet	*Johns Hopkins Bloomberg School of Public Health, Baltimore MD*	2
Markus Schulze	*ThyssenKrupp Steel, Duisburg, Germany*	2
Eleanor Setton	*University of Victoria, Victoria BC*	1
John Shortreed	*University of Waterloo, Waterloo ON*	2
Adolfo Silva	*Canadian Petroleum Products Institute, Toronto ON*	1
Dean Stinson O'Gorman	*Environment Canada, Gatineau QC*	2
Brian Stocks	*Ontario Lung Association, Windsor ON*	1
Natalie Suzuki	*Ministry of Environment, Victoria BC*	2
Rob Taalman	*Shell International, The Hague, Netherlands*	1
Suzanne Therien	*University of Ottawa, Ottawa ON*	2
Annemoon van Erp	*Health Effects Institute, Boston MA*	1
Anton Van Heusden	*Environment Canada, Gatineau QC*	2
Cindy Walsh	*BC Ministry of Environment, Surrey BC*	1
Corinna Watt	*Environment Canada, Edmonton AB*	2
Martin Williams	*DEFRA, London U.K.*	1

Guidance Document on

Air Quality Risk Management

Prepared for NERAM V Colloquium

Strategic Policy Directions for
Air Quality Risk Management

October 16-18, 2006

Morris J. Wosk Centre for Dialogue
Simon Fraser University
Vancouver, B.C.

DRAFT for Discussion

Executive Summary

To be prepared following the Colloquium

Overarching Guidance Statements

The Colloquium series over the last five years has seen a rapid development of partial resolutions to key issues in air quality risk management and the emergence of a new regulatory paradigm to augment traditional public health standard setting. Underlying these developments are two key overarching guidance statements. These statements synthesize key ideas expressed at NERAM III (Rome 2003) and NERAM IV (Mexico 2005) on future directions for air quality risk management. The statements are also intended to capture the current thinking of public health organizations (i.e. WHO Air Quality Guidelines Global Update - 2006) and NERAM Colloquium planning committee members.

1. Health has traditionally been the major driver for air quality standard setting. There is sufficient scientific evidence to indicate that air pollution causes significant mortality and morbidity around the world. While there are still uncertainties to guide effective source-specific air quality management strategies, there are no regrets strategies, such as general reductions in emissions of air pollution sources. It is now recognized that care is needed in selecting policies to improve air quality that will also be health cost-effective.
2. At NERAM I (Ottawa 2001), Bob Maynard of the UK Department of Health suggested that the traditional standard setting approach to air quality management to protect public health was perhaps not the best approach. Over the course of the colloquium series this idea has gained momentum and acceptance through the discussion of "hot spots", inability to meet air quality standards in areas such as busy roadways, and an appreciation of the no threshold, linear nature of damage such that reduction in air pollution at any level has health benefits. The NERAM guidance supports taking both a traditional approach to air quality with standard setting, but to add to this a policy framework that recognizes that in some locations standards will be increasingly costly to achieve and will not be met for some period of time. The extension of the policy framework will incorporate concepts such as; i) maximizing benefits within given resources, ii) examining trade offs between location/time, fairness and overall population health improvements, iii) consideration of non traditional interventions that recognize the complexity and intricacies of the source to receptor systems for air quality – for example market mechanisms that need not have accurate cause and effect knowledge to effect improvements, and v) risk communication to advise the population of alternative policies in order to determine acceptable trade-offs between fairness and population health.

A number of interrelated concepts collectively recommend a new approach to air quality risk management. These include:

a) inability to meet standards without a gross disproportion of expenditure on specific locations and time periods;

b) the precautionary approach to deal with uncertainty and find acceptable public policy. This includes the need for extensive risk communication and consultation (see New Directions Group report http://www.newdirectionsgroup.org/projects/precaution.php-; Ortwin Renn's dialogue criteria; Dealing Sensibly with Risks by the Netherlands Environmental Assessment Agency - a differentiated approach to overcome the shortcomings of current risk principles);

c) regulatory fairness;

d) success of cap and trade approaches in other areas such as acid rain, particularly the ability of this concept to stimulate new technologies and business arrangements;

e) trends towards longer transboundary distances as the critical PM characteristic goes from PM_{10} to $PM_{2.5}$ to PM_{10}. This makes the impact of regional and international events on local air quality much more significant. This is one dimension of globalization which creates a regulatory challenge for national governments; continuing uncertainty as to which PM particles are more harmful to health

f) a "risk based" approach with explicit treatment of uncertainty in evaluation of options for the most health benefit for the least expenditure, compatible with stakeholders concerns – including ALARA;

g) co-benefits of air pollution and global warming and implications for modifying policy option selection;

h) keeping clean areas clean.

The Guidance document will build these concepts into a framework which might look like Figure 1.

Development of the Guidance Document

This draft of the Guidance Document will be revised based on inputs from the Vancouver Colloquium and inputs from the participants. A final draft will be circulated for comment to all participants and the Planning committee.

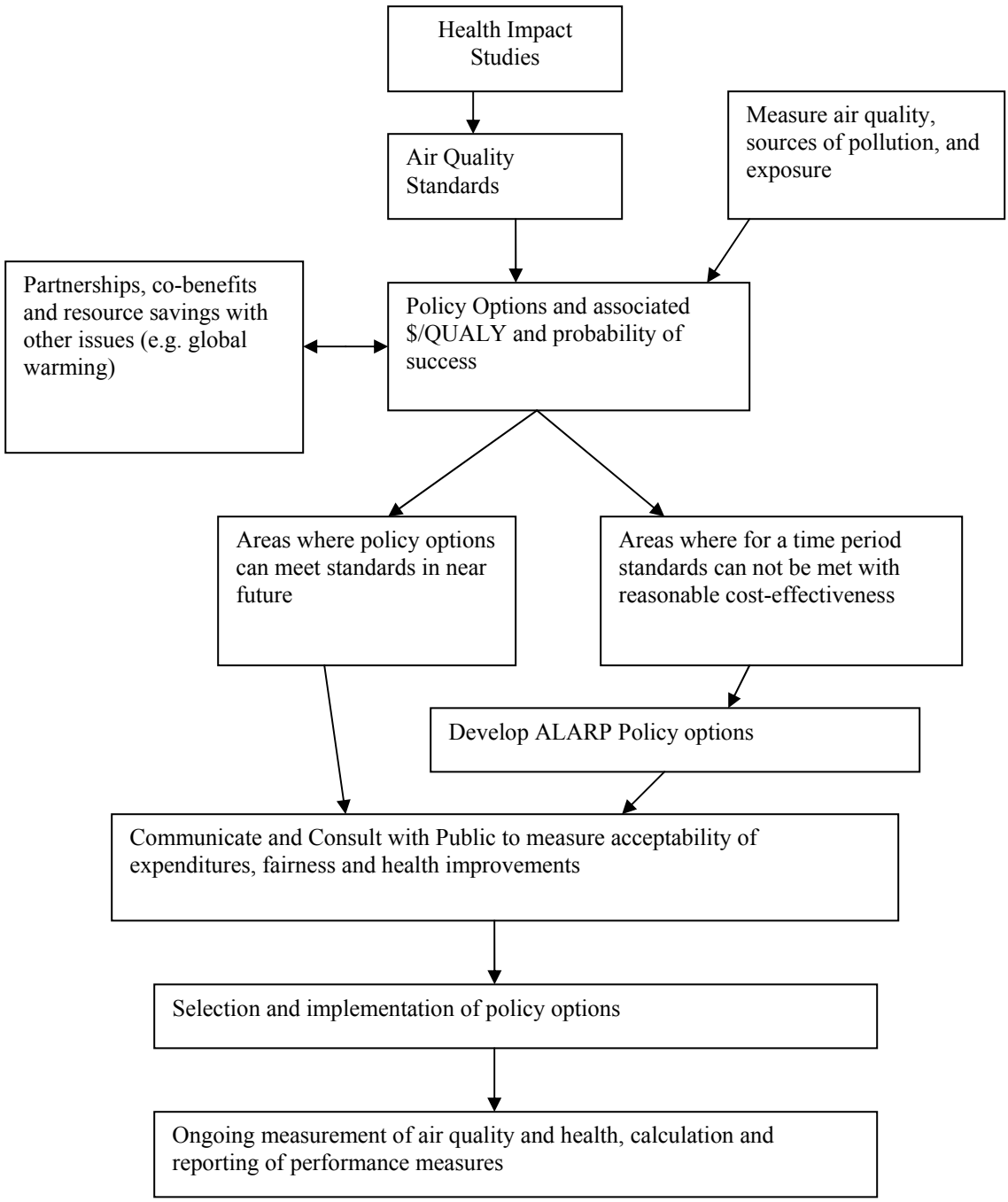

Figure 1 Possible Framework for an Approach to Air Quality and Health

CHAPTER 1 - Introduction

Lorraine Craig[1], John Shortreed[1], Jeffrey R. Brook[2]

[1]Network for Environmental Risk Assessment and Management, University of Waterloo
[2]Environment Canada

Air quality projections in several locations in developed and developing countries indicate that pollutant levels may not be significantly reduced over the next 15 to 20 years. In many cases, sizable expenditures and/or significant societal changes will be required to meet ambient air quality standards.

While there are some uncertainties, there is extensive scientific evidence of population health effects associated with short and long term exposure to ambient air pollution, even in areas where the standards are already met. Air quality decision-makers are faced with uncertainties concerning the costs of abatement, identifying pollutants and sources that are most harmful, the magnitude of public health benefits associated with emission reduction measures, and the extent to which present day and future transboundary and intercontinental airflows will compromise local and regional efforts to control air pollution. A more important challenge, however, is that as the more obvious cost- effective emissions control options are implemented, decision-makers are faced with uncertainty concerning how to achieve further reductions with the greatest health benefit per unit cost of reduction.

Given the contribution and importance that emissions from local sources have to regional, continental and global airsheds, it is critical that local emission reduction initiatives are an integral part of national and global clean air strategies. The effectiveness of new market-based mechanisms such as emission trading schemes and legal approaches to air quality management has not been clearly demonstrated. There are opportunities to achieve sizable co-benefits through joint strategies for greenhouse gas mitigation and air pollutant emission reduction.

Clean air is an important aspect of quality of life. As population growth, urban sprawl and the number of vehicles and other sources increases, the impacts of air pollution on quality of life become more apparent, including impaired visibility, breathing difficulties among asthmatics and the elderly, restrictions in outdoor physical activity, etc. Long-term exposures are associated with the development of chronic disease and a loss of life expectancy, estimated to be over 8 months due to $PM_{2.5}$, equivalent to 3.6 million life years lost annually in the EU (CEC, 2005). Rising public concern and demand for governments to take further action to improve air quality suggest that guidance to support policy-makers in formulating wise air quality management strategies is timely.

This Guidance Document aims to serve as a reference for air quality policy-makers and managers and by providing state of the art, evidence-based information on key determinants of air quality management decisions. The Document reflects the findings of the five annual meetings of the NERAM (Network for Environmental Risk Assessment and Management) International Colloquium Series on Air Quality Management, as well as the results of supporting international research.

The contributors to the Guidance Document are recognized experts in the science and policy dimensions of air pollution and health. They represent a range of international perspectives including academia (*Michael Brauer, University of British Columbia; Daniel Krewski, McLaughlin Centre for Population Health Risk Assessment, University of Ottawa; Jonathan Samet, Johns Hopkins University*); state and national government organizations (*Jeffrey Brook, Environment Canada; Martin Williams, UK Environment; Jurgen Schneider, Austrian FEA; Andre Zuber, European Commission*); international organizations (*Michal Krzyzanowski, WHO European Centre for Environment and Health; Bill Pennell, NARSTO*); non-governmental organizations (*Quentin Chiotti, Pollution Probe; Alan Krupnick, Resources for the Future*) and industry (*Geoff Granville, Shell Canada Ltd.*).

Insights from the NERAM Colloquium Series

The NERAM Colloquium Series was launched in 2001 with the objective of strengthening linkages between the scientific and policy communities to improve air quality policy decisions and public health. The series was spearheaded by NERAM in collaboration with an international multi-stakeholder steering committee including representatives from national-level regulatory agencies in Canada, the US, Europe, and South East Asia, as well as international environment and health organizations, industry groups, state and provincial regulators, environmental non-governmental organizations, and academia. Five annual meetings were held in Canada (University of Ottawa - 2001), the US (Johns Hopkins University - 2002, Europe (Rome E Health Authority - 2003), Mexico (National Institute for Public Health – 2005), and Canada (Vancouver – 2006).
Over the course of the Series, a number of key principles for air quality management were proposed both in individual keynote presentations and through collective opinions expressed in break-out groups (See Appendix A for Conference Statements from Rome and Mexico Colloquia). Perhaps the most unanticipated observation was the recognition by regulators of the limitations of traditional regulatory standard-setting as a key air quality management strategy. While air quality standards have historically and continue to play a central and useful role in regulating air pollutants, the findings of key epidemiological studies suggest that air quality management based on standard-setting for single pollutants is simplistic and probably suboptimal in protecting public health.

For example, particulate matter mass is a good starting indicator for a broad class of what is recognized to be a serious threat to human health. However, cost-effective air particulate strategies require an understanding of:

i) local components of the mixture including size, chemical constituents (e.g. ultrafines, organic species, metals);
ii) sources of the various components;
iii) effects on health of the various components, their potential interactions with and synergistic and/or additive effects with gaseous air pollutants, and the benefits likely to accrue from various reductions; and
iv) the costs of reducing the various components. In certain situations, including so called “hot spots”, the estimated costs of additional abatement requirements to achieve incrementally smaller pollutant reductions to meet air quality standards may outweigh any related public health benefits (Maynard, 2003; Maynard et al. 2003; Williams, 2005; Craig et al. in press).

Innovative approaches that focus on reducing harmful exposures in a cost-effective way are required to make further gains in air quality and public health. The Guidance Document provides a forward-looking perspective based on lessons learned and best practice in air quality management to guide decision-makers towards the development of cost-effective air quality management strategies.

Structure of the Document

A conceptual framework for air quality policy development was proposed by NERAM to provide a foundation for the Colloquium series presentations and discussions (see Figure 1). The framework identifies key factors underlying the policy process and illustrates the interplay between scientific assessments of air quality and health effects, policy analysis to assess costs and benefits of proposed options, and aspects of the policy environment (fairness, equity, stakeholder acceptability, technical feasibility, enforceability, government commitment) that influence decision-making. The framework recognizes that scientific uncertainty is inherent in the inputs to the decision-making process. The topics covered in the Guidance Document address the key Framework elements.

Chapter 2 reviews the scientific evidence on the health effects of exposure to ambient air pollution. The chapter reflects the Colloquium series’ focus on the health significance of exposures to particulate matter. Evidence from epidemiological, toxicological and clinical studies in Canada, the United States, Europe, and internationally will be presented. The chapter also summarizes new insights from emerging literature and address challenges for risk management.

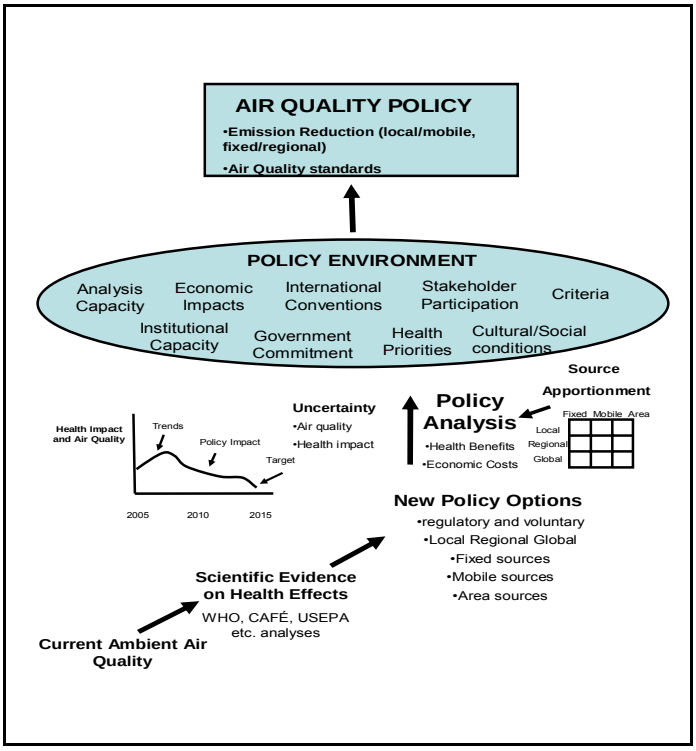

Figure 1. NERAM Air Quality Policy Development Framework

Chapter 3 provides an overview of the role of ambient air quality measurement, emission inventories and modeling in air quality management. The Chapter provides examples from North America and Europe to illustrate the current status, strengths and limitations of emission inventories, air quality monitoring networks and air quality modeling activities. The Chapter provides guidance on current best practice to inform the development of measurement, monitoring and modeling capacity relevant to air quality management policy development and policy evaluation.

Chapter 4 presents strategies for improving ambient air quality at the local, regional and global levels. Case studies from North America, and Europe provide examples to illustrate each of the approaches and identify factors associated with successful policy development and implementation. Evidence of effectiveness is emphasized. The Chapter reviews policy approaches including mobile source, point source and areas source emission reductions strategies; standard-setting approaches, economic approaches, transboundary strategies, and multi-pollutant strategies; and public education/behavioral approaches.

Chapter 5 discusses key emerging issues faced by air quality managers and policy makers with the growing awareness of the health impacts of poor air quality and the increasing costs to achieve further reductions. These issues include the challenges managing hot spots and environmental justice and equity considerations. Innovative policy initiatives to complement standards-based air quality management approaches are identified, including integrated strategies oriented towards achieving climate change co-benefits and broader sustainability objectives.

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CHAPTER 2. Air Quality and Human Health

J. Samet, D. Krewski, M. Krzyzanowski and L. Craig

KEY MESSAGES

- A substantial body of epidemiological evidence now exists that establishes a link between exposure to air pollution, especially airborne particulate matter, and increased mortality and morbidity, including a wide range of adverse cardiorespiratory health outcomes. Many time-series studies, conducted throughout the world, relate day to day variation in air pollution to health with remarkable consistency. A smaller number of longer-term cohort studies find that air pollution increases risk for mortality.
- Health effects are evident at current levels of exposure, and there is little evidence to indicate a threshold concentration below which air pollution has no effect on population health.
- It is estimated that the shortening of life expectancy of the average population associated with long-term exposure to particulate matter is 1-2 years.
- Recent epidemiological studies show more consistent evidence of lung cancer effects related to chronic exposures than found previously.
- In general, methodologic problems with exposure classification tend to diminish the risks observed in epidemiological studies so that the true risks may be greater than observed.
- Human clinical and animal experimental studies have identified a number of plausible mechanistic pathways of injury, including systemic inflammation, that could lead to the development of atherosclerosis and alter cardiac autonomic function so as to increase susceptibility to heart attack and stroke.
- The question of which physical and chemical characteristics of particulate matter are most important in determining health risks is still unresolved. There is some evidence to suggest that components related to traffic exhaust and transition metal content may be important.
- Despite continuing uncertainties, the evidence overall tends to substantiate that PM effects are at least partly due to ambient PM acting alone or in the presence of other covarying gaseous pollutants.
- Several studies of interventions that sharply reduced air pollution exposures found evidence of benefits to health. New findings from an extended follow up of the Six City study cohort show reduced mortality risk as $PM_{2.5}$ concentrations declined over the course of follow-up. These studies provide evidence of public health benefit from the regulations that have improved air quality.

2.1 Introduction

The primary objective of any air quality management strategy is to protect human health and the environment. From a policymaker's perspective, several key questions on the issue of health effects arise: i) what is currently known about the impacts of air pollution on public health, ii) which populations are most susceptible, iii) which sources are most damaging to health, iv) what levels of air pollution are safe and how much health improvement can be expected with air quality improvements. A background paper prepared for the NERAM III Colloquium *"Strategies for Clean Air and Health"* held in Rome in 2003 framed the discussion of scientific evidence on health effects around these key policy questions. A number of major critical reviews have since been published by the World Health Organization (2005, 2006), the US Environmental Protection Agency (2004; 2005; 2006) and Air & Waste Management Association (Pope and Dockery, 2006). This chapter will build on the Rome background paper by presenting new evidence and conclusions from these major reviews.

The focus of this capstone document, as for the NERAM Colloquium series, is on the scientific understanding of outdoor air pollution and its implications for evidence-based risk management. However, there needs to be recognition that air pollution is a broader public health problem with implications for children and adults worldwide. While much of the epidemiological evidence linking air pollution exposures to health impacts focuses on measures of air quality and health in North America and Europe, for millions of people living in developing countries, indoor pollution from the use of biomass fuel occurs at concentrations that are orders of magnitude higher than currently seen in the developed world. Deaths due to acute respiratory infection in children resulting from these exposures are estimated to be over 2 million per year (Brunekreef and Holgate, 2002). While indoor air pollution is responsible for up to 3.7% of the burden of disease in high mortality developing countries, it is no longer among the top 10 risk factors in industrialized countries in regard to burden of disease. More information about indoor air pollution and its consequences can be found in several recent reviews (WHO, 2002; CARB, 2005).

2.2 Effects of Air Pollution on Population Health

Air pollution is pervasive throughout the world, and represents one of the most widespread environmental threats to the population's health. The World Health Organization (2002) has identified ambient air pollution as a high priority in its Global Burden of Disease initiative, estimating that air pollution is responsible for 1.4% of all deaths and 0.8% of disability-adjusted life years globally. Although the magnitude of the estimated increased risk might appear to be small, the numbers of people affected are large when extrapolated to the entire population.

NERAM III convened 200 air quality scientists, policymakers, industry representatives and non-governmental organizations from 22 countries to exchange perspectives on the interface between policy and science on air pollution health effects, air quality modeling, clean air technology, and policy tools. The Conference Statement (http://www.irr-neram.ca/rome/rome.html), which was based on breakout group discussions, keynote presentations from North America and Europe and plenary discussions, highlighted the importance of air pollution as a local, national, and global public health concern.

Despite the seemingly consistent message from the public health community with regard to the need for reduction of risk to the extent possible, there are unresolved scientific issues with attendant uncertainties that are problematic for decision-makers. The recent decision by the United States Environmental Protection Agency (US EPA) to retain the annual average standard for $PM_{2.5}$ of 15 $\mu g/m^3$ averaged over 3 years, despite the recommendation of US EPA's Clean Air Scientific Advisory Committee (CASAC) for a lower value, is illustrative of how controversy can arise in the setting of uncertainty. In fact, as air pollution levels have declined in North America and Europe, epidemiological studies become less likely to detect the smaller absolute effects that would be anticipated and methodologic concerns assume greater credibility as an alternative to causation in producing observed findings. Uncertainty continues to persist even though many methodological concerns around epidemiological studies have now been addressed and several key reanalyses have been carried out. For example, the extensive reanalysis of two prospective cohort studies, the Harvard Six Cities Study and the American Cancer Society's Cancer Prevention Study II (Krewski et al., 2000; 2004; 2005a; 2005b), confirmed the original findings. Large, pooled time series studies have also been carried out that produce more precise risk estimates than single city studies, as frequently reported in the past (Stieb et al, 2002).

Scope of Health Concerns

The range of adverse health effects associated with exposure to air pollution has often been depicted as a pyramid (Figure

1). In this formulation, a smaller proportion of the population is affected by the most severe health outcomes such as premature death, hospital admissions and emergency room visits and a greater proportion is impacted by conditions that affect quality of life such as asthma exacerbations that result in work or school absences and by subclinical effects, such as slowed lung function growth in childhood and accelerated development of atherosclerosis. The range of effects is broad, affecting the respiratory and cardiovascular systems and impacting children, the elderly, and those with pre-existing diseases such as COPD and asthma. The risk for various adverse health outcomes has been shown to increase with exposure and there is little evidence to suggest a threshold below which no adverse health effects would be anticipated (WHO, 2005).

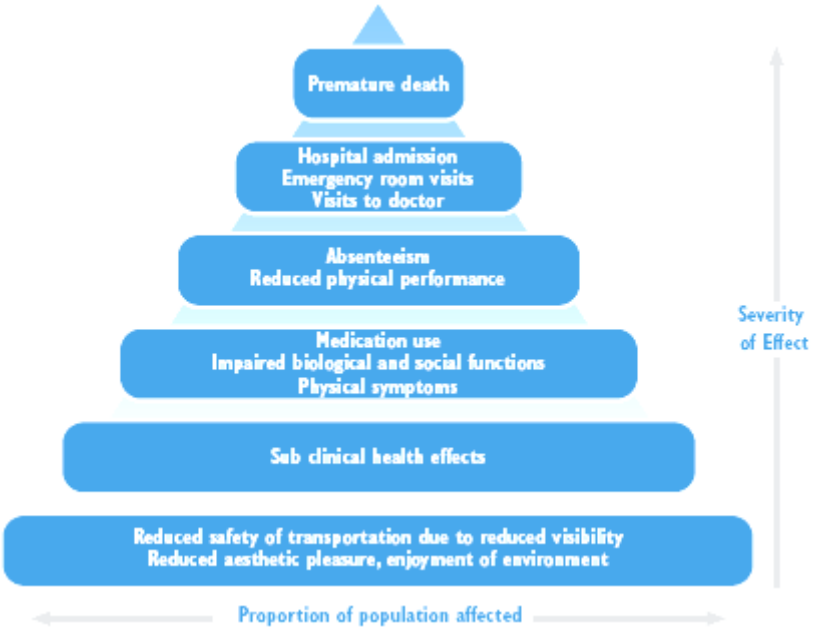

Figure 1. Pyramid of Air Pollution Health Effects

Source. British Columbia. Provincial Health Officer (2004). Every Breath you Take..Provincial Health Officer's Annual Report 2003. Air Quality in British Columbia, a Public Health Perspective. Victoria, BC. Ministry of Health Services. Adapted from Health Effects Air pollution (Pyramid of Health Effects), by Health Canada

Figure 2 describes the range of health outcomes measured in epidemiological and human clinical studies. The impacts of short term and long term air pollution exposures have been studied extensively in North America and Europe for health endpoints towards the peak of the pyramid (i.e. premature death, hospital admissions and emergency room visits). More recent studies have examined the health effects of air pollution in low and middle income countries where air pollution levels are the highest. The scope of health concerns has broadened from an emphasis on total morbidity and mortality from respiratory causes, such as exacerbations of chronic respiratory diseases, including COPD and asthma, and the respiratory health of children to several adverse cardiac and reproductive outcomes and impacts on susceptible subpopulations, including those with preexisting cardiopulmonary illnesses, children and older adults. Numerous recent single-city studies have expanded the health endpoints reported to be associated with PM exposures including 1) indicators of the development of atherosclerosis with long-term PM exposure; 2) indicators of changes in cardiac rhythm, including arrhythmia or ST-segment changes; 3) effects on developing children and infants; 4) markers of inflammation such as exhaled NO; and 5) effects on organ systems outside the cardiopulmonary systems (USEPA, 2006). The long-range implications for individuals of some of the intermediate markers of outcome remain to be established, but nonetheless they offer usual indicators of population health.

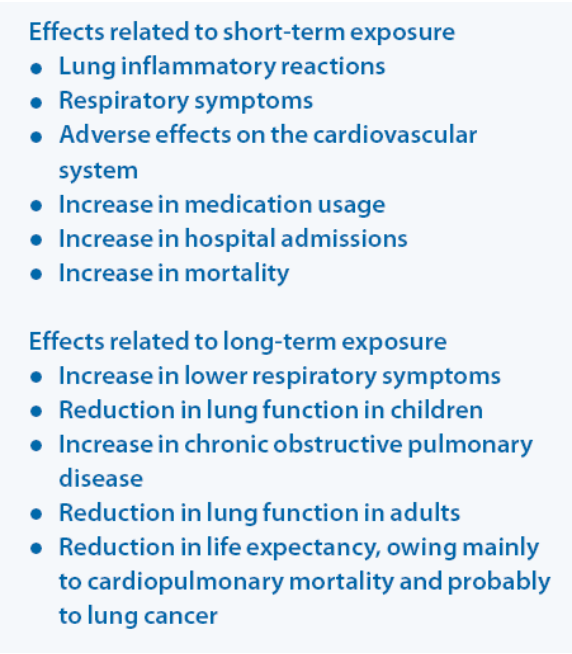

Figure 2. Health Outcomes Measured in Studies of Epidemiological and Human Clinical Studies
Source: WHO (2006)

2.3 Lines of Evidence

Sources of evidence from which to assess the health effects associated with air pollution exposures include observational epidemiology, toxicological and clinical studies. The findings of these different lines of investigation are complimentary and each has well-defined strengths and weaknesses. The findings of epidemiological studies have been assigned the greatest weight in standard-setting for airborne particles because they characterize the consequences of the exposures that are actually experienced in the community setting.

Epidemiologic Evidence

The evidence on airborne PM and public health is consistent in showing adverse health effects at exposures experienced in cities throughout the world in both developed and developing countries. The epidemiological evidence shows adverse effects of particles associated with both short term and long term exposures. Adverse health effects have been demonstrated at levels just above background concentrations which have been estimated at 3-5 ug/m^3 in the United States and western Europe for $PM_{2.5.}$ (WHO, 2005).

Mortality and Long term PM exposure

Associations between air pollution exposure and mortality have been assessed mainly through two types of epidemiological studies. Cohort studies follow large populations for years and typically relate mortality to an indicator of average exposure to PM over the follow-up interval. Time series studies investigate the association between daily mortality and variation in recent PM concentrations. To establish standards for short term exposures, regulatory agencies rely on the findings of time series studies while findings of cohort studies are used to establish annual standards.

Long term cohort studies of PM and mortality are fewer in number than those of day to day variations. They are typically expensive to carry out and require a substantial number of participants, lengthy follow-up and information on PM exposure as well as potential confounding and modifying factors. Most of the studies have been carried out in the US but findings have also been reported for two European studies. Two studies of the health effects of long term exposure to air pollution in large populations have been used extensively in the development of ambient air quality standards for PM_{10} and $PM_{2.5}$.

The Harvard Six Cities Study (Dockery et al., 1993) was the first large, prospective cohort study to demonstrate the adverse health impacts associated with long term air pollution exposures. This study demonstrated that chronic exposure to air pollutants is independently related to cardiovascular mortality. In the group of 8,111 adults with 14 to 16 years of follow up,

the increase in overall mortality for the most-polluted city versus the least polluted city was 26%. The range of exposure to PM across the six cities was 11 to 29.6 $\mu g/m^3$ for fine particles

The American Cancer Society established its Cancer Prevention Study (CPS) II in the early 1980s. A subcohort with air pollution data available for counties of residence has been used to assess mortality in relation to air pollution (Pope et al. 1995). The cohort includes approximately 552,138 adults who resided in all 50 states. This study linked chronic exposure to multiple air pollutants to mortality over a 16 year period. In these two studies robust associations were reported between long term exposure to $PM_{2.5}$ and mortality (Dockery et al., 1993; Pope et al., 1995).

An independent reanalysis of these two studies was undertaken by the Health Effects Institute in response to industry demands and a Congressional request (Krewski et al., 2000, Pope 2002). The HEI re-analysis largely corroborated the findings of the two studies. In the Six Cities Reanalysis the increase in all-causes of death linked to fine particles was 28 percent across the pollution gradient from the most to the least polluted city, compared to the original estimate of 26%. For the ACS study, the increased risk of all cause death associated with fine particles was 18% in the reanalysis, compared to 17% reported by the original investigators. An extended follow up of the ACS study indicated that the long term exposures were most strongly associated with mortality from ischemic heart disease, dysrhythmias, heart failure and cardiac arrest (Pope et al. 2004). For these cardiovascular causes of death, a 10 $\mu g/m^3$ elevation of $PM_{2.5}$ was associated with an 8-18% increase in risk of death. Mortality attributable to respiratory disease had relatively weak associations. Recent analysis of the Los Angeles component of the ACS cohort suggests that the chronic health effects associated with within-city gradients in exposure to $PM_{2.5}$ may be even larger than those reported across metropolitan areas (Jerrett et al. 2005).

An extended analysis to include deaths to the year 2000 confirmed previous findings. The increased risk of all cause and cardiopulmonary and lung cancer death rose 18 to 30 percent respectively, though that of lung cancer was 2 % (Pope et al., 2002).

Laden's (2006) report on the extended follow up of the Harvard Six Cities Study found effects of long term exposure to particulate air pollution that are consistent with previous studies. Total, cardiovascular, and lung cancer mortality were positively associated with ambient $PM_{2.5}$ concentrations. Reduced $PM_{2.5}$ concentrations (mean $PM_{2.5}$ concentrations across the six cities were 18 $\mu g/m^3$ in the first period and 14.8$\mu g/m^3$ in the follow up period) were associated with a statistically significant reduction in mortality risk for deaths due to cardiovascular and respiratory causes, but not for lung cancer. This is equivalent to a relative risk of 1.27 for reduced mortality risk, suggesting a larger effect than in the cross sectional analysis. The study strongly suggests that reduction in fine PM pollution yields positive health benefits; however, $PM_{2.5}$ concentrations for the more recent years were estimated from visibility data, which introduces uncertainty in the interpretation of the results of the study.

The Adventist Health and Smog (AHSMOG) study followed cancer incidence and mortality for six years in a group of 6,338 nonsmoking California Seventh-day Adventists, from 1977 to 1987. In 1999, researchers updated the study to follow the group through 1992. In the original analysis, levels of inhalable particles (PM_{10}) were estimated. In the update, data from pollution monitors were available. Among men, increased particle exposure was associated with a rise in lung cancer deaths of 138 percent and in men and among women exposure was associated with increased mortality from non-malignant respiratory disease of 12 percent (Abbey et al., 1999). In 2005, 3239 nonsmoking non-Hispanic white adults who had been followed for 22 years were examined. Monitoring data was available for both PM_{10} and $PM_{2.5}$. As levels of $PM_{2.5}$ rose, the risk of death from cardiopulmonary disease increased by 42 percent (Chen et al. 2005).

The relative risk estimates from the major North American cohort mortality studies are summarized in Figure 3.

A new study involving selected California participants in the first CPS study indicated an association between $PM_{2.5}$ and all-cause death in the first time period of the study (1973-1982) but no significant association in the later time period (1983-2002) when $PM_{2.5}$ levels had declined in the most polluted counties. It is noted that the study's use of average $PM_{2.5}$ values for California counties as the exposure indicator likely leads to exposure error as California counties are large and quite topographically variable (Enstrom et al., 2005).

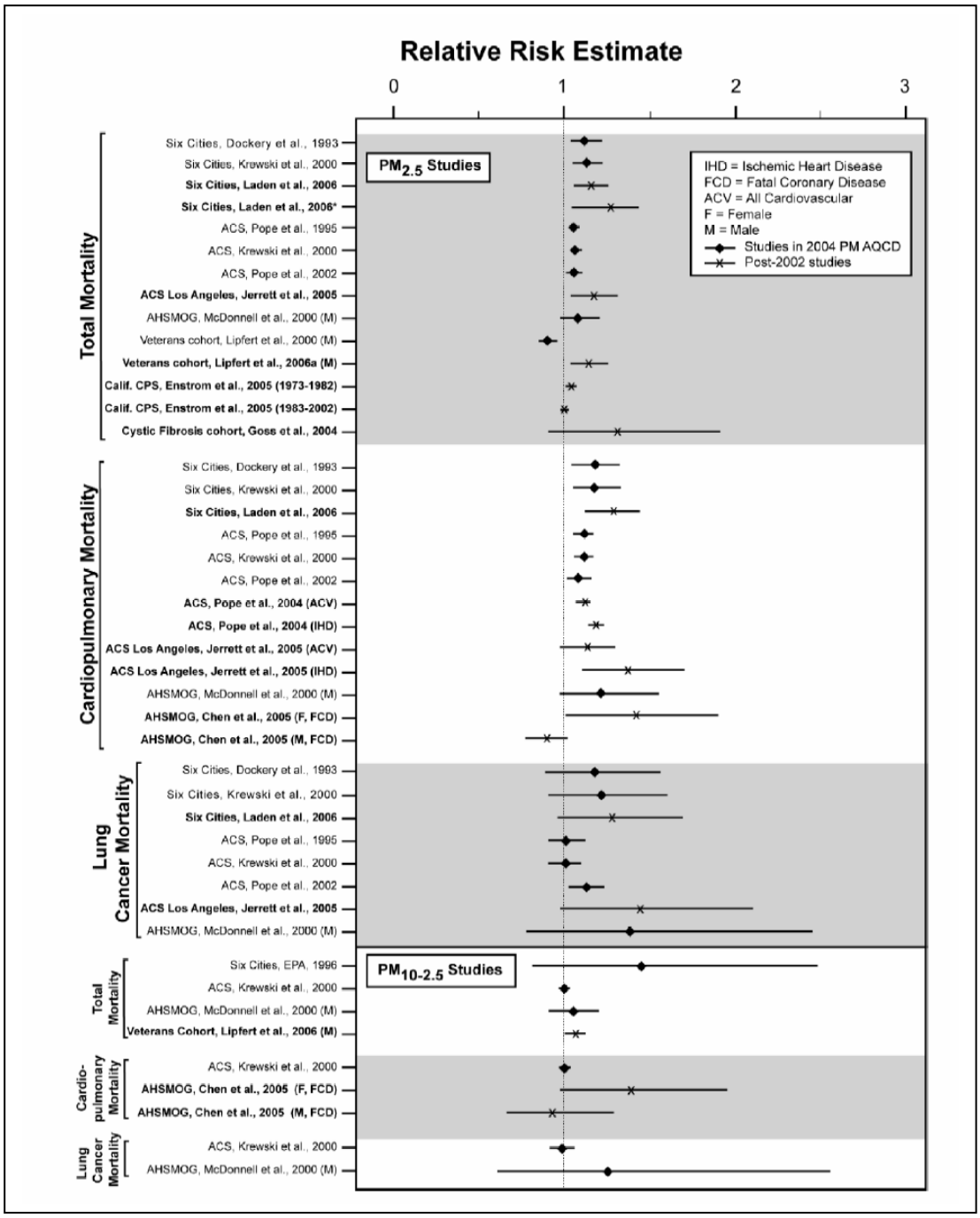

Figure 3. Relative risk estimates (and 95% confidence intervals) for associations between long-term exposure to PM (per 10 $PM_{10-2.5}$) and mortality. *Note the second result presented for Laden et al. (2006) is for the intervention study results. Source: US EPA (2006)

The EPRI-Washington University Veterans' Cohort Mortality Study used a prospective cohort of up to 70 000 middle-aged men (51 ±12 years) assembled by the Veterans Administration several decades ago. No consistent effects of PM on mortality were found. However, statistical models included up to 230 terms and the effects of active smoking on mortality in this cohort were clearly smaller than in other studies, calling into question the modelling approach. Also, only data on total mortality were reported, precluding conclusions with respect to cause-specific deaths. A recent analysis of the Veteran's cohort data reported a larger risk estimate for total mortality related to $PM_{2.5}$ in single pollutant models than reported in the previous analysis. There was a strong relationship between mortality and long term exposure to traffic (traffic density based on traffic flow rate data and road segment length) than with $PM_{2.5}$ mass. In multi-pollutant models including traffic density, the association with $PM_{2.5}$ was not statistically significant (Lipfert et al., 2006).

A positive but not statistically significant association was reported in a cohort of persons in the US. with cystic fibrosis cohort that focused primarily on evidence of exacerbation of respiratory symptoms. The power of the study to detect association was limited as only 200 deaths had occurred in the cohort of over 11,000 people. The mean $PM_{2.5}$ concentration was 13.7 $\mu g/m^3$ (Goss et al., 2004).

Further evidence to support an association between long-term air pollution exposure and fatal cardiovascular disease comes from recent cohort studies conducted in Sweden (Rosenlund et al., 2006) and Germany (Gehring et al., 2006). These European studies support US studies and increase confidence in the global applicability of the observations.

Mortality and short term exposure studies

Daily time series studies examine variations in day-to-day mortality counts in relation to ambient PM concentration measured by air quality monitoring networks. In general, the evidence from daily time series studies shows that elevated PM exposure of a few days is associated with a small increased risk of mortality. Large multi-city studies in Europe (APHEA2 (Air Pollution and Health: A European Approach 2), and the US (NMMAPS based on the largest 90 US cities) indicate that the increase in daily all-cause mortality risk is small but consistent. Concern over the statistical software used in the original analyses prompted a re-analysis of the NMMAPS and APHEA data, along with some other key studies, that was organized by the Health Effects Institute (HEI).

The NMMAPS estimate, based on the largest 90 cities was revised downward from 0.51% to 0.21% per 10 $\mu g/m^3$ PM_{10} (95% CI, 0.09 – 0.33) and from 0.51% to 0.31% for cardiorespiratory mortality. The APHEA mortality data reanalysis revealed that European results were more robust to the method of analysis. The WHO meta-analysis estimate (21 of 33 estimates from APHEA2) was 0.6% per 10 $\mu g/m^3$ (95% CI, 0.4-0.8) for daily all cause mortality and 0.9% for cardiovascular mortality. For PM_{10} and $PM_{2.5}$ the effect estimates are larger for cardiovascular and respiratory causes than for all-cause mortality. The higher European estimates may be due to differences in analytic approaches and other aspects of the methodology as well as the possibility of a difference in the true effect of PM arising from differing pollution or population characteristics or exposure patterns in the two continents. Figure 4 shows pooled estimates of the relative risks of mortality for a 10 $\mu g/m^3$ increase in various pollutants for all cause and cause-specific mortality from the meta-analysis of European studies (WHO, 2004).

A review of time series studies conducted in Asia also indicates that short-term exposure to air pollution is associated with increases in daily mortality and morbidity (HEI, 2004).

Morbidity

Evidence of associations between exposures and morbidity is complimentary to the information on mortality as it covers a broad range of adverse health effects from changes in biomarkers to clinical disease. Numerous studies have measured the short-term effects of air pollution on morbidity, using clinical indicators such as hospital admissions, counts of emergency room or clinic visits, symptom status, pulmonary function and various biomarkers. These studies have include multi-city time series studies (APHEA-2 hospital admission study; NMMAPS), panel studies of volunteers (PEACE- Pollution Effects on Asthmatic Children in Europe) which have provided data on acute effects on respiratory and cardiovascular systems, and objective measures of lung or cardiac function on a daily or weekly basis, and cross-sectional studies. The case-crossover design has been used to measure risk for acute events, such as myocardial infarction and stroke. In this design, the individual is the unit of analysis and exposures are compared in the "case" period during which the event of interest took place and in one or more "control" periods.

Figure 5 provides a summary of risk estimates for hospital admission and emergency department visits for cardiovascular and respiratory diseases from US and Canadian studies including aggregate results from one multi-city study. There is consistent evidence of increased risk for hospitalization and emergency room admissions for cardiovascular and respiratory diseases. Recent studies, including a new multi-city study of 11.5 million people in 204 US counties provide further evidence of increased risk for cardiovascular and respiratory disease hospitalization related to short term $PM_{2.5}$ exposure in individuals over 65 years (Dominici et al. 2006). A number of recent Canadian studies show significant associations between respiratory hospitalization and acute exposure to $PM_{10-2.5}$. For example, studies in Vancouver show increased risk of hospitalization for respiratory illness among children under 3, and for COPD and respiratory in the elderly. Studies in Toronto found an increased risk of hospitalization for asthma in children and associations with respiratory illness in the elderly.

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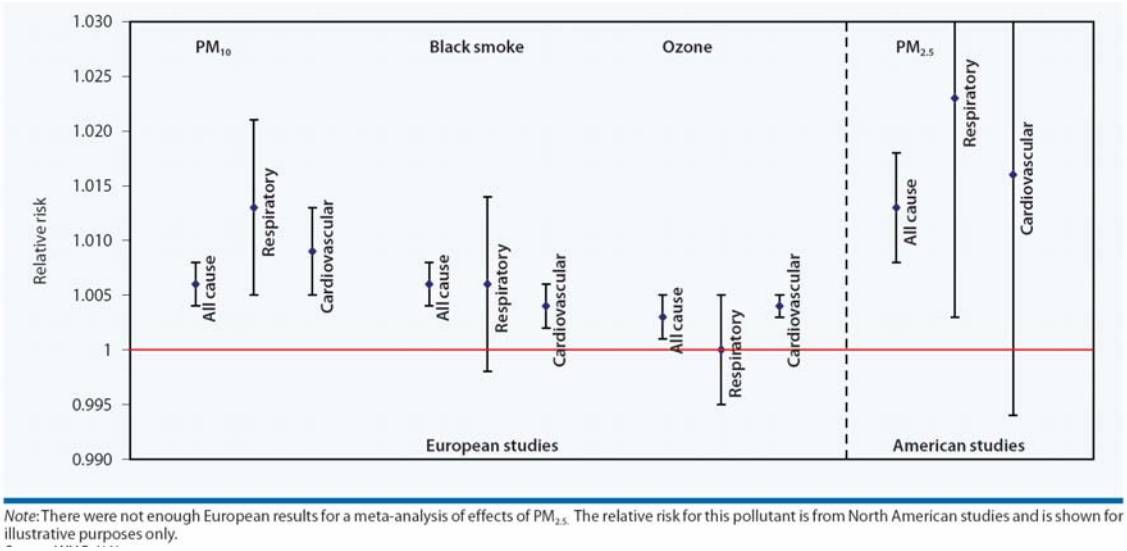

Figure 4. Pooled estimates of relative risks of mortality for a 10ug/m^3 increase in pollutant from Meta-analysis of European time series studies. Source: WHO (2006)

Public Health Burden of Mortality

Time series and cohort studies indicate that both short-term and long-term exposures to particulate matter can lead to increased mortality. It is important for public health planning to understand the amount of life-shortening that is attributable to those premature deaths. Researchers have investigated the possibility that short-term exposures may primarily affect frail individuals with pre-existing heart and lung diseases. Studies by Schwartz (2000), Zanobetti et al. (2000a), Zanobetti et al. (2000b); Fung et al. (2003); reanalysis by Zanobetti and Schwartz (2003); Zeger et al's analysis (1999) ; reanalysis by Dominici et al. (2003a, 2003b) all indicate that that the so-called "harvesting" hypothesis cannot fully explain the excess mortality associated with short term exposures to particulate air pollution. These studies suggest that any advance of the timing of death by PM is more than just a few days. Brunekreef (1997) estimated a difference in overall life expectancy of 1.11 years between exposed and clean air cohorts of Dutch men at age 25 using risk estimates from the Dockery et al (1993) and Pope et al. (1995) cohort studies and life table methods. Similar calculation for US white males yielded a larger estimated reduction of 1.31 years at age 25 (US EPA, 2004) These calculations are referred to as "back of the envelope" informal estimates that provide some insight into the potential life-shortening associated with ambient PM exposures.

Susceptible Populations

The question of who is most at risk for PM health effects depends on the level and length of exposure, as well as individual susceptibility. For acute or short-term exposures to moderately elevated PM concentrations persons with COPD, influenza, and asthma, especially among the elderly or very young are most likely to be susceptible. Although there may be broad susceptibility to long-term repeated exposure, the cumulative effects are most likely to be observed in older age groups with longer exposures and higher baseline risks of mortality (Pope and Dockery, 2006). Recent work suggests that effects on life expectancy are not uniformly distributed but depend on factors such as educational attainment and socio-economic status (Krewski et al. 2000) suggesting that life expectancy could be reduced among disadvantaged population groups (Brunekreef, 2002).

Toxicity of PM Components

The question of which air pollutants, sources, or combinations of pollutants are most responsible for health effects is still unresolved. The literature provides little evidence of a single source or well-defined combination of sources most responsible for health effects. With respect to particle size, the epidemiological, physiological and toxicological evidence suggests that fine particles ($PM_{2.5}$) play the largest role in affecting human health. These particles are generated by combustion processes and can be breathed deeply into the lungs. They are relatively complex mixtures including sulfates, nitrates, acids, metals and carbon particles with various chemical adsorbed onto their surfaces. The roles of coarse particles and ultrafine particles are yet to be fully resolved as are the roles of atmospheric secondary inorganic PM. Other characteristics of PM pollution that are likely related to relative toxicity include solubility, metal content and surface area and reactivity.

Role of Gaseous Co-pollutants

A major methodological issue affecting epidemiology studies of both short-term and long-term exposure effects relates to the use of appropriate methods for evaluating the extent to which gaseous co-pollutants (e.g. O_3, NO_2, SO_2, CO), air toxics, and/or bioaerosols may confound or modify PM-related effects estimates (US EPA, 2004). Gaseous co-pollutants are candidates for confounders because all are known to cause at least some adverse health effects that are also associated with particles. In addition, gaseous pollutants may be emitted from common sources and dispersed by common meteorological factors. For example, both CO and particles are emitted from motor vehicles, SO_2 and $PM_{2.5}$ are both emitted from coal-fired power plants. Krewski et al. (2000) found significant associations for both PM and SO_2 in their reanalysis for the Health Effects Institute of the Pope et al. (1995) study. Numerous new short-term PM exposure studies not only continue to report significant associations between various PM indices and mortality, but also between gaseous pollutants and mortality. In some cities the estimated PM effect is relatively stable when the co-pollutant is included in the model, whereas the estimated PM effect in other cities changes substantially when certain co-pollutants are included. Despite continuing uncertainties, the evidence overall tends to substantiate that PM effects are at least partly due to ambient PM acting alone or in the presence of other covarying gaseous pollutants (US EPA, 2004).

2.4 New Insights

The body of epidemiological, toxicological and clinical evidence on health effects has strengthened considerably over the past few years. A number of areas of advancement in the understanding of PM health effects have emerged (Chow, 2006). While new studies provide important insights, in general they support previous evidence regarding health effects of air pollution exposures (USEPA, 2006).

Cardiovascular Effects: While earlier research focused on the respiratory effects of PM exposure, evidence on cardiovascular outcomes has grown rapidly since 2000. A scientific statement published by the American Heart Association in 2004 indicated concern that the association of airborne particles with adverse cardiovascular outcomes is causal (Brook et al., 2004). Recent epidemiological, clinical and toxicologic studies report new evidence linking long-term exposure to fine particles with the development of atherosclerosis. A meta-analysis of cardiovascular hospitalization studies in Europe and the US consistently shows an increase in relative risk of cardiovascular hospitalizations associated with increments of 10ug/m^3 and 20 ug/m^3 $PM_{2.5}$. (Pope and Dockery, 2006). Numerous new studies have reported associations between ambient $PM_{2.5}$ and subtle cardiovascular effects such as changes in cardiac rhythm or heart rate variability (EPA, 2006). An extended follow up of the Harvard Six Cities adult cohort study found that cardiovascular (and lung cancer) mortality was associated with $PM_{2.5}$ exposure (Laden et al. 2006).

Mechanisms of Effect: Substantial progress has been made in understanding the biological and chemical mechanisms and pathways by which PM causes adverse effects on human health. Recent research has increased confidence that PM-cardiopulmonary health effects observed in epidemiologic studies are "biologically plausible". Figure 6 indicates the various hypothetical pathways of effect that have been explored. Much remains to be learned; however, it appears from human and animal experimental studies that multiple pathways linking exposure to cardiompulmonary health effects are involved with complex interactions and interdependencies. While the evidence is still evolving and is not yet definitive, there is some evidence to suggest that PM exposure is associated with increased heart rate and reductions in heart rate variability suggesting adverse effects on cardiac autonomic function. Other studies have observed, but not consistently, pulmonary or systemic inflammation and related markers of cardiovascular risk such as cardiac arrhythmia, blood pressure changes, arterial vasoconstriction, ST-segment depression, and changes in cardio repolarization (Pope and Dockery, 2006). It is hypothesized that low to moderate grade inflammation induced by long-term chronic PM exposure may initiate and accelerate atherosclerosis. Short-term elevated exposures and related inflammation may increase the risk of making atherosclerotic plaques more vulnerable to rupture, clotting and eventually causing heart attack or stroke.

Exacerbation of existing pulmonary disease, oxidative stress and inflammation, changes in cardiac autonomic functions, vasculature alterations, translocation of PM across internal biological barriers, reduced defense mechanisms and lung damage have all been related to different levels of PM exposure, as well as to different particle sizes and compositions.

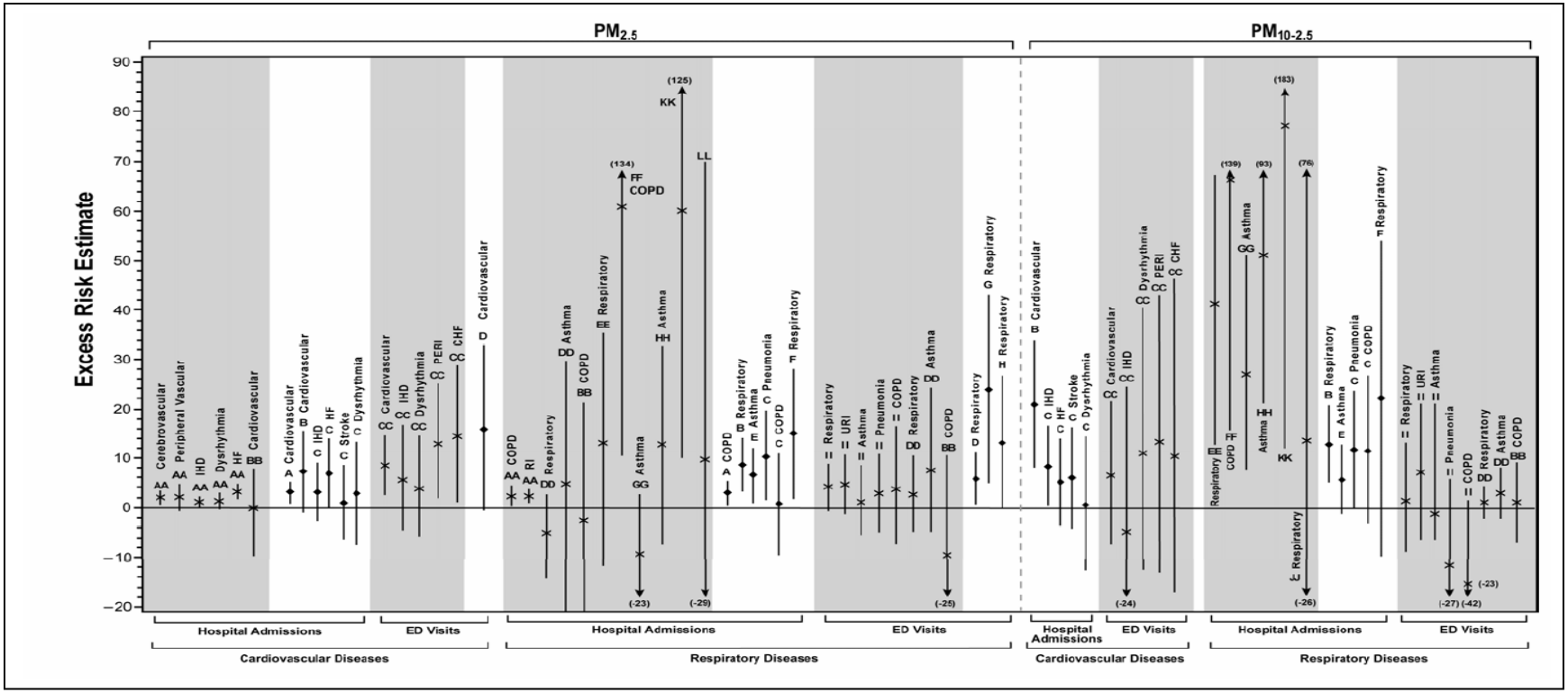

Figure 5. Excess risk estimates for hospital admissions and emergency department visits for cardiovascular and respiratory diseases in single-pollutant models for U.S. and Canadian studies, including aggregate results from a multicity study (denoted in bold print below). PM increment used for standardization was 25 μg/m^3 for both $PM_{2.5}$ and $PM_{10-2.5}$. Results presented in the 2004 PM AQCD are marked as ♦, in the figure (studies A through H). Results from recent studies are shaded in grey and marked as × in the figure (studies AA through JJ). (CHF = congestive heart failure; COPD = chronic obstructive pulmonary disease; HF = heart failure; IHD = ischemic heart disease; PERI = peripheral vascular and cerebrovascular disease; RI = respiratory infection; URI = upper respiratory infection). Source: USEPA (2006)

A. Moolgavkar (2003), Los Angeles
B. Burnett et al. (1997), Toronto
C. Ito (2003), Detroit
D. Stieb et al. (2000), St. John
E. Sheppard (2003), Seattle
F. Thurston et al. (1994), Toronto
G. Delfino et al. (1997), Montreal
H. Delfino et al. (1998), Montreal
AA. Dominici et al. (2006), 204 U.S. counties (age >65 yr)
BB. Slaughter et al. (2005), Spokane (age 15+ yr)
CC. Metzger et al. (2004), Atlanta
DD. Slaughter et al. (2005), Atlanta
EE. Chen et al. (2005), Vancouver, Canada (age 65+ yr)
FF. Chen et al. (2004), Vancouver, Canada (age 65+ yr)
GG. Lin et al. (2002), Toronto, Canada (age 6-12 yr, boys)
HH. Lin et al. (2002), Toronto, Canada (age 6-12 yr, girls)
II. Peel et al. (2005), Atlanta
JJ. Yang et al. (2004), Vancouver, Canada (age >3 yr)
KK. Lin et al. (2005), Toronto, Canada (age <16 yr, boys)
LL. Lin et al. (2005), Toronto, Canada (age <16 yr, boys)

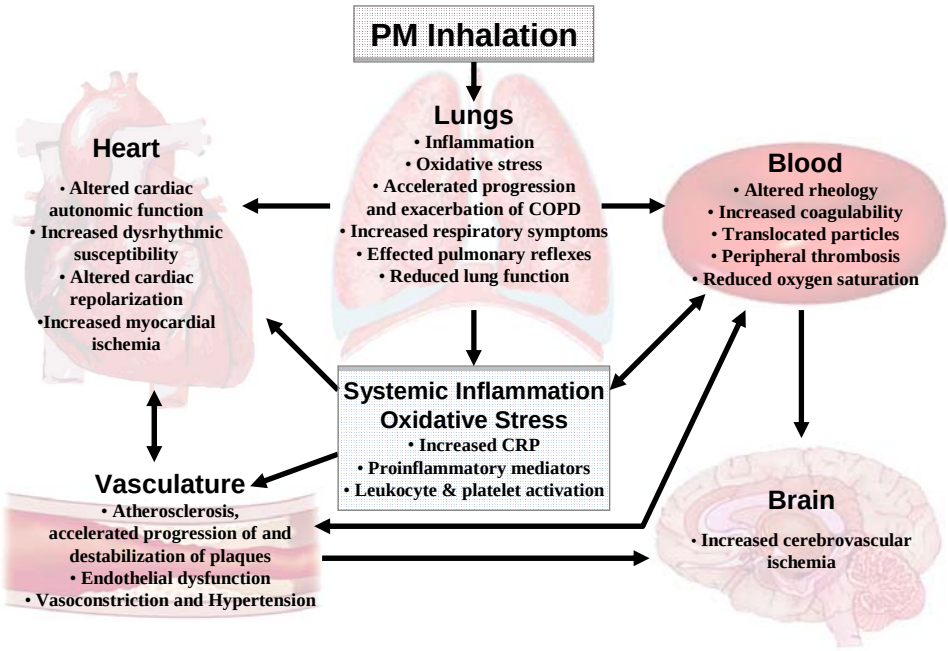

Figure 6: Hypothesized pathophysiological pathways linking PM exposure with cardiopulmonary morbidity and mortality. Source: Pope and Dockery (2006)

Local Level Mortality Risk: A new analysis of ACS data focused on neighbourhood to neighbourhood differences in urban air pollution in Los Angeles using more precise exposure assessment methods found death rates from all causes and cardiopulmonary diseases at least two times higher than previously reported in analyses of the ACS cohort (Jerrett et al., 2005). The highest estimated from original ACS study (Pope et al., 2002) for all cause mortality was 6 percent. Taking into account neighbourhood confounders, the risk was about 11 percent. The annual average level of $PM_{2.5}$ in the most contaminated area was about 24 ug/m^3.

Risks to Diabetics: There is growing evidence to suggest that people with diabetes are more sensitive to cardiovascular effects from air pollution (Jerrett et al., 2005, O'Neill et al., 2005, Zanobetti et al., 2001; Goldberg et al., 2001). Goldberg et al. (2006) reported significant associations between $PM_{2.5}$ and diabetes deaths, as well as total mortality in people with previous diagnoses of diabetes. The acute risk for cardiovascular events in patients with diabetes mellitus may be two-fold higher than for non-diabetics. A study of Boston-area residents found that blood vessel reactivity was impaired in people with diabetes on days when concentrations of particles from traffic and coal-burning power plants were elevated (O'Neill, 2005). These findings are of particular concern given the increasing incidence of diabetes in North America. A recent study has indicated mechanistic evidence for diabetes-related susceptibility (Proctor et al., 2006).

Risks to Children: There is substantial evidence to indicate that PM exposure in children is associated with adverse effects on lung function, aggravation of asthma, increased incidence of cough and bronchitis. In addition, there is evidence to suggest an increase risk of postneonatal respiratory mortality as concentrations of $PM_{2.5}$ risk by $\mu g/m^3$. Studies on birth weight, preterm births and intrauterine growth retardation also suggest a link with air pollution, but these studies are not sufficient to draw conclusions about causality (WHO, 2005).

Traffic Exposures: Recent evidence has shown that exposures of people living near busy roads are insufficiently characterized by air pollution measurements obtained from urban background locations (Finkelstein et al., 2004; Jerrett, 2005; Brunekreef et al., 2003) . In some cities, a significant part of the urban population may be affected by roadway sources. In some urban areas, elevated exposures may particularly affect socially disadvantaged groups (Finkelstein et al., 2003; Finkelstein et al., 2005; Gunier et al., 2003). A new analysis of the Veterans cohort data reported a stronger relationship between mortality with long-term exposure to traffic than with $PM_{2.5}$ mass (Lipfert et al. 2006).

Thoracic coarse particles: While the 2004 US EPA Air quality criteria document concluded that there was insufficient evidence of an association between long-term exposure to thoracic particles ($PM_{10-2.5}$) and mortality the ASHMOG (Chen et al., 2005) and Veterans cohort study (Lipfert, 2006) provide limited suggestive evidence for associations between long-term exposure to $PM_{10-2.5}$ and mortality in areas with mean concentrations from 16 to 25 ug/m^3. The extended analyses of the Six Cities and ACS cohort studies did not evaluate linkages between health effects and exposure to $PM_{10-2.5}$. Recent epidemiologic studies strengthen the evidence for health effects associated with acute exposure to thoracic coarse particles. Several Canadian studies studies report respiratory morbidity in cities with low $PM_{10-2.5}$ concentrations. Many studies do not show statistically significant associations with mortality with the exception of a recent study showing a link with cardiovascular mortality in Vancouver. New toxicology studies have demonstrated inflammation and other health endpoints as a result of exposure to thoracic coarse particles. Clinical exposure studies show changes in heart rate and heart rate variability measures among exposed healthy and asthmatic adults. It appears that the observed responses may be linked to endotoxins and metals.

2.5 Conclusions

1. Expanded analyses of ongoing cohort studies continue to provide evidence of associations between of long term exposures to fine particles and mortality (10 µg/m^3 $PM_{2.5}$ is associated with approximately a 6 to 17% increase in relative risk of mortality with some outliers). Mixed results have been seen in the AHSMOG study and VA cohort study and 11 California ACS study. Across the range of particulate air pollution observed in recent studies, the concentration response relationship can reasonably be modeled as linear with no threshold.

2. Previous cohort studies may have underestimated the magnitude of mortality risks. PM mortality effects estimates tend to be larger when exposure estimates are based on more focused spatial resolution and/or when local sources of exposure, especially traffic sources, are considered.

3. The available evidence suggests a small increased lung cancer risk due to combustion-related ambient PM air pollution. The extended follow-up of the ACS and Harvard Six Cities cohort studies both observed PM lung cancer associations which were statistically significant in the ACS study. Outdoor air pollution typically includes combustion-generated respiratory carcinogens.

4. Multi-city time series studies in North America and a meta-analyses of European time series studies support single-city study evidence of an adverse effect of daily PM_{10} exposures on short term mortality at current concentrations (a 10 µg/m^3 $PM_{2.5}$ or 20 µg/m^3 PM_{10} increase is associated with a 0.4% to 1.5% increased in relative risk of mortality).

5. While earlier studies focused on evidence of respiratory effects, studies emerging over last 10 years have found a link between both short term and long term exposure to particulate matter and risk of cardiovascular disease and death.

6. Understanding the shape of the concentration-response function and the existence of a no-effects threshold level has played a key role in setting air quality standards. Recent empirical evidence concerning the shape of the PM concentration-response function is not consistent with a well-defined no effect threshold.

7. With respect to acute or short term exposures to moderately elevated PM concentrations, persons with chronic cardiopulmonary disease, influenza, and asthma, especially the elderly or very young, are most susceptible. A number of indicators of susceptibility have been identified including pre-existing respiratory or cardiovascular disease, diabetes, socioeconomic status and educational attainment.

8. PM exposure impacts the health of children including deficits in lung function and lung function growth, increased respiratory illness and symptoms, increased school absences, and hospitalizations for respiratory disease. Several recent reviews generally conclude that PM exposure is most strongly and consistently associated with postneonatal respiratory mortality with less compelling evidence of a link between PM and SIDS, fetal growth, premature birth and related birth outcomes.

9. Recent research has increased confidence that cardiopulmonary health effects observed in epidemiologic studies are biologically plausible. While a single definitive mechanism was not been identified four interrelated pathways involving i) accelerated progression and exacerbation of COPD, ii) pulmonary/systemic oxidative stress, inflammation leading to accelerated atherosclerosis; iii) altered cardiac autonomic function; and iv) vasculature alterations have been hypothesized.

10. There is little evidence of a single major component of PM or a single source or combination of sources that are most responsible for observed health effects, however epidemiological, physiological and toxicological evidence suggests that fine particles play a substantial role in affecting human health. The roles of coarse particles and ultrafine particles are yet to be fully resolved, as are the roles of atmospheric secondary inorganic PM. Other characteristics of PM pollution that are likely related to relative toxicity include solubility, metal content and surface area and reactivity.

11. Despite continuing uncertainties, the evidence overall tends to substantiate that PM effects are at least partly due to ambient PM acting alone or in the presence of other covarying gaseous pollutants.

2.6 Issues for Risk Management

The lack of evidence of a threshold concentration for health effects suggests that continued reductions in ambient pollutant levels will result in public health benefits. It further suggests that the target for reduction should be the background concentration. In view of the potentially large costs associated with further abatement measures to achieve cleaner air, questions arise concerning tradeoffs between expenditures on air quality management and other measures to achieve public health benefits. In air pollution hot spots and areas where standards have been achieved, air quality risk management becomes more complex. Chapter 5 examines these issues and describes innovative air quality risk management approaches intended to compliment the traditional regulatory approach focused on the attainment of national ambient air quality standards.

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CHAPTER 3. Emission Inventories, Air Quality Measurements and Modeling: Guidance on Their Use for Air Quality Risk Management

Jeffrey R. Brook (measurements and chapter lead author), William Pennell (emissions) and Michael D. Moran (modeling)

KEY MESSAGES

- Accurate emission inventories provide essential information to understand the effects of air pollutants on human and ecosystem health, to identify which sources need to be controlled in order to protect health and the environment, and to determine whether or not actions taken to reduce emissions have been effective.
- In principle, the development of emission inventories would seem to be a relatively straightforward process, but in practice their production is found to be a very complex and demanding task. As initial actions to reduce emissions from large point sources find success, understanding and addressing residual AQ problems requires greater effort and emission inventories of increasing sophistication.
- AQ measurements are essential for public health protection. Measurements are the basis for determining the current level of population health risk and consequently for prioritizing the need for reductions. They are used to develop exposure estimates for epidemiological studies aimed at quantifying concentration-response functions, which are at the heart of cost-benefit analyses. Measurements are critical for evaluating the effectiveness of AQ management strategies and altering such strategies if the desired outcomes are not being achieved.
- Considerable knowledge exists today to develop efficient AQ management measurement programs, in conjunction with emissions data and air quality models, that do not involve extensive numbers of sites and yet provide a large amount of information to support multiple objectives.
- AQ measurement, emissions, and modeling can each provide useful information for AQ risk management, and when they are considered together they can provide additional insights and guidance. However, AQ models depend upon the availability of information about measurements, emissions, and meteorology whereas the converse is not true. AQ modeling should thus follow and not precede the development of measurement and emissions information.
- AQ models quantify the links between emissions of primary pollutants or precursors of secondary pollutants and ambient pollutant concentrations and other physiologically, environmentally, and optically important properties. They are also the only AQ tool available that can predict *future* air concentration and deposition patterns based on possible future emission levels. AQ models can account for the impacts of nonlinear processes and are able to predict whether a candidate abatement strategy will lead to benefits or disbenefits or both. They are naturally amenable to a "one-atmosphere" approach and can address multiple pollutants simultaneously and quantify possible co-benefits.
- AQ models have been applied both directly or indirectly to support AQ management and policy formulation. These applications include: the evaluation of impact of emissions changes, including proposed control measures; source apportionment and source attribution; input to conceptual model formulation and development; emission inventory evaluation; measurement network and field experiment design; AQ forecasting; and testing science.
- There are a large number of possible sources of AQ model error and uncertainty. AQ model uncertainty is impossible to quantify, but it is possible to characterize through model performance evaluations, model comparisons, and sensitivity and bounding tests. There are also varying degrees of uncertainties across pollutants and their components
- AQ modeling uncertainty can be managed and limited by following "best practice" at all stages of the modeling process. Best practice requires thoughtful and careful selection, set up, and application of a credible AQ modeling system accompanied by careful scrutiny and consistency checking of the results by various means, including measurements and results from both alternate configurations of the selected AQ model and from other AQ models.
- The AQ modeling process is demanding in terms of requirements for input data, skilled modeling personnel, calendar time, and computer resources. AQ modeling should be viewed as a significant undertaking. Such programs must be resourced appropriately, including a long-term commitment, in order to obtain credible information useful to AQ risk management.

3.1 Introduction

Emission inventories, air quality measurements and air quality modeling provide the scientific foundation for air quality risk management. To comprehensively assess air quality management options, reliable information is required on air pollutant emissions (including GHGs) and ambient concentrations, as well as air quality models, that realistically and quantitatively link emissions and observations (including visibility and climate). This chapter provides an overview of these basic tools for air quality management.

Figure 3.1.1 depicts the roles of emission inventories, measurements, modeling, and analysis and interpretation (e.g., source apportionment), in understanding how chemistry, meteorology and natural and human emissions interact to produce observed levels of outdoor air pollution. Conceptual and numerical models provide the capability to predict future concentrations resulting from emissions changes, as well as atmospheric and global change.

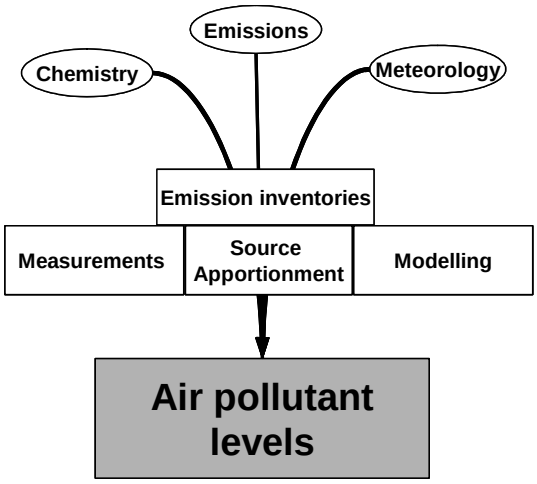

Figure 3.1.1 Emission inventories, ambient measurements and air quality models are the tools needed to understand the current air pollutant levels and predict future levels under various policy options.

Several reviews on air pollutant emissions, ambient measurements and air quality modeling have been published in the past. The NARSTO particulate matter assessment for policy makers (McMurry, Shephard, and Vickery, 2004 or NARSTO, 2004) describes measurement methods for particulate and gaseous pollutants, as well as their emissions and observations in North America. Receptor-based methods of data analysis and interpretation and the status of source-based modeling are also discussed in this assessment. This chapter highlights and extends key findings from the NARSTO assessment to provide readers with an understanding of the role of emissions, measurements and models in air quality risk management, best practices in applying these tools, and issues that arise in their use.

3.1.1 Role of emissions, measurements and models in air quality risk management

To understand the role of emissions, measurement and modeling in air quality (AQ) management, the entire science-policy process needs to be considered. Figure 3.1.2 shows the basic steps of AQ risk management and specifies how science inputs from emissions, measurement and modeling "tools" play a direct role in the policy process. Emissions, measurement and modeling enables the prediction of air quality improvements associated with emission reduction options, as well as the analysis of the costs and benefits of air quality management options. Figure 3.1.2 depicts the process in a linear, sequential fashion, however the order of steps may be reversed or steps may occur in parallel. For example, in developing a conceptual model of the sources and atmospheric processes that lead to current ambient pollutant concentrations, there may be a need to gather additional measurements to test and refine the model before one can thoroughly evaluate whether or not the tools are reliable. In addition, depending upon the maturity of air quality risk management in a particular location, not all steps may be required. Existing measurement programs may be fully adequate or the AQ models may have already been widely accepted for the intended use.

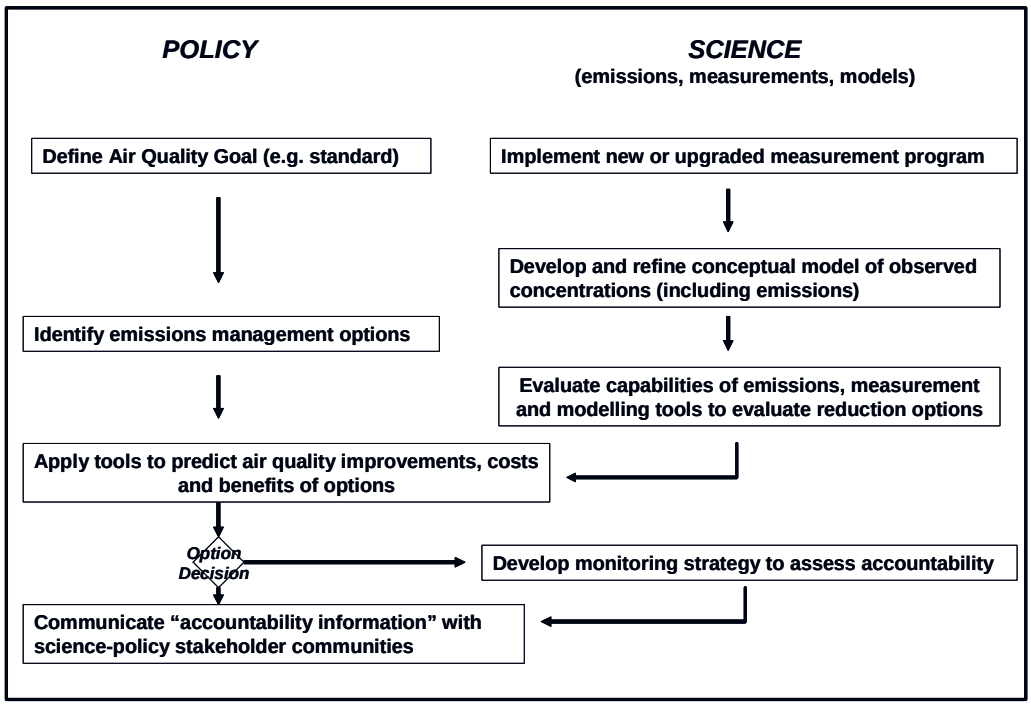

Figure 3.1.2. The role of emissions, measurement and modeling in local/regional air quality risk management.

A crucial step in 'classical' AQ management is to quantitatively link ambient pollutant concentrations at specific locations or within specific geographic regions to specific emissions (emissions to concentration relationship). This linkage is studied through both receptor and source-based AQ models. Source-based models, which are discussed later in this chapter, are capable of predicting future ambient air quality concentrations and are applied to evaluate emission reduction scenarios in the context of Figure 3.1.1.

Model estimates of concentration changes can then be integrated with concentration-response functions (CRFs) to estimate health benefits. Derivation of CRFs depend upon the availability of AQ measurements. Thus, emissions, measurements and models are foundation of AQ risk management and are even more critical for quantitative cost-benefit analysis. Table 3.1.1 summarizes the various ways in which these tools are applied in AQ risk management.

Emissions, measurements and modeling also play an indirect role in AQ management through the provision of information to the general public or specific stakeholder groups. For example, the North American Commission for Environmental Cooperation or CEC publish reports ranking major sources and reports on progress (http://www.cec.org/takingstock/index.cfm). *Right-to-know* websites such as the Toxic Release Inventory in the U.S. (http://www.epa.gov/tri/) and the Canadian National Pollutant Release Inventory (http://www.ec.gc.ca/pdb/npri/npri_home_e.cfm) provide specific emissions information for local areas. Public access to emissions information is increasing worldwide (e.g., Mexico: http://app1.semarnat.gob.mx/retc/index.php and http://www.epa.gov/ttn/chief/net/mexico.html) and international standards for a Pollutant Release and Transfer Register (PRTR) have been established (http://www.epa.gov/tri/programs/prtrs.htm). Media reports conveying current AQ concentrations through the air quality index (AQI) or maps (e.g., http://airnow.gov/), as well as AQ forecasts and/or smog advisories may mobilize NGOs and influence public opinion, thereby creating political pressure to enact more stringent clean air regulations.

This chapter elaborates on the functions of emissions, measurements and modeling in AQ risk management. Section 3.2 provides the basics of contemporary emissions inventory development, evaluation, and dissemination based on a recent emission inventories assessment conducted by NARSTO. Section 3.3 describes the various areas of application of measurement data in air quality risk management and provides guidance on technical issues to be considered in establishing a robust measurement program. Section 3.4 describes the application of models for air quality risk management, identifies key technical issues to consider in using air quality modeling systems and reviews eight steps for best practice in using models in air quality management. Section 3.5 describes current efforts and future directions to combine the capabilities of emissions and measurement information and air quality modeling to better support air quality management. Section 3.6 provides the chapter conclusions and recommendations.

Table 3.1.1: The application of emissions data, air quality measurements and air quality modeling in air quality risk management

Tool	Area of Application in Air Quality Risk Management
Emissions	• Current emission rates for criteria gases and particles by source type and location • Projected emission rates for criteria gases and particles by source type and location and detailed information on the causes of the changes in emissions • Identification of broad based and detailed emission reduction strategies and/or technologies by source type and their effectiveness for the emissions of criteria gases and particles
Measurements	• Characterization of past and current pollutant levels and identification of exceedances of AQ standards, objectives, or targets • Time series of ambient concentrations at population-based monitoring sites for trend analysis in relation of emission reductions • The relationship between ambient concentrations at population-based monitoring sites and a range of health endpoints (concentration-response function) • The relationship between ambient concentrations of primary and secondary pollutant and emission source categories (source apportionment or receptor models) • Development and evaluation of conceptual models and source-oriented models
Models	• Simulation of emission scenarios and quantification of resulting benefits and disbenefits by prediction of ambient concentrations at multiple time and space scales by physically-based, source-oriented air quality models for: o Base case (e.g., current emissions) o Emission levels when policies currently "on-the-books" are fully implemented o New emission reduction scenarios • Estimation of emission changes required to attain AQ objectives or standards • Evaluation of emission estimates • Quantification of source-receptor relationships • Characterization of governing chemical regimes and limiting reactants for current and future conditions • Development of conceptual models • Simulation and design of new or modified measurement systems (network optimization, site selection, input to data assimilation and analysis routines)

3.2 Emissions Information for Air Quality Risk Management

3.2.1 Introduction

Emission inventories are the foundation of effective air quality management. Without accurate information on the sources of air pollutants – what they are, where they are located, what they are emitting, and how much – it is impossible to identify which sources are most important to control, to predict the air-quality consequences of these emissions, or to monitor the effectiveness of emission reduction programs. In this section of Chapter 3, the basics of contemporary emissions inventory development, evaluation, and dissemination are discussed. This includes:

1. The development of emission inventories.
2. Methods for evaluating them and quantifying their uncertainties.
3. Typical strengths and weaknesses of mature emission inventories.
4. Actions for addressing the weaknesses.
5. The costs of emission inventory development and improvement.

Most of the material in this section is drawn from a recent report: *Improving Emission Inventories for Effective Air Quality Management Across North America: A NARSTO Assessment* (NARSTO, 2005). Thus, the discussion may have a North American flavor. However, as the three countries represent both mature (Canada and the United States) and emerging (Mexico) emission inventory development programs, this discussion will likely be enlightening to air quality managers in other parts of the world.

3.2.2 Emission Inventory Development1

Emission inventories are usually developed using the following model

$$E = EF \cdot A \cdot (1\text{-}ER) \qquad (1)$$

Where E is the emission rate (e.g., kg/hr or tonnes/yr) of a given pollutant or pollutant precursor, EF and A are the emission factor and activity factors, respectively, and ER ($0 < ER < 1$) an emission reduction factor, which accounts for any emission control devices that may be applied to the source. The emission factor, EF, is the mass of a given pollutant or chemical species emitted per unit process variable. The activity factor, A, is the related process variable such as mass of fuel consumed, vehicle kilometers traveled, etc. in a given amount of time. In reality, emission and emission reduction factors can vary from source to source as well as with the value of the activity factor, type of fuel, operating conditions, age of the source, geographical location, time of year, and so forth. Not all of these complexities can be accurately represented in such a simple relationship, and more sophisticated emission models have been developed for very complex categories such as mobile source emissions (Miller et al., 2006).

Except in rare instances, emission factors or their equivalents are based on measurements. The easiest source class to characterize is large point sources, such as electric generating units or stack emissions from large industrial operations. Emissions from these facilities can be measured by direct sampling of flue gases – as long as reliable sensors and methods are available (they can be in situ or remote) and appropriate sampling techniques are used.[2] Using these techniques, gaseous emissions from large point sources, such as CO_2, SO_2, and NO_x, can be estimated to better than +/- 20% over time periods as short as one hour (NARSTO, 2005).

Emissions from more dispersed and numerous individual sources are much more difficult to characterize and evaluate. Examples include fugitive (i.e., inadvertent) emissions from industrial sources, natural emissions from vegetation, agricultural emissions, emissions from small industrial or commercial sources, residential sources (e.g., particulate matter emissions from cooking or space heating), and large-scale biomass burning. For these types of emission sources, direct measurements may be difficult or they may be feasible for only a small sample of the sources in question. All of these factors lead to emission estimates that are more uncertain than for large point sources. These uncertainties can range from a factor of two to complete neglect of an unknown source or chemical precursor that upon later analysis turns out to be significant.

[1] Additional information on emission inventory development can be found at http://www.epa.gov/ttn/chief/eiip/techreport/, which is the website for the U.S. Environmental Protection Agency's Emission Inventory Improvement Program.

[2] For example, when measuring particulate matter emissions it is necessary to mimic the cooling and flue (or exhaust) gas dilution processes that occur immediately after the emissions enter the atmosphere, as many "primary" particles are formed in this near-source region.

On-road and non-road mobile source (automobiles, trucks, aircraft, locomotives, construction equipment, ships, etc.) are a good example of an important, but widely dispersed and variable, component of pollutant emissions. Over the past 50 years, considerable effort and resources have been expended in several developed countries to develop procedures for estimating emissions from mobile sources. The traditional approach for estimating automobile and truck emissions has been to measure emissions from dynamometer tests of representative vehicles in the laboratory. The dynamometer tests are run to represent typical driving cycles, and the vehicle emissions are measured in real time. These measurements are used as input to complex mobile source emission models that attempt to simulate vehicle fleet operating conditions for a wide range of urban, suburban, and rural settings. The problems with dynamometer test are that number of sampled vehicles may be too small to represent a statistically valid sample, and they may not represent the range of fuels used, driving cycles or conditions, environmental factors, and states of repair of the actual vehicles in use. Consequently, field measurements using instrumented chase vehicles, road-side remote sensing of vehicle plumes, chemical sampling of the air in traffic tunnels, and other experimental set-ups are used to check and fine tune mobile source emission models. In the future low-cost portable emission measurement systems (PEMS) and on-board diagnostic sensors (OBDs) may allow cost-effective sampling of a much larger sample of in-service on-road and non-road vehicles under real-world operating conditions. These data could greatly improve the accuracy of mobile source emission estimates.

Because most sources are not equipped with continuous emission monitors to measure actual emissions, Equation 1 forms the basis of most data reported in inventories. Although the focus is often placed on the value of the emission or emission reduction factor, the activity factor is equally important. Activity factors can be developed from continuously monitored process data, but as with continuous emission monitors, these data are generally scarce. More frequently, activity factors are developed from economic activity data or activity surveys. Fuel consumption data are a good example of the use of economic activity data that are collected for reasons other than emissions, but can be used in the development of emission inventories. Data are usually available in the United States by type of fuel used, for various time periods (monthly or sometimes weekly), and for various geographical areas (counties or states). Information on construction activities can be used to develop emissions from off-road construction sources. Population densities coupled with activity surveys can provide inventory developers with information on emissions from residential woodstove, fireplace, and open burning. Land use maps and satellite data are useful for estimating the types and densities of vegetative cover, which in turn are used to estimate biogenic emissions. In each case, emission estimates can be developed using data that have been collected for other purposes, such as for tax estimation, economic development, or land-use planning.

Projections of future emissions also depend upon these data and estimates in their growth rates in future years. In the absence of significant technological change, past relationships between population growth and the types of activity factors noted above provide a good starting point for estimating future activity levels, and subsequently, future emission levels.

3.2.3 Evaluating Uncertainties in Emission Estimates

Uncertainty information is introduced into emission inventories in a number of ways. Emission factors do not usually account for variability in emissions due to changes in source operating conditions, or across the individual sources that make up a source category. New technologies can change processes and emissions, and such changes will not be reflected in emission factors that were based on the original process. Emission factors that are based on idealized operations, such as use of vehicle operating cycles, do not accurately capture actual operation and therefore actual emissions. Measurement biases or errors introduce additional uncertainties into the reported inventory data. These differences can be associated with the location, time, or composition of emissions, leading to uncertainties in the spatial, temporal, or chemical data used in air quality models.

Generally, emission uncertainties are lower when the models are used to estimate changes in air quality over longer periods of time and across geographical areas that are of the same order of magnitude as the spatial scale covered by the inventories. National annual average pollutant concentration estimates are likely to have a lower uncertainty than concentration estimates for a specific urban area over the course of a single day, because the differences between actual and estimated emissions based on emission factors are more likely to average out over a longer period of time and over a broader area.

As air quality models become more sophisticated, however, more detailed emissions information is needed. The ability to model atmospheric processes over the course of a single hour with more detailed chemical reaction mechanisms and in smaller areas means that the differences between the actual emissions within the modeled area and time and the

estimate based on an annual average emission factor may be significantly different. Such discrepancies can result in estimates of pollutant concentrations that do not reflect actual conditions. These differences can lead to misidentification of the most important sources within a given area or erroneous estimates of the specific emissions that need to be controlled for a given source type. Thus, the sources that are identified as needing control, and the types of controls that are installed, do not result in the desired changes in pollutant concentrations.

Characterizing the uncertainties in non point-source emission estimates is not a simple task. Two general approaches are usually taken – bottom-up and top-down. In the bottom-up approach, uncertainties (bias and random error) in the individual measurements or parameters that make up the emission model are estimated (e.g., from field measurements similar to those described in the previous section) and represented as a probability distribution function (pdf). These uncertainties are then propagated through the model to provide an estimate of the uncertainty in the emission estimate. A straightforward way of doing this is the Monte-Carlo approach in which random samples are taken from the individual input pdfs and the output emissions calculated. Repeating this a large number of times provides a pdf of the emission model estimate from which the uncertainty in the emission estimate can be estimated. An alternative bottom-up approach is sensitivity analysis. A simple form of sensitivity analysis is to evaluate the sensitivity of emission-model output to its various input values in terms of the partial derivative of the model output to the input parameter in question. This approach indicates the relative sensitivity of the emission model to its various inputs and enables a crude estimate of uncertainty by providing a measure of how much the emission estimate would change for a given change in an input parameter.

In top-down evaluations of emission inventories or emission models, ambient measurements or other independent data are used to evaluate the accuracy of the emission estimates. One method is to compare temporal trends in emission estimates with past trends or to compare trends in the ambient concentrations of a pollutant (or in ratios of pollutants) with the trend in estimated emissions under conditions in which the effects of transport, chemical transformation, and removal can be neglected or accounted for. Results from of this kind of analysis are reported in Parrish et al. (2002) and CRC (2004). A description of this approach is also provided in NARSTO (2005).

Other top-down methods for evaluating emission inventory uncertainties include using alternative methods for estimating emissions (such as comparing vehicular emission estimates based on vehicular distance traveled with those based on total fuel consumption), source apportionment techniques, and inversing modeling. Source apportionment (or receptor modeling) techniques use various multivariate statistical methods to infer source types, source location, and relative contribution from ambient measurements (Watson and Chow, 2005). These methods assume that the measured concentration of ambient species at a particular location is a linear combination of the emissions from all sources affecting that site. These techniques are most useful when there are significant differences in the chemical species emitted by contributing sources or if there are unique chemical signatures associated with them. These methods have been used to evaluate inventories of $PM_{2.5}$ and VOCs (McMurry, Shephard, and Vickery, 2004; Watson, Chow, and Fujita, 2001).

Inverse modeling involves reformulating source-based air quality models so that emission source strengths are expressed in terms of the observed concentrations. In other words, the model is used to deduce the temporal and spatial emissions that explain the observed concentration field. Because of the limited spatial resolution of air quality models, this technique is most commonly used to deduce area sources (Petron et al., 2002; Park et al., 2003; and Gilliand et al., 2003).

Miller et al. (2006) provide some useful guidance for using these methods in evaluating emission inventory uncertainty:

> "Top-down methods must be applied with caution to ensure that the comparisons are as appropriate as possible. Although the data used in top-down evaluations are (ideally) independently derived relative to the inventory data, top-down data are similarly subject to data uncertainty and limitations. Ambient data, for instance, include contributions from sources other than the source categories being evaluated, and care must be taken to verify that such contributions are minimal relative to the contributions of the categories of interest. The most effective applications of top-down evaluations are those that are combined with concurrent examination of the original bottom-up inventory data, so that the source of the errors can be identified rather than simply stating that the inventory is in error."

It is also important to remember that it is more efficient to assess uncertainty at the time the data are developed than afterwards.

3.2.4 Strengths and Weaknesses of Current Emission Inventories

Over the past 40 years, considerable work has been conducted in Canada and the United States on improving the accuracy and completeness of emission inventories. Nevertheless, problems still remain. A review of the strengths and weaknesses of emission inventories in North America provides an indication of the state-of-the-art and of the work that remains in order to improve it.

As of today, North American air quality managers have a good understanding of the emissions from major point sources, and they have used this knowledge in developing effective actions for reducing them. Models for estimating emissions from mobile sources have been continuously improved. The importance of natural and biogenic emissions has been recognized, and this knowledge has affected the design of air quality management strategies in regions where these emissions are significant. In Canada and the United States, emission inventories and models can provide quantitative estimates of emissions at national, state or provincial, and county (or their equivalent) levels for many source categories, and there is an improved understanding of the relative importance of various source categories to specific air quality problems. Air quality managers can use these inventories to track emission trends and to evaluate the effectiveness of measures designed to reduce these emissions. In Mexico, emission inventories have been completed for the Valley of Mexico and the states bordering the United States. The first national inventory will be released in the near future.

In spite of this progress, emission inventories in all three countries have significant weaknesses that will become increasingly important to address for future air quality management problems. These problems are probably typical of those in other industrialized countries. A discussion of them can provide useful guidance for other nation emission inventory development programs.

- Development of mobile source inventories, particularly regarding the speciation of volatile organic compounds, remains a challenging problem. National inventories in Canada and the United States have also indicated problems with the magnitude of carbon monoxide emissions and the temporal trend of nitrogen oxide emissions.
- Emissions for many important categories such as fine particulate matter and its precursors, biogenic emissions, ammonia, fugitive emissions, open biomass burning, and many other area sources are difficult to determine, and they remain inadequately characterized. Emission estimates for air toxics (e.g., the 188 hazardous air pollutants designated by the U.S. EPA) are particularly uncertain since there are so many of these compounds, so many potential sources (many of them area sources), and so little data for establishing emission factors or speciation profiles.
- Quality assurance and quality control procedures have not been strictly applied in the development of most North American emission models and inventories. In addition, the documentation of uncertainties and data sources in emission inventories has not been adequate to allow the uncertainties of the entire inventory, or of air quality models using the inventory, to be accurately estimated. These are issues that must be addressed in the initial design of a national inventory development program. Addressing them retroactively is expensive.
- Of necessity, emission estimates must be based on a limited number of emission measurements. If this number is not representative of real-world activity, the precision and accuracy of estimates developed from these measurements will be limited.
- The process for developing information on emissions with the kinds of spatial and temporal resolution needed for location-specific air quality modeling is problematic and a source of unquantified uncertainty in model results.
- Methods used to estimate emissions of individual chemical species in emission models must be kept up-to-date if they are to provide accurate information.
- Emission inventories must be developed and updated in a timely manner.
- Differences in how emission inventories are developed in adjacent countries create difficulties for jointly managing air quality.

3.2.5 Actions for Addressing Weaknesses

Typically, air quality management actions are initially focused on large point and mobile source emissions. Large point sources are the easiest to characterize and frequently the easiest to control. Mobile sources may be more difficult to completely characterize, but there are few points of manufacture. Thus, control devices can be readily installed during the manufacturing process. As emissions from these sources decline, however, remaining emissions are more evenly distributed across source categories that are even more difficult to characterize, model and control. These remaining sources will also grow as both population and economic activity increase. In this situation, errors in emission estimates

from smaller individual sources will have greater consequences. These consequences could range from wrongly identifying a pollutant that should be controlled to overlooking source categories whose control could result in more cost-effective emission reductions. As this situation is reached, it becomes increasingly important to address the kinds of emission inventory weaknesses described above. Even better, if these pitfalls are recognized in the initial design phase of a national emission inventory development program, it may be possible to avoid them or to address them in a more efficient manner.

The recent NARSTO (2005) assessment provided eight recommendations to the countries of North America on how to address the shortcomings of their national emission inventories. These recommendations are discussed briefly, as they may prove valuable to others. The first recommendation was judged to be the most important. The others were ranked as having somewhat lower priority, but in some cases they may also need to be addressed in the course of meeting the first objective. Given the different degrees of inventory development across countries, regions, and even pollutants, the secondary priorities will differ for each situation. These recommendations are sufficiently general so that there is little difference in the benefits accrued no matter what spatial scales or governmental levels at which the inventories are developed.

1. Reduce uncertainties associated with emissions from key under-characterized sources.

Comparisons of national emission inventories with ambient measurements and other independent measures should be used to indicate which source categories and pollutants are inadequately characterized and reported. Of particular concern are non-point sources including on-road and non-road mobile sources as well as fugitive emissions from industrial facilities, landfills, sewage disposal systems and feedlots. Sources of organic compounds, carbonaceous particulate matter, ammonia, and hazardous air pollutants are typically not well characterized. The highest priority for emission inventory improvement should be placed on these categories. *Resources should be targeted to reduce the greatest sources of uncertainty and focused on those source categories (or individual sources and conditions within these categories) whose control will be most effective in reducing costs and health risks while achieving air quality management goals.*

This is the most important recommendation, although addressing this recommendation may require acting upon the remaining priorities in order to identify where the greatest uncertainties may lie. Reducing known uncertainties in an inventory will provide a more accurate starting point for air quality management strategy development, resulting in more cost-effective approaches.

2. Improve speciation estimates.

Contemporary air quality issues, such as particulate matter and ozone and the identification of hazardous air pollutant "hot spots", require detailed information about the species being emitted from sources. Contemporary emission inventories are weak in this regard. *It is essential that source speciation profiles be continually updated and assessed. In addition, the related activity data must be developed to estimate more accurately speciated emissions of particulate matter and precursors, volatile organic compounds, and toxic air pollutants.*

Speciation is most important to the management of ozone and fine PM, but it can also be important to air toxics and to some extent climate issues (black carbon vs. organic carbon; different GHGs). More accurate quantification of the species being emitted will result in better air quality modeling results, and in better identifying which sources should be controlled in order to gain the greatest improvements in air quality.

3. Improve existing and develop new emission inventory tools.

Technical advances in instrumentation and computation are enabling emission measurements and analyses that have been previously impractical. Examples of these innovations include portable emission measurement systems for direct measurement of emissions under real-world conditions and the application of various remote-sensing techniques for measuring emissions or verifying emission estimates. Continuing development of these and other technologies is likely to improve emission inventory measurements and analyses. *Funding agencies need to continue to support the development and application of new technologies for measurement of emissions and ambient concentrations of pollutants.* Application of these technologies will assist the development of emission models that more accurately represent emissions from real sources in time and space.

Better emission measurement tools, inventory models, and inventory processors will improve the overall accuracy of inventories, again resulting in better air quality management information.

4. Quantify and report uncertainty.

Uncertainties in emission estimates and the parameters or data are used in emission models must be quantified. Uncertainties in emission inventories, processors, and models of Canada, the United States, and Mexico are poorly documented. As a result, the reliability of emission estimates cannot be quantified. *Quantitative measures of uncertainty and variability must be a standard part of reported emission inventory data. Agencies responsible for producing emission inventories must develop specific guidance on how to prepare and report information on emission uncertainties.*

This recommendation is probably the most difficult to describe and implement. Quantifying the uncertainty of an inventory estimate, and subsequently the estimates of pollutant concentrations, will enable decision makers to assess the likelihood that the desired outcomes will be achieved. A highly uncertain inventory means that the desired air quality improvements are less likely to occur because there is less confidence that inventory estimates accurately describe the actual situation. The ability to assess the potential for strategies to achieve the desired results will help decision makers to determine what steps are most likely to yield improvements.

5. Increase trans-national inventory compatibility and comparability.

As air quality problems become increasingly global in nature, it is vital that emission inventory development and reporting programs be coordinated internationally. Although there have been substantial improvements in reporting national emission inventories in mutually consistent ways, further work is needed to make these diverse inventories more comparable across organizations, purposes, political boundaries, and time periods. *International standards for emission inventory structure, data documentation, and data reporting should be developed. Such standards are needed to facilitate management of long-range transport and trans-boundary air quality issues.*

Inventories that are compatible across boundaries, and with inventories of other pollutants (GHGs, toxics, etc.), allows air quality managers to account for emissions that occur outside their domain of responsibility. This reflects the physical reality that pollutants do not recognize political boundaries. Comparability with past inventories enables air quality managers to more accurately assess the degree to which previous air quality management strategies have been successful.

6. *Improve user accessibility.*

The accessibility of emission inventories and emission models is impeded by the sheer size of the files and the cumbersome manner in which the data are reported and archived. *As improved accessibility to emission data is critical to meeting the diverse needs of the user community, increased efforts should be made to facilitate user accessibility to emission inventory data and models through the Internet and other electronic formats.* It is also suggested that emission inventories be made more transparent and easy to update. National inventories do not always contain the most recent emission information. Methods for allowing continuous updating with appropriately validated data from a variety of sources (researchers, industry, government agencies, etc.) need to be developed.

As more groups are able to access inventory data quickly and easily, more alternative approaches to air quality management can be developed. The more alternative approaches that are available, the more likely it is that air quality will be improved in the most cost-effective manner possible.

7. Improve timeliness.

Timely and historically consistent emission inventories are essential for assessing the current emission environment (and for keeping abreast of economic conditions and changes in technology) and for tracking progress in improving air quality. *Plans and processes need to be put in place for preparing and reporting national emission inventory data on a yearly basis.*

Timely inventory data more accurately reflect the actual situation. Ideally, inventories would be updated almost continuously so that air quality managers could see how the mix of emissions are changing due to external changes (population growth, changes in technologies, economic forces, etc.) and to changes in air quality management strategies. Continuously updated inventories remain far off, but a realistic goal is to minimize the time between when the inventory data are submitted and when they are reported.

8. Assess and improve emission projections.

Realistic projections of future emissions are important for developing and assessing strategies for attaining air-quality

standards and for evaluating future-year effects of new regulations. *Emission projection methodologies for all emission inventory sectors should be evaluated to determine the accuracy of past projections and to identify areas for improvement. Attention should be paid to assuring the compatibility of short-range projections that are more typical of air-quality related emissions with the long-range projections that are made for climate-change applications. Projections of future emissions are also dependent upon the quality of the base-year emissions. Therefore, realistic projections cannot be made unless these base-year emissions are as accurate as possible.*

Because most air quality management approaches estimate the future effects of air quality management actions, it is important to understand what changes are due to the air quality management strategy and what changes are due to changes in population, technology, etc. Improved projections also provide guidance regarding the level of emission reduction that may be needed to achieve and maintain the desired air quality over the long term.

3.2.6 Further Issues Regarding Emission Inventory Improvement

Issues of Scale: At the local level the issues of scale obviously become more significant. A more detailed understanding of source variability and how sources may differ from the national average need to be included. This involves not only the differences in technologies (processes, control technologies, etc.), but also how the sources operate. If there are significant differences between the local practices and national practices, the inventories will need to account for them. It is also likely that area and mobile sources will be more important at the local scale. As well, the mix of source types may result in air pollution issues at the local scale being different from those at a national or regional scale. A good example is $PM_{2.5}$ – In the eastern United States sulfates are the key to $PM_{2.5}$ reductions at the regional scale. At the local scale, however, carbonaceous PM tends to be the issue of most concern, because it is locally emitted (traffic, industries, open burning, and biogenic sources) and because the timescale of SO_2 chemistry results in sulfur dioxide being a gas-phase problem locally, but a PM problem farther down wind.

Groundtruthing Efforts: In current practice, inventories are most effectively used as the starting point for understanding the contributors to air quality problems. They form the basis for air quality modeling and for identifying the sources that are most significant within an airshed. Additionally, best practice dictates that receptor models, fuel-based (vs. vehicle-travel based) inventories, inverse modeling, and other approaches be used, independently, for inventory verification. This is important to undertake to identify problem areas in the inventories in order to minimize the impacts of erroneous inventory data.

Costs of Emission Inventory Development and Improvement: The U.S. federal government currently invests approximately $25 million per year to develop and update emission inventories (NARSTO, 2005). This does not include the amounts spent by state and local agencies (estimated at about $10 million per year) or the additional costs that would be required to address the shortcomings identified in the current inventories, which are estimated to be on the order of an additional $35 million per year (NARSTO, 2005). In Canada, about $6 million (U.S.) per year is invested in compiling its national inventory, not counting local and provincial efforts, and Mexico has spent about $600,000 (U.S.) per year in developing its National Emission Inventory. The cost of addressing emission inventory shortcomings in Canada and Mexico are proportionally similar to those of the United States – about $6 million and $1 million, respectively (NARSTO, 2005).

The cost of developing emission inventories is a function of their purpose. The relatively low cost of developing Mexico's national emission inventory is a consequence of its relatively low level of detail and the use of previously existing information. At the other end of the scale, the Electric Power Research Institute spent $50 million to quantify emissions of hazardous air pollutants from electric generation units (EPRI, 1994). The American Petroleum Institute, the U.S. Department of Energy, and others spent about $6 million to measure combustion emissions from refineries. These higher cost examples reflect the greater expense of obtaining highly detailed information, such as speciation of organic and metal compounds, that are present in flue gasses at very low concentrations (NARSTO, 2005). These examples show that as emission inventories become more detailed, they become increasingly expensive.

3.2.7 Conclusions

Accurate emission inventories are the foundation of all air quality management programs. They provide the essential information needed to understand the effects of air pollutants on human and ecosystem health, to identify which sources need to be controlled in order to protect health and the environment, and they provide the information needed to determine whether or not actions taken to reduce emissions have been effective.

In principle, the development of emission inventories would seem to be a relatively straightforward process, but in practice their production is found to be a very complex and demanding task. As initial actions to reduce emissions from large point sources find success, understanding and addressing residual air quality problems requires greater effort and emission inventories of increasing sophistication. Fortunately, much has been learned over the past 40 years that can make the development of new inventories a more systematic process. New measurement technologies and better understanding of the chemistry and physics of pollutant formation will continue to multiply the number of sources that can be measured directly and assure that these measurements reflect what is actually entering the atmosphere. Likewise, new methods for deducing and characterizing uncertainty will result in better understanding of the accuracy with which we know primary and precursor emissions. Finally, better data management software, the ubiquitous availability of low-cost, high-end computing, and growing availability of high bandwidth communications have made the development, maintenance, dissemination and use of large data sets practical for nearly everyone.

3.3 Measurement of Ambient Pollutant Concentrations

All ambient air pollutant measurements have the potential to support policy development and air quality risk management. The degree of support varies depending upon a number of factors and while the type of measurements and the design of the measurement program play a key role, a critical factor is the amount of time and effort dedicated to analysis and interpretation of the data. There is no denying, however, that, like emissions, measurements are part of the foundation of effective air quality risk management. Their main uses are as follows:

- Describing current risks and detecting potential future risks to human and environmental health;
- Documenting trends in order to demonstrate efficacy of past and present policies (e.g., emission reductions);
- Developing models capable of predicting ambient pollutant concentrations from knowledge of emissions and emission changes;
- Providing information needed to derive quantitative relationships between ambient concentrations and human health (or other adverse effects on the environment, climate or visibility).

This section focuses on describing how measurement data are applied to air quality risk management and provides guidance on technical issues to be considered in establishing a robust measurement program. The information is intended to highlight and briefly describe the key components involved in a 'best practices' measurement program and to present some of the present-day challenges.

3.3.1 Application of Measurement Data

Ambient pollutant observations can only support multiple objectives if they are interpreted to the full extent possible and thus, human resources dedicated to data analysis is essential. Rarely are measurements and all the supporting information utilized to the full extent possible before investigators move on to another dataset or to the next round of measurements.

The sections below discuss a range of issues related to the measurements that will best support a number of different applications. Given that human health is currently one of the main drivers of AQ risk management, the emphasis below is on measurements for this purpose. As has been listed earlier in this chapter, AQ measurements support AQ risk management in a variety ways. Measurement data are central to AQ-health studies, which serve to refine or derive concentration-response functions (CRF) and ultimately to gain more insight into the specific pollutants or sources posing the greatest risk. They must be collected to demonstrate compliance with current AQ standards and for tracking progress. The greater understanding of atmospheric processes that measurements provide helps diagnose the source(s) or causes of AQ problems, which is critical knowledge for identifying specific AQ management strategies. This may involve AQ models, which depend heavily on measurements for their development and for input data (i.e., initial conditions), particularly for AQ forecasting. Ultimately, measurements that are actively publicized so that the media, the public and decision-makers are better informed of AQ issues have the potential to have a significant influence on progress towards reducing the risks posed by poor AQ.

3.3.1.1 Ambient Air Quality Measurement for Health Research

Exposure to air pollutants has the potential to lead to a variety of adverse health impacts. Not all such impacts are likely to have been identified and/or adequately characterized given the diversity of exposure scenarios, the diversity of the population and the myriad of possible biological pathways. A range of health research approaches and detailed air quality measurements are thus needed to continue to advance knowledge and subsequent AQ risk management strategies. However, linking ambient observations of a range of air pollutants, as measured by standard monitoring networks that are in operation today and in the future, to acute and chronic impacts occurring within the general population is critical for deriving CRFs appropriate for AQ risk management. Demonstrating such 'real world' associations also establishes that air pollutant effects are relevant to actual conditions. This necessitates that ambient measurements be obtained to support both acute and chronic exposure health studies. The level of detail that these measurements should provide will depend upon the type of study they are intended to support. Detailed, prospective studies may be able to capitalize on the advances in measurement capabilities (Wexler and Johnston, 2006).

Perhaps the greatest challenge in working with ambient data is to derive correct CRFs for individual pollutants and for pollutant mixtures. This is because of confounding and differential exposure error amongst pollutants. Advances in statistical methods that can simultaneously exploit geographic and temporal variations in the interrelationships among pollutants may provide a path forward.

Assuming the time activity and distribution of susceptible individuals are similar among different urban populations, one would expect CRFs also to be similar and, while overall, CRFs for PM from different studies are surprisingly similar, there remains some variability. The reasons for this, of which there could be many, have not been fully resolved. Differences in the AQ data likely play a role in that there are typically variations between how the monitoring sites relate to the population (i.e., their location(s)) and to their actual exposures (e.g., prevalence of air conditioning, which influences indoor penetration, differs geographically). It is also likely that the nature of the confounding amongst pollutants and their exposure errors will differ among locations or cities. These issues lead to uncertainty in regards to which CRFs should be used to guide risk management.

Obviously, for cost-benefit analysis it is not possible or reasonable to have separate CRFs for every city or population of concern. Therefore, meta-analyses have been undertaken to derive 'generic' CRFs. This implies that the true benefits of a risk management strategy for a specific location will likely differ from the predicted value. Therefore, it is important to minimize CRF uncertainty and to characterize its magnitude in relation to its causes. This requires a better understanding of how measurement/exposure and multi-pollutant issues influence CRFs.

Clearly, no matter how detailed, ambient measurements cannot reflect what a person or most members of a population are truly exposed to. These measurements are only an indicator for some aspect of the air pollutant stresses that the population is confronted with. In reality, the relationship between these indicators and the actual exposures of the population differs from pollutant to pollutant and from one monitoring site to the next. Therefore, site location relative to the population and selection of the pollutants to measure at each site are critical issues.

The best practice is to measure multiple pollutants at any site that is established and to operate more than one site in the region containing the population of interest. Gaseous air pollutant time series developed for epidemiological research on the Toronto population have been based upon the average hourly concentrations from 3-4 different sites (e.g., Burnett et al., 1997). Their locations reflect some of the main outdoor environments people commonly encounter, such as residential and commercial areas and near roadway conditions. In many European cities it is relatively common practice to measure in the urban background and at "kerbside" locations, which provide information on the range of concentrations experienced.

Short term field studies can help identify optimum site placement if, as is usually the case, the number of long-term sites that can be afforded is small. In addition, acute exposure-effect studies and chronic exposure-effect studies have different requirements regarding site placement. The former requires that the measurements appropriately reflect temporal variations in the population's exposure while the latter is interested in how exposure levels vary across space. This could range from differences from one location in a large city to another (intra-urban) or differences between cities (inter-urban). Many studies have demonstrated that day to day variations in outdoor pollutant concentrations (i.e., the acute exposure signal) are highly correlated across a city. Others have shown that outdoor central site measurements do correspond reasonably well with the temporal variability in the average personal exposure (e.g., Brauer and Brook, 1997). However, this has also been found to vary from city to city and among pollutants (Saranat et al, 2001; Kim et al.,

2006), the latter of which represents one source of differential exposure error. When developing a time series for an acute effects study it is important to test this correlation.

In terms of acute effects, $PM_{2.5}$ is considered to the most important pollutant to understand better. New monitoring programs for total mass and chemical speciation have been established in many countries. One such dataset has been obtained from sampling in downtown Toronto (Lee et al., 2002) and these data are being used to determine if significant differences in the strength of health associations can be detected among different constituents. Since only one daily sampling location was possible for the three years of measurement a short term study was undertaken to examine the degree to which measurements at up to seven different locations across the city were correlated and how magnitudes differed. The case study described in the next sub-section demonstrates how much spatial variability was found for fine particle ($PM_{2.5}$) total organic carbon, elemental carbon and sulphate. These are the main chemical components found on urban $PM_{2.5}$ in eastern North America in the summer.

Case Study: Urban Spatial Variability in $PM_{2.5}$ Chemical Constituents in Toronto, Canada

The purpose of the case study presented in this section is to demonstrate the value of special studies to better understand local scale spatial patterns around central monitoring sites used for health research. From Aug. 10 to Sept. 3, 2000, separate day and night filter samples were collected simultaneously at six sites in central Toronto. These locations surrounded the central site selected for development of a three year time series for an acute effect studies. The purpose was to gain more insight into the representativeness of the central site measurements in terms of population exposure and to assess the impact of local sources on particulate organic carbon (OC) and elemental carbon (EC) concentrations. The overall results of this campaign, referred to as the Urban Spatial Variability Study (USV), are summarized in Table 3.3.1.

Two mobile laboratories were used and were parked at four different public schools. The first lab, MSC, spent the first half of the study at the location indicated as WS on the map shown in Figure 3.3.1 and the second half at PS. The other mobile lab, MOE, was at location DS followed by location RS. GB is the location of the central site used for the three years of daily measurements for the purpose of epidemiological research, as discussed above. The other four locations were equipped with portable samplers mounted on utility poles. Site 3 was in the downtown core near the Bay Street financial district; Site 2 was downtown on Church St.; Site 1 was in a relatively low density residential area northeast of downtown several blocks north of the Danforth (indicated as Bloor on the map) with the Don Valley (a deep valley with a river and a highway at the bottom) just to the west; Site 4 was in a residential area southwest of downtown. As shown on Figure 3.3.1 these sites were all within 7 km of each other.

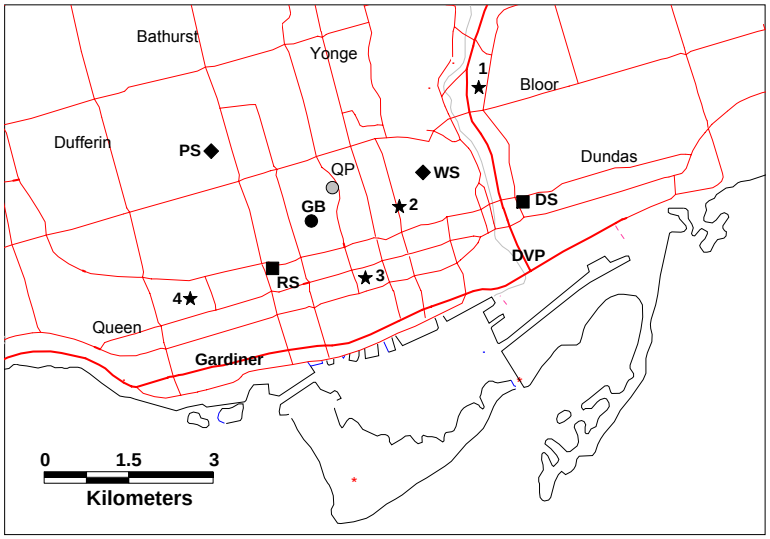

Figure 3.3.1 Location of the sampling sites in downtown Toronto during the Aug. 2000 USV study. The red lines are the major roadways. One mobile lab parked at WS for the first half of the period and at PS for the second half. Similarly, a second mobile lab was at DS followed by RS. The portable sampler locations are shown by stars and the central site being used for the epidemiological study (GB) is indicated by a dark circle. QP is the main downtown monitoring site of the Ministry of the Environment at Queen's Park.

Mean total particulate carbon (TC) and elemental carbon (EC) concentrations are included in Table 3.3.1. TC concentrations were notably lower at site 1 and GB locations followed by site 4. The lower level in the residential areas is consistent with expectations, but the low concentration at the GB site is surprising given its downtown location and the traffic in the area. The reason is that the sampler was on a third floor rooftop suggesting that there can be considerable vertical gradients, as might be expected with a near surface source such as traffic. TC was larger during the day each site due to increased emissions. The diurnal pattern in EC is much more likely to be dictated by traffic emissions and it was most dramatic at sites 2 and 3 and at GB. This behaviour is reasonable given that these three sites were closest to the central business district. The larger day-night differences are because people work during the day and are generally gone at night.

Table 3.3.1 Mean concentrations ($\mu gC/m^3$) of total particulate carbon and elemental carbon collected simultaneously at 7 locations in central Toronto during the USV study (Aug. 2000).

	Total Fine Particle Carbon Averages ($\mu gC/m^3$)						
Location	**1**	**2**	**3**	**4**	**WS-PS**	**DS-RS**	**GB**
24 hour	6.6	8.8	8.0	7.2	9.9	8.1	5.7
Day	8.6	11.1	10.0	9.5	13.0	10.3	7.4
Night	4.8	6.8	6.1	5.1	7.1	6.0	4.1
	Elemental or Black Carbon Averages ($\mu gC/m^3$)						
24 hour	0.82	1.19	0.97	0.73	1.21	1.23	0.80
Day	0.89	1.47	1.15	0.81	1.23	1.23	0.97
Night	0.75	0.95	0.80	0.66	1.19	1.23	0.66

There was a greater number of different measurements collected at the GB site and the two mobile labs and thus, their OC, EC and sulphate measurements are compared in Figure 3.3.2 a,b,c. As can be seen, sulphate varied by close to a factor of 50 from the lowest to highest concentrations, while OC exhibited much less variability. EC variation was between OC and sulphate. The relative concentration differences between sites were smallest for sulphate and largest for OC. Again, EC was in between. Both OC and EC exhibited systematic diurnal variation with higher concentrations more-frequently occurring during the day. OC had the largest day-night difference and sulphate showed no such pattern.

In Toronto, like many other cities, particulate sulphate is influenced by long range transport, which is dictated by synoptic weather patterns (i.e., positions of high and low pressure areas). The pattern in Figure 3.3.2c reflects the dominance of this transport, which within the scale of Toronto, was clearly more important than photochemical production (i.e., there was no systematic daytime peak). OC and EC can be affected by long range transport and local emissions, but the relative proportion of these factors is not well-understood. The comparisons in Figures 3.3.2a,b,c strongly suggest that local emissions dominated for OC and the spatial differences, including the vertical difference, further support this conclusion. There are relatively large changes in OC concentration over small distances indicating that very local sources play a role. In addition to traffic, one local source that was hypothesized to have contributed significantly was meat cooking. On Aug. 11-13 the 'Taste of the Danforth" festival was held. This entails many restaurants cooking outside all along Danforth Ave. (labeled as "Bloor" on the map), which is closest to the MSC site. In addition, the winds were from the northeast to southeast direction for about 75% of the hours during this period, including part of Aug. 14. Figure 3.3.2a shows that OC at the MSC site was much larger than the other two locations. Obviously, the festival would have brought more vehicular traffic to that region of Toronto and so it is not possible to attribute all of the extra OC to meat cooking. However, these results serve to highlight the complexity of apportioning urban OC to all of its potential sources.

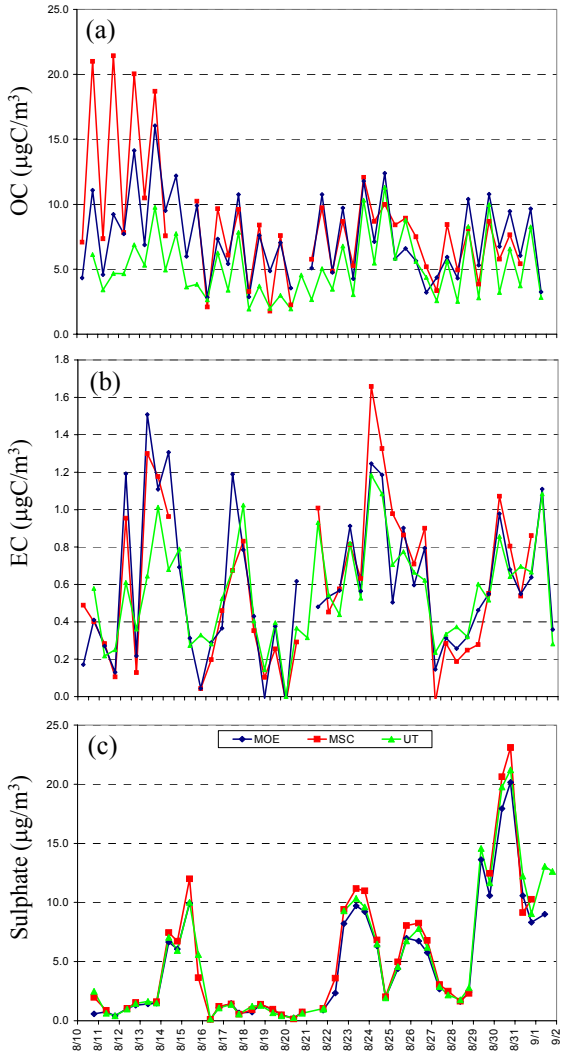

Figure 3.3.2 Time series of (a) OC, (b) EC and (c) Sulphate at the three main sampling sites operated during the USV study in Toronto in August 2000. 'UT' corresponds to the GB or central site used for three years of $PM_{2.5}$ sampling for epidemiological research.

One of the surprises resulting from the USV study was that compared to OC, EC had a much stronger correlation with the sulphate pattern and hence with the long-range transport or synoptic weather conditions. Initially, it was hypothesized that EC, being a strong traffic-related pollutant, particularly from diesel emissions, would exhibit the greatest spatial variability and that OC, having a secondary component, which would be greater in "aged airmasses", would be more associated with the long-range transport signal and have more spatial homogeneity. However, the USV study did not support this hypothesis. Based upon the period of measurement in August 2000 it appears that EC is more spatially homogeneous and that a central monitoring site, such as GB, is more likely to provide representative exposures for a time series study (i.e., an acute health effects study) for EC than for OC. Estimating representative exposures for the latter will be problematic. This case study highlights the difficulty of selecting an ideal site to represent population exposure for all $PM_{2.5}$ constituents. Thus, while $PM_{2.5}$ mass maybe relatively homogeneous, spatially, and exhibit similar day to day variability across an urban area, this may not be the case for the different chemical constituents. Similarly, there are differences in how well gaseous pollutant measurements from a central site represents concentrations and temporal variations across the Toronto population. Average inter-site correlations (i.e., for all available pairs of sites in Toronto in 2000-03) for CO, SO_2, NO_2, O_3, $PM_{2.5}$ and PM_{10} are: 0.41, 0.78, 0.77, 0.90, 0.96 and 0.84, respectively. Ideally, all of these separate exposure errors (particulate and gaseous pollutants), arising from reliance on a limited number of central ambient measurement sites, need to be quantified and extended to personal exposures, as well. Without this information and methods to include it in epidemiological studies, it will be difficult to determine, definitively, which gaseous pollutants and/or particulate species are most responsible for the observed associations (i.e., adverse acute health effects).

Limitations in Linking AQ Measurements and Health Data

Short term field studies and analysis of existing data, as highlighted above, provides insight on site representativeness and exposure errors and also demonstrates the difficulty of site selection. While more time for site selection could allow for deployment of mobile measurement platforms and short term saturation monitoring studies to further optimize site selection, it is also clear that the "perfect site" does not exist. However, in addition to the exposure errors, that are inevitable, it is also important to consider the nature of the health effects being studied and the type of health data that are or will be available.

True acute responses likely arise immediately after exposure (i.e., within minutes to hours), but may linger and/or continue to grow in severity for at least a few days. This time course is not well understood, but can be expected to differ depending upon the physiological functions being affected (e.g., respiratory, cardiovascular), from person to person and depending upon how the exposure conditions change during this period. Administrative health data do not provide the time resolution needed to improve understanding of this etiology. Emergency room visits, hospital admissions, deaths, absenteeism, etc., are recorded as daily counts corresponding to midnight to midnight. While at a minimum this implies that only daily pollutant averages and exposure metrics derived from these values are necessary, hourly measurements can be utilized in the form of daily maximum readings over shorter durations. This permits some assessment as to whether the acute responses are associated more strongly with short term peak levels or higher concentrations sustained over 24-72 hr periods. Confounding amongst these different metrics will hinder definitive conclusions about what acute exposure period is of most importance to the population at large. However, it may be possible to gain some insight for more detailed follow-up using different study designs (e.g., prospective panel studies) and thus, hourly measurement data are valuable to obtain.

A rule of thumb is that 2-3 years of daily air pollutant measurements for a population of about 1 million provides sufficient power for reasonably precise determination of the magnitude of an acute effect on total non-accidental mortality. Obviously, the more data the better, as shown in Volume 2 of the 2004 USEPA Criteria Document for Particulate Matter (PM). The precision of the risk estimates tends to increase as the product of the daily death rate and number of PM measurement days increases (EPA, 2004). Constructing a time series of data with values every day is not an issue when hourly measurements are available. When integrated samples collected over 24 hr periods (daily samples) are all that is possible (e.g., for measurements of some PM chemical constituents) then common practice is to collect samples less frequently than every day. For many years TSP and PM_{10} samples were only collected every sixth day in most circumstances. Although an improvement, many new networks are only providing samples every third day, which means it will still take three times longer to have sufficient statistical power for an acute effects or time series study.

There are relatively few health and air quality/exposure databases for chronic effect studies. Health data for such purposes are extremely valuable. Thus, when discovered and access is granted exposure information is derived from whatever is available. The key exposure signal needed for these cohorts is based upon spatial or geographic variations in concentrations. If the cohort is developed prospectively then it is more likely that air quality measurements can be included in the program (e.g., Spengler et al., 1996; Avol et al. 200x). However, demonstrating that such measurements are representative of longer term or even lifetime exposure remains an issue (Brook and Spengler, 1997) and mobility of cohort members may need to be taken into consideration (Jerrett et al, 2006). Some prospective AQ measurements can also be undertaken to support chronic exposure studies utilizing existing cohorts. This may be necessary since often such cohorts of opportunity were established for entirely different purposes (i.e., to study other health issues unrelated to environmental health). However, it is more likely for existing AQ information and various modeling techniques to serve as the basis for the exposure information.

As monitoring programs are established or expanded, enhancing the spatial coverage of the sites could potentially improve the applicability of the data for future chronic health studies. However, even with close contact with health researchers it is difficult to anticipate all the areas that might need to be covered unless there are specific cohorts in mind. Even still, resource limitations usually make it impossible to monitor all locations, especially if only done on the speculation that a health study might use the information. In addition, intra-urban variations in exposure are now recognized as an important signal to exploit in chronic studies and again, it is not likely possible to operate enough monitors. Mobile measurement platforms (e.g., Bukowiecki et al., 2002; Westerdahl et al. 2005; Yli-Tuomi et al., 2005; Kolb et al., 2004; Polina et al., 2004; Lee et al., 2006;) can assist in studying spatial patterns and in optimizing site placement. They are becoming more common as a facility for monitoring agencies and current advances in technology are allowing even more-sophisticated measurements to be obtained. There are a range of challenges related to appropriate deployment of mobile labs that need to be considered (e.g., Xu et al, 2006). However, measurements alone cannot provide all the spatial detail desired. Thus, alternate sources of information and/or a range of spatial models, statistical or physical, are becoming increasingly important to develop. These will be discussed further below.

More targeted AQ risk management will ultimately require more knowledge regarding the specific pollutants or sources posing the greatest risk to human health. Presumably, standards for these pollutants (e.g., a chemical component found on $PM_{2.5}$) could be established or control measures could be implemented on specific sources. Acquiring the necessary knowledge requires highly specialized health studies. These need to be supported with much more detailed ambient air pollutant measurements and a better understanding of the relationship between ambient concentrations and actual exposures. These are likely to come from field studies as opposed to monitoring programs. The types of measurements needed and other health research related needs that the atmospheric science community can potentially satisfy were recently discussed by NARSTO (www.narsto.org). There have been considerable advances in measurement capabilities largely due to extensive new research, worldwide but especially in the U.S., on $PM_{2.5}$. Applying these capabilities for health research can be expected to lead to new understanding (Wexler and Johnston, 2006). However, even stronger collaboration amongst health, exposure and atmospheric scientists will be needed to take full advantage of these advances given the level of understanding required to operate equipment and design effective studies. At the same time there is increasing interest in studying a wider range of health endpoints (e.g., cancer risks, impacts on the fetus and trans-generational effects) and a wider range of air pollutants (e.g., toxics, radon, aeroallergens). This will hopefully lead to a more holistic understanding and approach to risk management. However, achieving this level of understanding will be extremely challenging, requiring a sustained multi-disciplinary research effort in environmental health for many years to come.

3.3.1.2 Ambient Air Quality Measurement for Tracking Progress

One of the primary objectives of most monitoring networks is to track progress towards achieving standards or maintaining good air quality. Even so, there are differences among countries with respect to the overall philosophy and objectives of the measurement. The resources available and the relative amount dedicated to AQ monitoring also differ. It is beyond the scope of this chapter, however, to list and discuss strengths and weakness of all the different monitoring networks in the world. Information on some of these networks can be found at the various websites referenced below.

In most mature networks the sites are generally in standardized locations, all the criteria pollutants are being monitored and long data records have been and are continuing to be collected at several locations. To study trends, ensuring a long, unbroken record is important because changes can be small and gradual and are obscured by meteorological variability. To demonstrate progress in response to a new policy (i.e., accountability) time series length is not as important. However, sufficient, high quality baseline measurements in advance of implementation of the emission reduction measures are critical as is continuation of these measurements after the reductions have been implemented. Existing sites providing the base line data need to be identified and if necessary improved (e.g., adding high sensitivity instruments or new measurements). AQ model runs simulating the expected changes may provide insight regarding the size of the expected change at these "accountability sites" and hence may indicate that more sensitive measurements will be needed. These runs may also point towards other, unmonitored locations, where collecting measurements could help to more-effectively and more-rapidly demonstrate accountability. Furthermore, in the future if the measurements agree with the model predictions this lends considerable credence to the model. Conversely, disagreement will likely help identify model weaknesses.

Lack of site continuity, lack of sensitive enough measurements and under-representation in some geographic areas hinders trend analysis or demonstration of accountability. Consequently, closing or moving monitoring sites with long records should be considered very carefully. When a site is moved both the new and old site should be operated simultaneously for as long as possible in order to quantify concentration differences. As indicated above, in many areas the concentrations of some of the criteria pollutants are well below their standards, but instruments with the necessary degree of accuracy and precision for documenting and studying low level variations are not being utilized. This is particularly the case for CO and SO_2.

A weakness of many existing networks is that they have focused more-extensively on urban areas. Consequently, there are much fewer rural measurements and the length of their time series is shorter. For example, worldwide, there are few long term trends on rural NO_x levels, despite its importance to understanding regional O_3 and $PM_{2.5}$ and the impact nitrogen deposition can have on ecosystems. Furthermore, this limits our current ability to assess how the growth in the size and density urban areas (e.g., sprawl) is impacting proximate regional air quality. Urban-rural pairs of measurements (Brook et al., 1999) are becoming more common. They are essential for untangling contributions and trends attributed to local/urban sources vs. upwind sources from regional scale transport. Conversely, some classes of pollutants have traditionally been measured in rural and remote areas (e.g., POPs, acid deposition, sulphate) and thus, understanding of the conditions in populated areas is more-limited.

In some countries or jurisdictions, previous data and models have provided knowledge to allow some downsizing of networks with limited or no loss of information (e.g., leaving sufficient sites to monitor trends and model population exposures (RIVM)). This helps reduce costs. Reducing the number of pollutants measured at each site can also preserve valuable resources. However, this will limit the ability to understand the reason(s) for any observed changes and to undertake source apportionment. Adding more equipment increases the need for qualified personnel to maintain increasingly sophisticated equipment and also for data QA and interpretation. It should be kept in mind however, that the main expenses associated with a site are the initial installation, maintenance of the infrastructure, rent and a local operator. Adding more equipment once these costs are incurred is incremental.

More and more agencies responsible for AQ measurements are enhancing activities at selected sites (supersites or AQI sites or core sites) and then intending to collect fewer measurements at sites in between these locations. This approach supports a greater amount of AQ science at the supersites and can provide cost savings. The level of activity at the supersites often varies over time depending upon funding and the current "hot issues". However, stability for a core set of measurement is very important since these supersites are likely to be ideal for studying long term trends and potentially for accountability purposes. The measurements obtained at the "in between sites" or "satellite sites" typically depends upon the objectives of the program funding the work. With respect to air quality and smog, such sites are most likely to monitor O_3 and $PM_{2.5}$ or PM_{10}. This is driven by the fact that these pollutants are closer to or over existing standards and there is a perception that they are most important with respect to health effects. While this latter point is valid, in many areas these pollutants exhibit less spatial variability implying that fewer sites are needed. This is, perhaps, less likely for $PM_{2.5}$ because although its total mass may vary relatively slowly over space some chemical constituents, particularly those related to primary emissions, will potentially vary much more rapidly. Thus, more knowledge is needed to in order to determine how best to optimize the number of $PM_{2.5}$ sites and $PM_{2.5}$ speciation sites.

Air quality models and the supporting emissions inventories are continually improving and are being more widely applied over longer time periods. This presents an opportunity to apply models to identify the more useful locations and for "filling in" AQ information in between sites. This will be discussed more below.

3.3.1.3 Ambient Air Quality Measurements for Modeling, Process Studies and Source Apportionment

Both monitoring and field study data are necessary to determine which sources and source areas are contributing to the observed concentrations. Multiple pollutant and PM and VOC speciation datasets are necessary and are of most value when the measurements are collocated. Spatial and temporal coverage is the greatest asset of monitoring network data. Combined with meteorological information (e.g., trajectories) relatively complete conceptual models can be devised and considerable information about contributing sources areas can be obtained (e.g., Brook et al., 2005). As highlighted by discussion and examples in Chapter 6 of the NARSTO PM Assessment, all forms of data analysis, from the simplest to the most complex, will provide insight. However, this requires that resources be continually dedicated to this purpose and that experienced analysts, with backgrounds in atmospheric chemistry, meteorology and statistics, are employed.

Monitoring network data can also play a valuable role in model evaluation. This role is becoming more important because many models are being run continuously for AQ forecasting and detailed field studies cannot be undertaken continuously. These new long term modeled datasets are offering many new opportunities to learn a great deal about how the models perform and the quality of the emissions information. While the network data tend to lend themselves more to operational evaluations of the model, there are opportunities for diagnostic evaluations as well (see Section 3.3). Increased use of continuous or semi-continuous particle composition instruments for monitoring can be expected to provide greater opportunities for such evaluations. Network data are also critical to define the model's initial conditions. Advances in rapid data assimilation and QA (Menard and Robichaud, 2005) have been occurring for this purpose. These need to be continued since there are many potential applications of these assimilated datasets.

More detailed measurements, which can only be sustained for relatively short field studies are ultimately needed to study atmospheric processes (dynamical, chemical and physical) and for more-detailed diagnostic model evaluations. The scope of these studies can vary greatly from a small team collecting measurements to study one process (e.g., Makar et al., 1998) related to one model module to large collaborations across institutions (e.g., EMEFS, 1988 and 1991; Atlanta and Houston Supersites, ICARTT, 2004), all focusing, simultaneously, on their own specialties and interests. Just as there has been advances in what can be reliably measured at monitoring sites there have been significant gains in what can be measured during field studies. Methods have improved and some advanced technologies are commercially available and are reasonably reliable and straightforward to operate with highly qualified technicians. This means that the ease with which a field study with highly comprehensive and technical measurements can be launched is now much greater than 5-10 years ago. This is beneficial for obtaining, quickly, much more data at more locations from which one

can study, in detail, source apportionment and atmospheric processes. However, the risk has never been greater for valuable data to be under-interpreted. Thus, with these new capabilities comes an increased need for highly qualified and creative experts in data analysis along with careful advance thinking regarding the underlying hypotheses motivating any venture into the field.

3.3.1.4 Ambient Air Quality Measurements for Public Information

In the long run informing the public is critical to the process of AQ management. They are the ultimate decision-maker since when a large enough majority decide that an issue is important elected officials are more likely to respond. This can create opportunity for progress on AQ. Science/environmental advisors must be ready with the right advice, based upon emissions, measurements and modeling information, when government leaders are prepared to make decisions. In recent years, health research results have garnered considerable attention in the media and, as discussed above, AQ measurements are critical to this research.

In terms of public information, the Air Quality Index (AQI) has been utilized for many years. In cities or regions where there are more frequent bad air days the public is generally more aware of AQ issues, at least partly due to the publicity of the AQI. The form of the AQI is similar in many countries, reporting air quality using descriptive terms such as good, moderate, poor, very poor, unhealthy, etc., (http://www.msc-smc.ec.gc.ca/aq_smog/aqcurrent_e.cfm; http://airnow.gov/index.cfm?action=static.aqi). This approach is easier for the public to understand and act upon as opposed to reporting actual pollutant concentrations. It has been designed to identify the worst effects that may result from the mixture of pollutants currently being measured and to describe the prevailing air quality. However, ozone and particulate matter are more often the driving pollutants (i.e., leading to an AQI other than very good or good). The AirNow website (http://airnow.gov/), which reports actual ozone and $PM_{2.5}$ concentrations all across the U.S. and Canada along with colour codes indicating the AQI, represents a significant advance in the information being publicly provided. It allows for the spatial extent of elevated air pollutant levels to be visualized and for users to quickly compare their region to others. The systems developed to obtain and synthesize this information and present it in near real time have only recently become possible and this infrastructure is also critical for improved AQ forecasts. Similar systems for providing realtime AQ data and/or AQI values exist for many countries (e.g., the Netherlands: http://www.lml.rivm.nl/data/smog/index.html; Mexico: http://www.ine.gob.mx/). These are continually evolving and being integrated into multi-national systems.

The increase in knowledge regarding air pollutant health effects has been leading to growing interest in upgrading or modernizing the AQI. One such program is the Air Quality Health Index (AQHI) being developed in Canada. This is currently being pilot tested in the Province of British Columbia (http://www.airplaytoday.org/). The goal of this pilot is to introduce the AQHI to the public and gather feedback, especially from people who are sensitive to air pollution. The unique feature of the AQHI is that it is based upon recent epidemiological results from across Canada. In addition, it considers multiple air pollutants simultaneously and they all contribute to the index value in every case.

The AQI has typically been for reporting current conditions so that the public can respond immediately. In recent years, however, the capacity to predict future O_3 and $PM_{2.5}$ concentrations, using physical and/or statistical models, has improved. Thus, the public can be informed in advance. There are also efforts being planned to predict more of the pollutants considered in the AQI or AQHI (e.g., SO_2, NO_2). Consequently, AQ advisories are being supplemented with daily air quality forecasts (http://www.msc-smc.ec.gc.ca/aq_smog/aqforecasts_e.cfm; http://www.msc-smc.ec.gc.ca/aq_smog/chronos_e.cfm), but the information is not usually widely distributed (e.g., radio, major television networks) until high levels, warranting an advisory, are predicted. Thus, despite the value of daily information for susceptible individuals, it is unclear if it is influencing the general public. Daily forecasts are also available in the U.S. (http://www.arl.noaa.gov/ready/ozone/), where some media are publishing maps every day. Countries outside of North America are also providing forecasts (e.g., http://www.epa.vic.gov.au/Air/AAQFS/default.asp).

Regular AQ forecasts, made possible through real time reporting and assimilation of measurements and advances in AQ modeling, clearly does present an additional opportunity, beyond advisories, to inform the public of AQ issues (i.e., increase public awareness). A telephone-based survey undertaken after the original AQ advisory program for Canada had started indicated there was partial success in achieving this goal (Stieb et al., 1996), but few actually changed their behaviour. Ideally, when public information programs are planned there should be some collection of baseline data, as opposed to retrospectively initiating such activities. This should give a truer picture of how the public's awareness and/or behaviour changed.

Ultimately, poor AQ situations need to be minimized through preventative measures (i.e., new policies on emissions or activities producing emissions) and increased public awareness help create the climate for political action. However,

providing routine, reliable and understandable AQ information to susceptible members of the population allows them to reduce their own exposure. This should not be underestimated as an important component of AQ risk management. Therefore, communication plans and health messages require careful consideration and regular evaluation for effectiveness.

3.3.2 Technical Issues in Establishing a Measurement Program

Air pollutant measurements should only be taken if there is an ongoing commitment to a recognized standard of quality, and a plan for data archival and for interpretation. Poor quality and/or incomplete data or data of unknown quality have limited usefulness. It must also be recognized that knowledge and technology are continually improving and thus, to the extent possible, new measurements should seek to use the most current, accepted methods. This will increase the probability that the data are acceptable far into the future. In the long run, a small amount of high quality measurements will be of more value than many measurements collected with insufficient documentation, quality assurance and interpretation.

Air quality measurement activities generally fit into one of two categories:

1. Monitoring - A core set of systematic measurements at well-selected locations that are maintained indefinitely for trend analyses (i.e., evaluate effectiveness of current policies), to determine if an area is complying with or achieving an official air quality standard or guideline and to identify emerging problems as soon as possible, which may involve ongoing environmental health studies (e.g., epidemiological studies).
2. Field studies - A relatively short period (<2 years) of more-detailed or more-specific measurements collected within a well-defined geographic area or at a given location or for a given population. These data are essential for development of conceptual models, source-oriented models, more-refined source apportionment studies and for understanding the relationship between emissions, ambient concentrations and personal and/or population exposure. A variety of prospective health studies may also derive their exposure information from air quality field studies.

Data from monitoring programs lend themselves to a standard set of reports documenting current conditions, trends and comparison with other geographic areas. Quick release of such information into the 'right hands' helps to keep AQ management issues in the forefront. A range of user-friendly software tools that can process the air quality data along with meteorological data are becoming common. This enables air quality scientists and managers to examine some of the causes of pollution events in near real time, providing information that, in the past, could take a year or more to obtain.

Field study data are usually more complex and less standardized. Sufficient time and resources need to be dedicated to working with the data after the study. At a minimum, 1-2 years is likely necessary. To guide the planning of the study and subsequent data analysis there needs to be a set of testable hypotheses in place well before the study begins. Generation of new scientific knowledge is likely part of the study objectives and thus the first official reporting of the data tends to be in peer-reviewed journals. This is the only approach for ensuring credibility of the data, which is ultimately necessary for AQ managers to be confident with subsequent policy decisions. Prior to journal publication, preliminary results often appear at conferences and before that formal workshops help maintain momentum and offer an opportunity to combine data and build consensus. AQ managers or their advisors should participate at this stage in order to stress the policy issues they are expecting the results will help inform.

Air quality measurement programs are expensive and so in advance, must have clear short and long-term objectives. Ideally, the program will be sufficiently flexible and broad to support multiple objectives, some foreseen and others not yet appreciated. Additional resources/expenses to insure data completeness, quality, analysis, interpretation and reporting should not be overlooked since they are likely to be incremental (i.e., a small cost relative to the overall cost of obtaining and maintaining the data). The key technical issues to consider when establishing a measurement program are:

- What to measure and how often
    - Ideally, multiple pollutants should be measured at the same site to assist in interpretation and to serve more than one objective
    - Temporal resolution; could range from seconds to days
    - Measurement methods to be utilized
    - Length of time series to be collected
    - Personnel needs in the field and lab/office

    - Criteria for introduction of new technology or additional pollutant measurements if the measurements are part of a longer term program.
- Siting criteria and where to measure
    - Impact of local sources - may be desired or important to avoid
    - Type of sites include: source-oriented, such as curbside or other high impact areas (e.g., hot spots); local or neighborhood scale; urban background; regional background (indicative of long range transport)
    - Representativeness to population and/or to region needs to be assessed
    - Geographic coverage and spatial density of sites if the program involves a network
    - Site access, serviceability and security
    - Documentation of site meta-data
- Quality assurance (QA)
    - Required level of accuracy, precision and data completeness
    - Frequency of collection of specific QA measures (e.g., duplicates, blanks, zeros, spans, calibrations, external audits)
- Data archiving and reporting
    - Data turn-around time and policy for data exchange and criteria for permitting use in publications
    - Provision of data to national or international public archives
- Consistency with methods used at other sites in the same and different networks and between countries

Addressing these technical issues, which will be expanded upon below, before measurements start helps ensure that the data are of greatest value. In establishing a measurement program it may also be relevant to consider the potential applicability of the data for evaluating models and also for integrating the data with model output and other information to improve the detail and coverage of ambient concentration information (see below). Linkages to personal exposures and the ability to quantify the degree of exposure error associated with using the measurements for health research may also be necessary to consider.

3.3.2.1 What to measure and how often

For a wide range of both gaseous and particulate pollutants Chapter 5 of the assessment published by NARSTO (McMurry, Shephard, and Vickery, 2004) provides considerable detail on what can be measured, the methods available, how reliable they are and reasons such measurements might be needed (e.g., for health effects studies, compliance monitoring, visibility, scientific understanding, etc.).

When feasible, greater frequency of measurement (i.e., finer time resolution) is preferred because this permits a much better understanding of source contributions and atmospheric processes (Wexler and Johnston, 2006). If a standard exists then its 'form' or 'metric' (e.g., hourly maximum, 8 hour maximum, 24 hour, annual) will dictate that a certain resolution be achieved. Inclusion in a real time reporting program, such as may be needed for air quality index and air quality advisory purposes or for air quality forecasting, will also likely demand that data be available on a frequent basis (e.g., hourly). Choice of resolution also has an impact upon the resources needed for QA, data storage, as well as data analysis and interpretation activities. Ultimately, the time resolution that is measured is dictated by instrument capabilities.

Although air quality standards or other types of regulations/guidelines require that several common pollutants are monitored indefinitely, several other pollutants or trace gases are important to measure in support of AQ management. This wide range of trace atmospheric chemicals can be classified in a variety of ways. Here we choose to consider four classifications, however with any such attempt, the distinctions are blurred. These are: pollutants formed during combustion; pollutants released from the surface or fugitive releases; volatile organic compounds (VOC); secondary pollutants.

Arguably the largest group, in terms of quantity of emissions, contain pollutants associated with combustion emissions. Many of these pollutants are produced and emitted simultaneously, which presents opportunities for co-management. This includes nitrogen oxides (NO, NO_2 or NO_x), carbon monoxide (CO), fine particles ($PM_{2.5}$), ultrafine particles ('ultrafines' or $PM_{0.1}$) and, depending upon the presence of sulphur in the fuel, sulphur dioxide (SO_2). The first two, plus SO_2 and some form of particles (e.g., total suspended particulate-TSP, particulate matter less than 10 μm in diameter-PM_{10}) are generally referred to as criteria pollutants.

Particles are monitored according to total mass below a specific size, but in the case of ultrafines measurements are

based upon total number. Combustion particles are also composed of many different chemical compounds, which will be discussed below. However, one important particle phase constituent that should be explicitly included in this group is black carbon (BC). Also, referred to as elemental carbon or soot, BC is being measured more frequently due to its strong link to traffic emissions, especially diesel particulate matter, which is becoming more and more recognized as posing a risk to health. In Germany, the National Environment Agency has implemented a BC ambient concentration limit of 8 $\mu g/m^3$ (arithmetic annual average value) and regulations in other jurisdictions (e.g., California) are being considered or are in place).

Carbon dioxide (CO_2) is typically the most abundant trace gas in combustion emissions thereby linking, at the source, the issues of air quality and climate change (note that BC and aerosols are also important pollutants regarding climate). All of the pollutants mentioned above can be measured with instruments that provide hourly or better time resolution and that are capable of reliable, realtime data storage and transmission. With the exception of the particle-related measures, regular, automated QA or calibration can be included in monitoring routines. Calibration of the particle instruments is generally not possible because absolute standards are not available. Other approaches to insure data quality are thus, necessary.

Ensuring accurate, precise measurements is much more challenging when concentrations are low (approximately <5 ppb for SO_2 and NO_x; <1 ppm for CO; <10$\mu g/m^3$ for PM) and thus, extra care must be exercised. In this case, standard calibration procedures may not be appropriate. Low concentration measurements have become more of a concern because past policies have led to decreased emissions and also because of more interest in rural measurements. Such low concentrations are typically well below current AQ standards and thus there is a tendency for network managers to be less concerned about accuracy and precision. This should be avoided, however, because epidemiological studies have and will continue to use these data. Furthermore, associations, albeit controversial, with adverse health endpoints have been detected at low concentrations and thus, reliability of these data can be expected to come under more scrutiny. Additionally, low concentration temporal and spatial variations in these pollutants provide valuable information for source apportionment. For example, in Vancouver, BC, sub-hourly variations in SO_2 concentration at and below 10 ppb proved to be informative in identifying the impacts of marine vessel emissions on urban AQ (Lu et al., 2006). Research grade instrumentation, which tend to be highly specialized and challenging to operate, are necessary when concentrations are very low and/or when high time resolution is required (e.g., such as in the free troposphere or remote areas). These are costly and unlikely to be suitable for routine monitoring.

Proper measurement of 'true' NO_x or NO_2 is an important consideration. The standard chemiluminescence technique detects NO and then various pre-treatments of the air sample allow for NO_2 to be determined. However, 'off the shelf' use of most instrumentation will not correctly measure NO_2 due to interference from other forms of oxidized nitrogen such as nitric acid, particle nitrate and peroxyacetyl nitrate (PAN). Furthermore, unknown inlet losses for some of these species leads to additional uncertainty. The relative size of this interference increases further away from high NO emissions areas (e.g., large cities) and when the atmosphere is more photochemically active (i.e., summertime). Therefore, before reporting and using NO_2 concentrations the nature of this interference should be understood. This is particularly important as more human health studies draw attention to associations with NO_2 (e.g., Stieb et al., 2003; Burnett et al., 2004; WHO, 2004).

Another relatively distinct class of pollutants of importance to measure can be characterized as being associated with non-combustion surface or fugitive releases. This includes both those related to human activities and natural emissions. For many of these compounds current capabilities do not permit high frequency measurements. Agricultural emissions such as ammonia (NH_3) and methane (CH_4), as well as dust, pesticides (persistent organic pollutants - POPs) and bioaerosols fit in this class. Releases from a range of industrial processes and waste management are also important sources. In addition to the pollutants listed just above, reduced sulphur compounds (e.g., H_2S) best fit into this class of pollutants. Similarly, resuspended dust, which is typically in the coarse ($PM_{10-2.5}$ or PM_{coarse}) particle and giant particle (i.e., >PM_{10}) size ranges, best fits in this class. This includes road dust, wind blown dust from agricultural practices and from certain types of industrial facilities (e.g., mining and smelting).

Very few of these pollutants are measured routinely due to the lack of reliable, cost-effective techniques, because there are no air quality standards that necessitate monitoring and/or because they are only a problem in localized areas. For example, total reduced sulphur compounds (TRS) or H_2S can be serious issues in the vicinity of pulp mills, sour gas flaring and certain agricultural practices. In addition to country-specific air quality standards, international agreements often necessitate some level of monitoring. For example, POPs are routinely measured at several locations in support of the Stockholm Convention (http://www.pops.int/).

With respect to current AQ risk management issues, ammonia (NH_3) and coarse particles (PM_{coarse}) are considered to be the most important to monitor or otherwise gain more information on their levels and spatial and temporal variation. Ammonia plays a role in $PM_{2.5}$ formation and thus, improved knowledge of its behaviour and emissions is needed to manage $PM_{2.5}$. PM_{coarse} has traditionally been included as part of PM_{10} or TSP measurements. However, as the monitoring focus shifts to $PM_{2.5}$ the need to continue to manage the coarse particle fraction is becoming an independent issue. At present, data limitations hinder determination of the health risk posed by PM_{coarse} and assessment of the feasibility of managing its level, especially given the range of natural and anthropogenic sources that contribute.

Secondary compounds, which are distinguished by the fact that they form in the atmosphere, represent an important class of air pollutants to measure. A large fraction of the chemical constituents found on $PM_{2.5}$ (fine particles) are secondary. This includes sulphate, nitrate, ammonium and some organic species. In the gas phase, ozone (O_3) is the most well-known and commonly measured pollutant. However, there are several other secondary oxidants, acids or VOCs of importance, either because of their potential to have health or environmental effects or because of their role in atmospheric chemistry. Examples are hydrogen peroxide, the hydroxyl radical, PAN, nitric acid, nitrous acid, hydrochloric acid, formic acid, acetic acid, formaldehyde, acetaldehyde, 1-3 butadiene and acrolein. Some of these may be emitted directly (i.e., primary pollutant), but atmospheric formation is likely the most important source.

Although O_3, $PM_{2.5}$ and some specific secondary constituents of $PM_{2.5}$ (e.g., sulphate and nitrate) can be measured with relative ease, it is much more difficult for most of the other secondary compounds. The methods available are expensive, experimental and challenging to implement and some of the compounds of interest are very short-lived in the atmosphere. Yet their measurement can be very important for understanding atmospheric chemistry. Consequently, when possible, measurements are undertaken during detailed research studies. Even the more routine approaches for measurement, such as exist for compounds such as PAN and formaldehyde, requires highly trained operators and/or very capable analytical laboratories.

Volatile and semivolatile organic compounds (VOCs and SVOCs) are associated with both combustion and fugitive emissions as well as secondary formation and are thus part of all of the three classes above. SVOCs are also found in both gas and particle phase depending upon their properties and ambient conditions. However, due to the large number of compounds, their complexity, the challenging nature of their measurements and the tendency for many to be toxic, VOCs and SVOCs are considered here to be a separate class of air pollutants.

Total gas phase non-methane hydrocarbons are often measured with relatively simple instrumentation at routine monitoring sites. However, such measurements provide a limited amount of information since the compound(s) responsible for the higher concentrations (i.e., the dominant fraction of the total VOC) and/or the temporal variation will vary and cannot be discerned. In many cases the dominant species are not those of greatest interest from the standpoint of either atmospheric chemistry (i.e., O_3 or $PM_{2.5}$ formation) or toxicity. In addition, inlet losses, which are not necessarily the same for all the VOC and SVOC species, adds uncertainty to the measurement. It is important to distinguish between CH_4 and the remaining VOCs because the former is often found in much higher concentrations and behaves differently in the atmosphere (e.g., has a much longer lifetime).

Measurement of individual VOC and SVOC compounds is necessary to provide insight into their contributions to O_3 and $PM_{2.5}$ formation. In addition, they are useful for source apportionment and in order to characterize 'hot spots' of high exposure to toxics. Air samples can be collected in the field and analyzed with a good degree of accuracy and precision for the lower molecular weight compounds (i.e., fewer than about 10 carbon atoms). Larger molecules and the more-oxygenated species are more difficult to measure with confidence. In all cases, VOC and SVOC measurement is a labour-intensive and expensive. Although there are commercially-available systems capable of time-resolved, real-time measurement of some compounds, these require highly trained operators and/or very capable analytical laboratories and careful consideration of the uses of the data and their subsequent storage is necessary.

Particle Composition and Size

Similar to VOCs, particulate matter (PM) consists of many different chemical species. These can be separated into three main classes. Carbonaceous compounds, inorganic ions and trace metals. The main contributors to $PM_{2.5}$ mass are organic carbon compounds (40-60%), sulphate (20-50%), nitrate (0-50%) and ammonium (5-15%). The latter three have been successfully measured in many locations and countries via filter sampling and laboratory analysis. Measurement of the carbonaceous material is more uncertain. Both the organic fraction (OC) and the elemental fraction (EC or BC for "black carbon") can be measured relatively simply from filter samples, but the distinction between them is operationally defined and method specific. A significant amount of difference exists among the different approaches and or among different laboratories (Schmitt et al., 199x). In addition, sample collection artifacts caused by SVOCs and relatively high blank filter concentrations for OC lead to considerable uncertainty (Fan et al., 2003).

A large amount of information on $PM_{2.5}$ mass measurement and $PM_{2.5}$ sampling and chemical analysis is provided by Chow (199?). Since that publication there have been significant advances in semi-continuous measurement of nitrate and sulphate, OC and EC. These measurements, although more challenging than semi-continuous mass measurement and than the traditional approach of using filters and laboratory methods, are providing new insights in the sources of $PM_{2.5}$. Some discussion on the various technologies being developed and applied is included in the NARSTO Assessment (McMurray, Shepherd and Vickery, 2004) and an overview of some of the new insights these instruments have enabled can be found in Wexler et al. (2006).

Organic carbon is one of the most challenging aspects of $PM_{2.5}$ measurement. As indicated above, sampling leads to uncertainties. The other main difficulty is that not all of the specific chemical compounds contributing to the total OC are known. In general, only about 20% of the OC can be consistently identified. These are chemical species such as polycyclic aromatic hydrocarbons (PAHs), alkanes and a large variety of organic acids. There is a considerable amount of ongoing research on the chemical speciation of specific organic compounds and much more data, although far from routine, now exist. There is a growing body of evidence of particle surface chemistry contributing to the uptake of organic mass and altering the chemical nature of the compounds present. Such processes need to be understood much better, as does the emissions of OC and the contribution for natural sources (primary and secondary OC) before OC can be understood sufficiently to model specific control strategies. Thus, organic material on $PM_{2.5}$ currently represents one of the greatest and most-important challenges to the scientific community.

To fully characterize PM, information on size distribution (i.e., number per size range) is necessary. The measurement of ultrafine particles (UFP) is based upon the total number of particle counts (per cubic centimeter) for all sizes below 0.1 μm. In terms of particle number, those below this size (i.e., UFP) completely overwhelm the remaining counts above this size (i.e., for all the rest of the particles from 0.1 to ~100 μm). Consequently, a condensation particle counter (CPC) operated with no specific size separation at the inlet essentially measures UFP. Much greater variability in the performance of an individual UFP measurement is caused by the smallest particle size that can be efficiently counted by the CPC because under many atmospheric conditions the number of particles increases as size decreases. The smallest size varies between 0.005 μm (5 nanometers) and 0.02 μm (20 nanometers), depending upon the model (and price) of CPC and even individual characteristics of each CPC. Thus, it is important to know the smallest detectable size and corresponding count efficiency for each CPC utilized for UFP measurements. One minute or better time resolution is possible when CPCs are used for UFP measurement. Electrostatic classifiers operated upstream from the CPC yields information on the numbers of particles within many size ranges (i.e., particle size distribution). Typical systems (scanning mobility particle sizers – SMPS) can discriminate amongst 32 or more size ranges or bins from 0.01 μm to 0.3 μm and provide a size distribution measurement every 15 minutes or better.

3.3.2.2 Siting criteria and where to measure

Location of measurement and the immediate surroundings have a large impact upon the concentrations observed. Standardization is important to consider for monitoring networks and several have established criteria (CAPMoN, EPA, siting criteria). In addition, the regulations may dictate the type of sites to be used for monitoring. For field studies the measurement locations are selected to achieve more-specific objectives, ranging from model evaluation and development to studying long range transport to exposure characterization.

On a broad geographic scale sites can be classified as either being remote, regional, urban background, urban exposure hot spot or industrial. The spatial scale or 'footprint' that each type of site can represent clearly varies from being nearly continental for remote sites to less than neighbourhood scale for industrial sites.

Sites supporting air quality risk management typically fall within the urban exposure hot spot to regional scale. The latter type of site is best situated to provide information on transboundary transport as well as the general conditions in the rural area surrounding, but upwind of nearby urban areas (i.e., what is blowing into our cities from sources upwind). Ideally, such a site will be representative of at least a 20,000 km^2 area. The former type of site is best typified by measurements in high traffic areas (e.g., kerbside locations that are common in European cities). These urban exposure hot spot sites should be representative of conditions that the general population is exposed to on a regular basis and/or of the conditions over a neighbourhood experiencing higher concentrations due to the amount or type of emissions in the vicinity. Environmental justice issues are leading to an increase in the interest in studying high exposure neighbourhoods. With the exception of industrial sites, the specific location of measurement should not be directly impacted upon by local emissions or by nearby obstructions to wind flow. The definition of local varies from the urban background to the regional sites ranging from about 2 km to 25 km, respectively. Obstructions could be adjacent

buildings in an urban area or the edge of a forest clearing at a rural or regional site.

The height of measurement is also an important siting criteria. Closer to the surface the measurements can be impacted upon by local dry deposition or local surface emissions or resuspension of dust. At regional and remote locations, 10 m is often the standard height as long as there are no nearby (~50 m) obstructions. In urban areas the concept of breathing zone enters into consideration, but there is a much greater risk that a very local emission will influence such a low measurement. Furthermore, low nighttime mixing heights tend to enhance this effect. Breathing zone measurements may be desired for specific exposure field studies, but are less likely to be representative of the urban background and, as well, not likely of the conditions occurring within a hot spot. Alternatively, in cities higher measurement heights such as rooftop locations tend to provide a better indication of the urban background and its temporal variation (Brook et al., 1999). Rooftops offer additional security for the equipment because they are difficult to access. This raises a key point about finding secure, accessible sites with power, communications and a high likelihood of long term stability, especially in urban areas and given budget limitations. The point is that compromise is sometimes necessary when selecting a site because few locations are "perfect" and opportunity to have access to a reasonable location may be difficult to find given time constraints.

Existing measurements and/or past experiences can provide considerable insights in selecting measurement sites or designing networks. Air quality models can provide guidance and are also valuable for data analysis and for expanding spatial detail (i.e., conditions between sites). As many of these sources of information as possible should be used to optimize and enhance sampling strategies.

3.3.2.3 Quality Assurance

Documentation on the quality assurance (QA) measures and expectations that are or will be followed throughout a measurement program helps ensure the data's value and that they are not misinterpreted by other users. It is entirely the responsibility of the data generator(s) to initiate and uphold this plan and to make users of the data aware of the QA details. Providers of the funding for measurements should demand that evidence of a QA plan be available before a program proceeds. Conversely, QA details and other data limitations are important for data users to understand to ensure that correct conclusions are drawn.

Measurement methods and types and model numbers of the instrumentation used should be recorded, as well as any changes during the program. Details of the measurement site, such as latitude, longitude, elevation (above MSL), inlet height and design and proximate emissions and obstructions (photos), are also necessary to document. This is referred to as site "meta data". Most commercial instruments have known detection limits and levels of precision and accuracy as well as information on interferences. Nonetheless, precision and accuracy targets for the measurements need to be quantified and the actual values being achieved should be determined routinely to ensure a failing piece of equipment is identified and replaced quickly to avoid data loss. Duplicate or repeated measurements of the same samples and analysis of standards with known, traceable concentrations are, therefore, critical to undertake routinely to track precision and accuracy, respectively. The amount of resources needed for a measurement program is impacted upon by the frequency of QA measurements (e.g., number of site visits, amount of standard gases used), but, as stated above, this aspect should not be underappreciated. On the other hand, when QA measurements are being made (e.g., 'zero' or 'span' readings) actual measurements are being missed and thus the appropriate balance needs to be established.

AQ measurements that involve sample collection in the field followed by chemical or gravimetric analysis in the laboratory require QA in both the field and the lab. With respect to the field, one of the most important QA measures is the collection of field blanks. Whatever the approach to capturing the sample (e.g., filters, denuders, canisters, passive sorbants, traps or cartridges), at least 10% of the samples analyzed in the lab should be field blanks. These procedures should be clearly stated in the QA plan and the resulting data need to be rigorously analyzed to ensure data quality objectives are upheld and/or adjusted.

QA measurements should be based upon concentration levels that are typical for the site and that are within the measurement range of interest to the program. QA samples should also be introduced into the instrument or the analytical procedure (i.e., for laboratory analysis) in a manner that mimics the real measurement process and the conditions during measurement as much as possible. In general, the more experimental the measurement method the more challenging the QA and more frequent QA checks are usually necessary. As discussed in the NARSTO Assessment (McMurray, Shepherd and Vickery, 2004), for some measurements appropriate standards are not available. In this case, method intercomparison can provide information to judge the level of confidence in the data. With respect to current AQ health issues of concern, BC and UFP measurements are hindered by the lack of standards.

3.3.2.4 Data archiving and reporting

Long term storage of final measurement data, including the QA data and site meta-data, is of utmost importance. At the same time data accessibility throughout the future needs to be simplified and rules for providing the data to all users also need to be considered. Whenever possible, data should be archived at their native temporal resolution since averaging to longer time periods will likely lead to unrecoverable loss of information that may be of value in the future. Given the present costs of storage media (hard drives, DVDs, etc.) archiving one-minute data is no longer an issue.

A key issue with AQ measurement programs is 'data turn-around time'. Objectives for this and rules for how other users can or cannot report the data need to be established. Faster turn around can be expected to increase the value of the data assuming that the more current the information, the greater the number of interested users and the greater the impact of publicizing what is being observed. There is an increasing demand for real-time data and data products reported via websites and the media (see Section 3.2.3.4 below). As indicated above, this is possible for a growing number of pollutants because of improved instrumentation, automated QA and communications. Continued improvements in these areas can be expected to reduce subsequent QA work and to increase data usage and publicity.

There are a number of recognized national or international data archives or portals for data access. Each has their own criteria for accepting and then documenting and preparing or formatting the data for storage and exchange. Examples of national archives of standard monitoring data are the National Air Pollutant Surveillance Network (NAPS) maintained by Environment Canada's Environmental Technology Centre (http://www.etcentre.org/NAPS/index_e.html), the Air Quality System (AQS), which is the USEPA's repository of ambient air quality data (http://www.epa.gov/ttn/airs/airsaqs/), **Instituto Nacional de Ecología**'s archive of Mexico's air quality data (http://www.ine.gob.mx/) and the Air Quality Archive (AQA) for data for the United Kingdom (http://www.airquality.co.uk/archive/data_and_statistics_home.php). Data from many European countries are also available from AirBase, which is under the European Topic Centre on Air and Climate Change (http://etc-acc.eionet.europa.eu/databases/airbase/airbasexml/index_html).

A growing amount of North American AQ research data, are being kept in the NARSTO archive. These data are available from the NASA Langley Atmospheric Science Data Center as ASCII data files in the NARSTO Data Exchange Standard (DES) format. This format is described on the NARSTO Quality Systems Science Center website (http://cdiac.ornl.gov/programs/NARSTO/). Another common format in which data are provided is 'NASA Ames' (http://cloud1.arc.nasa.gov/solve/archiv/archive.tutorial.html). Similar to NARSTO, the header section of a file contains important metadata, including instrument type, instrument name, data resolution, and units. The Joint Research Centre of the European Commission uses the NASA Ames Data Exchange Standard (http://airispra.jrc.it/Start.cfm). The netCDF (network Common Data Form) library also defines a machine-independent format for representing scientific data (http://www.unidata.ucar.edu/software/netcdf/). Together, the interface, library, and format support the creation, access, and sharing of scientific data.

3.3.2.5 Consistency

Proper QA and data archiving significantly increases the likelihood that measurement data are consistent among countries. Utilization of instrumentation that has been approved by national or international standards organizations (e.g., USEPA reference or equivalent methods, NIST) also helps ensure consistency. This is critical for successful AQ management and thus, the more there can be international consensus on QA requirements and data storage and sharing protocols to better. Clearly, due to individual needs of each country and the fact that resources for measurements within a country will, by in large, be provided by that country implies that there will be differences. The people implementing the measurement programs will differ among countries. Establishing a small number of internationally supported, master stations where a wide range of ongoing measurement comparisons can be undertaken will provide valuable insight regarding consistency.

3.3.3 Conclusions

Air quality measurements are essential for public health protection. They define the problem to be managed (there is no problem until a measurement is made) and are essential to track progress, develop predictive models and devise risk management strategies.

Considerable knowledge exists today to develop efficient air quality management measurement programs, in conjunction with emissions data and air quality models, that do not involve extensive numbers of sites and yet provide a large amount of information to support multiple objectives.

Measurements are the basis for determining the current level of health risk a given population is experiencing and consequently for prioritizing the need for reductions. They are critical for evaluating the effectiveness of AQ management strategies and altering such strategies if the desired outcomes are not being achieved.

Measurements are used to develop exposure estimates for epidemiological studies aimed at quantifying concentration-response functions, which are at the heart of cost-benefit analyses.

When measurement programs are forward-looking, they can potentially provide new insights regarding additional air pollutants of concern and can support future epidemiological studies to uncover such new risks to the population.

Detailed analysis of measurement data can help target the most effective approaches to reduce ambient concentrations and, hopefully in the future the optimum (i.e., cost effective) approach to protect public health. Full understanding of particulate matter, in terms of impacts, formation processes and optimal control strategies remains the air pollutant requiring the greatest attention in terms of detailed measurement studies and data interpretation efforts.

Ultimately, the availability of air quality measurements dictates what can be studied to identify risks to human health posed by human and natural emissions of air pollutants. Thus, there is a continual need for forward-looking measurement programs in terms of the pollutants measured and the location and temporal resolution of such measurements. There is a need to develop air quality risk management methods that integrate multiple environmental issues, not just human health concerns. Long-term sustainability needs to be the main driver in such integrated assessments since an unsustainable socio-economic system, at the global level, poses the greatest risk to human health.

3.4 Air Quality Modeling for Risk Management

3.4.1 Introduction

As noted at the beginning of this chapter, a crucial component in understanding and managing atmospheric pollution is our ability to quantify the links between emissions of primary pollutants or precursors of secondary pollutants on the one hand and ambient pollutant concentrations and other physiologically, environmentally, and optically important properties on the other. AQ models provide this capability. Such models consist of mathematical representations of the relevant physical and chemical atmospheric processes that are then solved by means of numerical algorithms to obtain pollutant concentration fields as functions of space and time for a given set of pollutant emissions and meteorological conditions (e.g., Peters et al., 1995; Seinfeld and Pandis, 1998; Jacobson, 1999; Russell and Dennis, 2000; Reid et al., 2003). Figure 3.4.1 shows a schematic of the atmospheric "process web", including the numerous links and interconnections between different atmospheric components, that should be represented in an AQ model.

AQ models are also referred to by other names, including chemical transport models, long-range-transport models, emissions-based models, source-based models, source-oriented models, and source models. Depending on the particular set of atmospheric processes included in such models, they can be classified into various categories such as photochemical models, acid deposition models, and particulate-matter or aerosol models (e.g., Seigneur and Moran, 2004). All of these models, however, include a representation of some atmospheric *chemical* transformations along with representations of emissions, transport, diffusion, and removal processes. The inclusion of chemistry typically requires consideration of time scales ranging from fractions of seconds to days in order to account for many important chemical reactions, and hence AQ model domains need to extend at least several hundreds of kilometers in the horizontal and up to at least the middle of the troposphere in the vertical for compatibility with the transport that can occur during a multiple-day simulation. Models of air pollutants that do not consider chemistry, on the other hand, are generally referred to as "dispersion models".

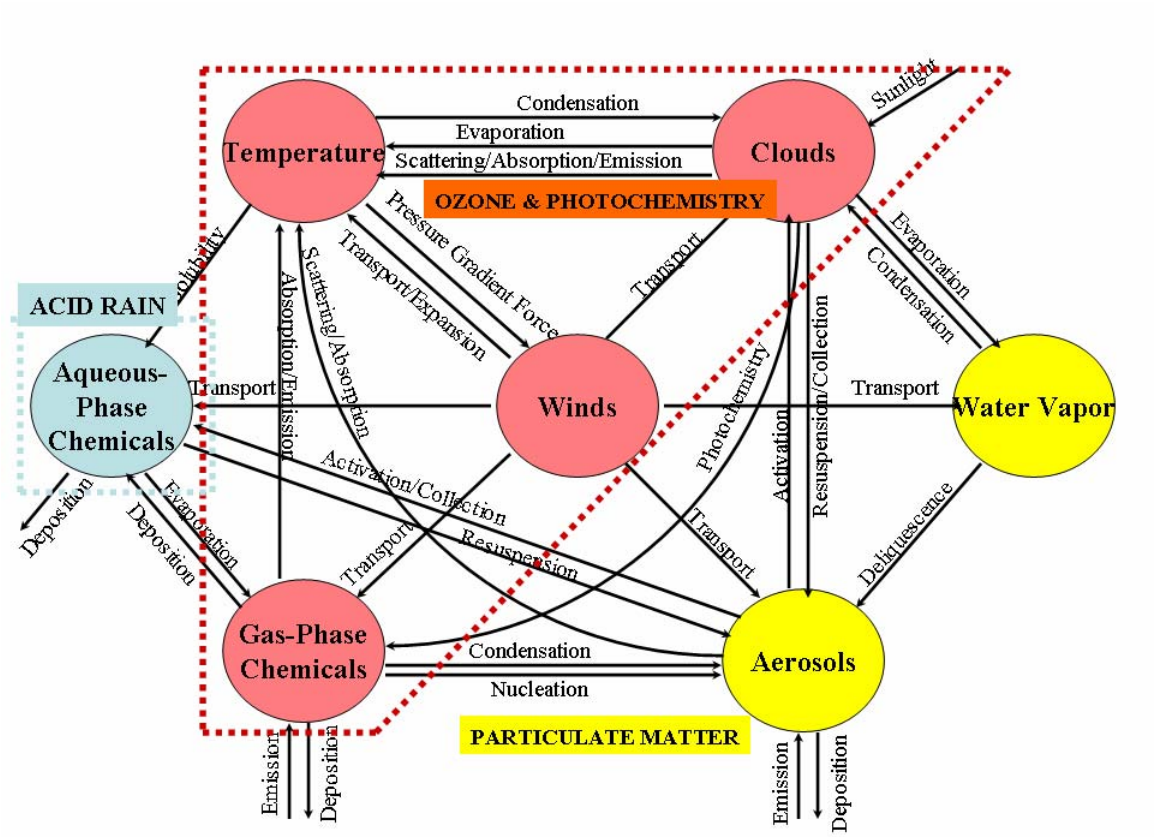

Figure 3.4.1. Schematic diagram of atmospheric physical and chemical components and their interactions (adapted from Peters et al., 1995).

Figure 3.4.2 shows a flowchart of the data flow required to apply an AQ model. In fact, as this figure makes clear, it is more accurate to refer to this as an AQ *modeling system* since the emissions files and meteorological files that are needed to drive an AQ model are provided by two other complex computer programs, namely an emissions processing system (e.g., Dickson and Oliver, 1991; Houyoux et al., 2000) and a numerical weather prediction model (e.g., Seaman, 2000). An emissions processing system in turn requires one or more emission inventories as its primary input plus such ancillary information as population, socioeconomic, and geophysical data. A numerical weather prediction model needs meteorological observations from a variety of observational platforms (e.g., surface instruments, rawinsondes, aircraft, satellites) as its primary inputs plus various geophysical data sets.

AQ modeling systems can be used to quantify source-receptor relationships for a wide range of air pollutants. They are also the only tool available to predict *future* air concentration and deposition patterns based on possible future emission levels. That is, they are prognostic in nature, unlike receptor models, which depend upon ambient measurements and hence are applicable only to periods for which measurements exist. For AQ models to be useful, however, there must already be information available about emissions and atmospheric measurements. If such data are not available for a region, then AQ model applications for that region can seldom provide much useful guidance for policymakers due to the much greater uncertainties associated with model predictions due to the uncertainties in model inputs. (The need for emissions data is obvious as illustrated by Figure 3.4.2. AQ measurement data are needed to specify chemical initial conditions and boundary conditions as well as to evaluate model performance.)

AQ models have been used for decades to support AQ management, but at the same time they have also undergone continued and rapid development. For example, the first meeting in the long-running NATO-CCMS series of international technical meetings on air pollution modeling was held in Eindhoven, The Netherlands in 1971. Most air-pollution models at that time were short-range, single-source dispersion models, and multiple-source models for modeling primary pollutants in urban settings were just being developed. Some of the earliest air-pollution models with parameterizations of chemistry were developed in the 1970s to simulate either the formation of photochemical smog in

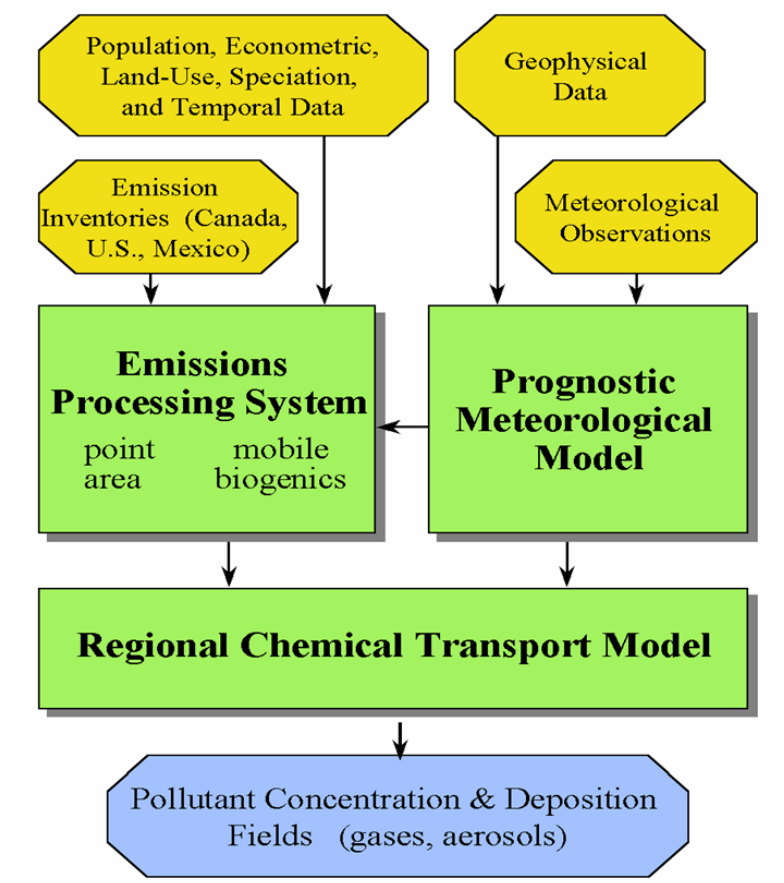

Figure 3.4.2. Schematic description of the components of an AQ modeling system (from Seigneur and Moran, 2004).

the Los Angeles basin or the long-range transport and transformation of air pollutants contributing to acid deposition in Europe and in North America. Given this considerable history, there have been a number of overviews of AQ models and AQ modeling over the years, including textbooks such as Jacobson (1999) and review articles such as Peters et al. (1995), Russell and Dennis (2000), Seigneur (2001), and Seigneur and Moran (2004).

The discussion in this section is not intended to provide a comprehensive review of AQ models and modeling practices. Instead, it builds upon the earlier NERAM paper on AQ modeling by Reid et al. (2003) and focuses on AQ model capabilities and uncertainties and on the management of these uncertainties in AQ model applications. Section 3.4.2 summarizes the variety of ways in which AQ models can contribute to AQ risk management. Section 3.4.3 reviews key technical choices and issues related to AQ model applications, especially those factors contributing to model uncertainty. Next, Section 3.4.4 provides an overview of "best practices" for using AQ models and their results and for managing associated uncertainties. Finally, Section 3.4.5 presents conclusions and recommendations.

3.4.2 Applications of models for AQ risk management

AQ models can be applied in a number of ways, both directly and indirectly, to support AQ management and policy formulation:

- evaluation of impact of emissions changes, including proposed control measures;
- source apportionment and source attribution;
- input to conceptual model development;
- emission inventory evaluation;
- measurement network and field experiment design;
- AQ forecasting;
- testing current understanding of science.

Let us consider each of these applications in turn.

Evaluation of impact of emissions changes. The use of AQ models to assess the impact on air quality of emission changes due to pollutant abatement strategies, new pollution sources, population and economic growth, etc. has probably been the most common application of AQ models. Reid et al. (2003) gave three examples of such applications for Spain, Australia, and Canada, respectively: (a) the assessment of ozone abatement strategies for the Greater Madrid area (Palacios et al., 2002); (b) the impact of alternative urban forms of the city of Melbourne on urban air pollution levels (Manins et al., 1998); and (c) the sensitivity of PM concentrations in Ontario to changes in emissions of primary PM and PM precursor gases. Four more examples include (d) the generation of intra-European source-receptor matrices ("blame matrices") by individual country for oxidized sulphur, oxidized nitrogen, and reduced nitrogen species for 2003 emissions and projected 2010 emissions (EMEP, 2005), (e) the assessment of the impact of possible emission control strategies in the Pearl River Delta region of southern China (Streets et al., 2006), (f) the estimation of AQ benefits from implementation of SO_2 and NO_x emission reductions under the 2005 U.S. Clean Air Interstate Rule (U.S. EPA, 2005a), and (g) the evaluation of the potential impacts of proposed SO_2 and NO_x emission control measures in Canada and in the U.S. on acid deposition in Canada (Moran, 2005).

Many of these studies follow a similar approach. The AQ model is first run for a "base-case" simulation, for which the emissions used are either historical or current and for which AQ measurements are available with which to evaluate model performance, and then again for one or more emission "scenarios", in which the assumed emissions correspond to a possible future state. For the simplest type of emission scenario, a "roll-back" scenario, emissions of one or more species may be changed by a fixed percentage for all source types across either the entire model domain or a selected subdomain. In more realistic emission scenarios, selected source types such as on-road motor vehicles or coal-fired electrical generating stations may be targeted. Sometimes one of the scenarios corresponds to a "business-as-usual" (BAU) scenario, in which emissions from the base case have been projected forward in time to account for population and economic growth and the implementation of any scheduled control measures expected by the scenario year. A companion future-year emission scenario may then be run that is identical to the BAU scenario except for the addition of a new candidate abatement strategy. Comparison of the base case, the BAU future scenario, and the companion future scenario then allows the impact of the abatement strategy to be estimated as well as any changes expected in future-year AQ relative to current conditions if the abatement strategy were or were not implemented. As one example, Figure 3.4.3 shows predictions of effective acidity wet deposition from an acid deposition model for two cases: a 1989 base case and a realistic 2020 emissions scenario.

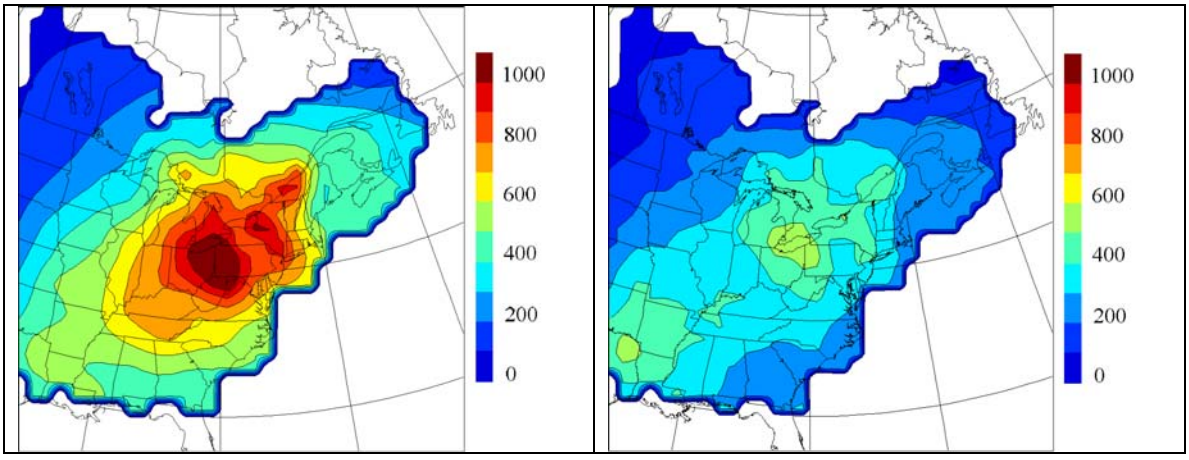

Figure 3.4.3. Plots of annual effective acidity wet deposition (units of eq/ha/yr) predicted by the ADOM acid deposition model for (left) 1989 base case and (right) 2020 SO_2 and NO_x emission scenario. Effective acidity is defined to be the sum of sulphate and nitrate wet deposition. See Moran (2005) for details.

Source apportionment and source attribution. AQ models can be used to investigate and quantify source-receptor relationships in a manner that is complementary to the use of receptor models, that is, based on a source-oriented framework rather than on a receptor-oriented framework (Blanchard, 1999). The simplest approach is the so-called "zero-out" approach, in which emissions from a particular source sector (e.g., petroleum refineries) or from a particular

jurisdiction (e.g., a city, a province/state, or a country) are set to zero while leaving emissions from other source sectors or jurisdictions unchanged. Predictions from this case can then be subtracted from predictions from a base run in which all emission sources are considered to estimate the impact of the targetted source sector or jurisdiction. Two more sophisticated approaches are "source tagging" and inverse methods. In the former, pollutant emissions from particular source sectors or geographic locations are tracked in the model as separate ("tagged") species (e.g., Kleinman, 1987; McHenry et al., 1992; Kleeman and Cass, 1999a,b; Zhang et al., 2005). In the latter, the adjoint of the AQ model can be constructed, used to quantify the sensitivity of the model to emission inputs, and then combined with ambient measurements, or else initial attribution results can be refined based on the synthesis inversion technique (e.g., Uliasz, 1993; Pudykiewicz, 1998; Mendoza-Dominguez and Russell, 2001; Mallet and Sportisse, 2005; Knipping et al., 2006).

Input to conceptual model development. An AQ conceptual model is a qualitative mental model for a geographic region that is based on a synthesis and simplification of available AQ information obtained from analysis of emissions, measurements, and AQ model results to distill the primary contributing factors, including key emission sources, terrain characteristics, and local weather and climate. Useful information from AQ models can include results from a suite of emissions scenarios, from source apportionment studies, and from sensitivity studies (see Section 3.4.3). One good example of the development of such a conceptual model is the analysis described by Pun and Seigneur (1999) for PM pollution in California's San Joaquin Valley. A number of the regional conceptual models for PM presented in the *2003 NARSTO PM Science Assessment* are based in part on AQ model results (NARSTO, 2003). One report (U.S. EPA, 2005b) gives a useful list of questions and points to consider in constructing a conceptual model of ozone pollution, including the relative contribution of local and distant sources, the role of certain weather patterns, and the nature of the local chemical regime. Answering some of these questions requires the application of AQ models. And in the paper by Zunckel et al. (2006), AQ model results contribute significantly to a conceptual model for surface ozone in southern Africa even with very limited availability of emissions data and AQ measurements.

Emission inventory evaluation. Since AQ model predictions depend directly on the input emission fields, the comparison of AQ model predictions with AQ measurements can give some indication of the accuracy of the input emissions. For example, if AQ model predictions are biased significantly high or low in a certain region as compared to measurements, one possible cause could be a corresponding high or low bias in the input emissions for that region. Or if measurements are stratified into rural and urban sites, then AQ model predictions for rural and urban locations can be checked independently to see if any systematic difference exists for rural vs. urban emissions (e.g., Yu et al., 2004). Inverse modeling analyses, in which enhanced AQ models are combined with ambient measurements, provide another, more quantitative, approach to estimating emissions strengths on either a regional or a global basis (e.g., Pudykiewicz, 1998; Mendoza-Dominguez and Russell, 2001; Martin et al., 2002; Palmer et al., 2003; Gilliland et al., 2006).

Measurement network and field experiment design. AQ models can also be employed as sophisticated interpolation schemes since their results are based on consistent and comprehensive representations of physical and chemical laws. For example, an analyst could use modelled pollutant fields as "true" patterns and then sample the model fields for different numbers of grid cells in order to investigate the impact of adding or removing stations to a network on the estimation of the actual pollutant spatial pattern from measurements, including the identification or non-identification of strong gradients and "hot spots". Similarly, scientists planning and designing a field experiment could use model predictions to help them choose measurement site locations, aircraft flight tracks, and so on.

AQ forecasting. When AQ models are used to provide policy guidance or to interpret field-experiment measurements, historical periods are typically considered. Such applications are referred to as retrospective runs or "hindcasts". If, however, an AQ model can be run in "real time" quickly enough (i.e., in a few hours), then it can provide an AQ forecast for the next day or two, that is, a prospective simulation. Both the Canadian and the U.S. national weather services now run AQ models in order to issue public regional AQ forecasts (e.g., www.msc-smc.ec.gc.ca/aq_smog/chronos_e.cfm and www.nws.noaa.gov/aq/) and the Australian national weather service issues AQ forecasts for two large urban areas, Melbourne and Sydney (Cope et al., 2004). Besides providing useful guidance to the public about next-day AQ, such forecast programs have the added benefits of (a) maintaining or raising public awareness about AQ and (b) challenging the AQ models with a broader range of weather conditions than they are typically subjected to in scenario modeling (e.g., photochemical pollutant scenarios are almost always summer cases). Performance evaluations for AQ forecasts can then provide additional insights into model skill and reliability and identify model weaknesses (e.g., Eder et al., 2006).

Testing current understanding of science. Finally, AQ models provide a means to represent and link in a single package our best understanding of all of the chemical and physical processes relevant to AQ. This knowledge synthesis can then be evaluated by comparing model predictions with enhanced measurement data sets obtained from dedicated AQ field campaigns such as SCAQS [1987], EMEFS [1988-90], NARE [1993], NARSTO-NE [1995], ESQUIF [1998-99],

BRAVO [1999], TexAQS [2000], ACE-ASIA [2001], ESCOMPTE [2001], Pacific 2001 [2001], TRACE-P [2001], and ICARTT [2004] (e.g., Dennis et al., 1993; Berkowitz et al., 1998; Heald et al., 2003, 2005; Frost et al., 2006; Hodzic et al., 2006; Pun et al., 2006; Smyth et al., 2006a). Conversely, an AQ model may also be used to help interpret field-campaign measurements, which can be difficult for a set of measurements of limited duration and restricted to a small number of locations due to the complexity of geography, meteorology, and interconnected chemical and physical processes. An AQ model can also be used as a test bed to test a new parameterization of a key chemical or physical process (e.g., Padro et al., 1993; Pierce et al., 1998). Such activities probe both our current scientific understanding and our representation of it in AQ models, often leading to improvements to both.

3.4.3 Key technical issues to consider in AQ modeling programs

Worldwide, there are a number of AQ modeling systems available, and each is typically composed of a set of large, complex computer programs. As a consequence, there are many choices to be made and issues to be considered by a modeller when using an AQ modeling system for any application. Such choices and issues, however, also need to be taken into account by users of model results when judging the robustness and reliability of those results. Consider the following.

Choice of model. Some race-track devotees offer the advice that there are "different horses for different courses". The same is true of AQ models. The first step in applying an AQ model is to define the questions that need to be answered, and then, if possible, identify or develop an appropriate conceptual model. Doing so should immediately narrow down the set of AQ models that might be used to answer the question. For example, a very detailed but computationally expensive AQ model might not be the best choice for performing a multi-year AQ simulation, if such is called for. Another consideration is that a model designed to address one AQ issue (e.g., photochemical smog) may not include representations of all of the processes necessary to address another issue (e.g., deposition of acidic species – see Figure 3.4.1). And a model designed for highly polluted atmospheres may not be appropriate to model a clean atmosphere and vice versa (e.g., regional atmospheric chemistry in source regions vs. background global chemistry).

Model configuration. There are many choices to be made in configuring (i.e., setting up) an AQ model run. These include (a) the location and (b) the size, in both the horizontal and vertical, of the model domain, (c) the map projection to be used, (d) the grid spacing in both the horizontal and vertical, (e) the integration time step, (f) the simulation period, including any required "spin-up" time (the time for atmospheric concentration fields to reach an equilibrium between emissions and removal processes), (g) the "refresh" rate (the length of time that the *meteorological* model will be run before being re-initialized using a new set of meteorological analyses), (h) the set of chemical species to be considered, (i) the choice (in some cases) of parameterizations to be used for different chemical and physical processes, (j) the specification of initial chemical conditions, (k) the treatment of chemical lateral and upper boundary conditions, and so on. Each choice has implications. For example, the use of large horizontal grid spacing may "average out" a suspected hot spot or not represent small-scale meteorological circulations forced by local terrain features. See U.S. EPA (2005b) for a discussion about the choice of horizontal grid spacing, Berge et al. (2001) for a discussion about the specification of chemical initial conditions, and Brost (1988) for a discussion about the specification of chemical lateral boundary conditions.

Science considerations. One key limitation of AQ models is in fact gaps in our scientific understanding of the pollutants of interest. For example, it is well known that the sources of much of the carbonaceous component of atmospheric $PM_{2.5}$ are not presently known in spite of the fact that this component typically contributes ~40-50% of total $PM_{2.5}$ mass. Another example is our limited understanding of nighttime NOx chemistry (e.g., Brown et al., 2006). A second limitation is the use in current AQ models of process parameterizations of limited fidelity to the real atmosphere. For example, Dabberdt et al. (2004) recently identified the need for improved treatments of the influence on AQ of (i) the planetary boundary layer and (ii) clouds and cloud processes. Another important consideration is the presence of nonlinear effects in the chemical reactions that generate some pollutants of interest. For example, the possibility of a nonlinear response in sulphate deposition to SO_2 emission reductions due to oxidant limitations was identified in the 1980s as a potential concern for managing acid deposition (e.g., Misra et al., 1989). Nonlinearities in ozone photochemistry are also well known (e.g., Seinfeld and Pandis, 1998), but PM chemistry possesses even more nonlinearities. For example, Meng et al. (1997) presented AQ model predictions for two simple ozone control scenarios run for a Los Angeles smog episode in which VOC emission reductions reduced ozone levels but caused increases in $PM_{2.5}$ mass. And West et al. (1999) presented AQ model results in which reductions in SO_2 emissions in eastern North America increased $PM_{2.5}$ concentrations due to so-called "nitrate substitution". Such nonlinear responses can further complicate the interpretation of model predictions and the formulation of possible emission control measures.

Model parameterization and algorithmic limitations. Even when AQ processes are well understood scientifically, they must still be represented mathematically in AQ models by so-called process parameterizations, and then the complex, coupled system of governing equations that comprise the AQ model must be solved numerically. Both steps have limitations and can introduce errors. For example, in many cases a number of different parameterizations have been developed to describe the same chemical or physical process, and these different parameterizations will produce different results (e.g., Kuhn et al., 1998; Zhang et al., 2000, 2001; Mallet and Sportisse, 2006). Typically, more sophisticated (and complex) parameterizations have a greater number of parameters and coefficients that must be specified, but measurements to do so may be scarce or lacking completely. This implies that while a more sophisticated scheme may have the potential to do a better job in describing a process, there are no guarantees that it will actually do a better job in practice. For example, some gas-phase chemistry mechanisms consider a few dozen species whereas others consider hundreds or even thousands of species. But besides reaction rates for all of the reactions that these species participate in, emissions must also be specified for each species as must a number of chemical and physical properties such as molecular diffusivity and Henry's Law constant that are needed to estimate dry and wet removal rates. Such physico-chemical data may not be available for every species (e.g., Zhang et al., 2002).

The numerical integration of the AQ model also introduces errors since it usually requires the solution of large coupled systems of both ordinary and partial differential equations. As discussed by Pielke (1984) and Jacobson (1999) among others, finite-difference methods are usually employed in both time and space. Any time-stepping scheme used to integrate the AQ model in time will have truncation errors that depend upon both the order of the scheme and the chosen time step. Operator splitting is usually employed on the right-hand side of the governing equations to allow each process parameterization to be calculated separately, but operator splitting also introduces errors that depend upon the order of the splitting and the overall time step. Advection is well known to be a difficult process to solve, and literally hundreds of numerical schemes have been developed for advection. All suffer to varying degrees from some or all of truncation errors, numerical diffusion, phase errors, lack of positive definiteness, and violation of mass conservation.

Model "resolution" is another important consideration. The choice of a discrete model time step and grid-cell size implicitly imposes numerical filtering on the model solution. In essence, no temporal feature shorter than $2\Delta t$ and no spatial feature smaller than $2\Delta x$ can be predicted by the model, and $4\Delta t$ and $4\Delta x$ are probably a more realistic threshold (e.g., Pielke, 1984; Grasso, 2000). This has important implications for processes operating at smaller temporal and spatial scales (e.g., Uliasz and Pielke, 1998). As a consequence, many parameterization schemes to represent the influence of subgrid-scale processes at grid scale have been developed. One obvious example is the representation of point source emissions. In any Eulerian (grid) model, all or most point sources will be represented as volume sources since the emissions are assumed to be well-mixed throughout at least one grid cell, thus introducing large numerical (i.e., artificial) diffusion in the vicinity of major point sources. To address this problem, which will be most pronounced for isolated sources, so-called plume-in-grid schemes have been developed to represent near-source diffusion and chemistry in plumes from major point sources more realistically. The treatment of vertical diffusion in any AQ model is also intrinsically a subgrid-scale parameterization since it must represent the impact of a spectrum of atmospheric motions that cannot be resolved by the model. Cumulus parameterizations that represent the impact of unresolved clouds on model fields are another important class of subgrid-scale parameterizations that are employed in weather and AQ models (e.g., Haltiner and Williams, 1980).

Input data. AQ models require a number of input data sets in order to run. First are emission input files. Emission rates of a number of gaseous and particulate species must be specified for each model time step at each model grid cell at all levels. As discussed in Section 3.2, there are significant uncertainties associated with such emission files due both to errors in the emission inventories themselves and to additional uncertainties introduced by the emissions processing systems that perform the speciation and spatial and temporal disaggregation steps needed to create model-ready emission files (e.g., Hogrefe et al., 2003). For large point sources, ancillary information about smokestack characteristics such as stack height, stack diameter, stack-gas exit velocity, and stack-gas exit temperature is also needed. And if a future-year scenario is being considered, current emission inventories must be manipulated and modified to account for all assumptions built into the scenario.

Second, meteorological input files are needed if an "off-line" AQ model is being used (i.e., meteorological model and AQ model are separate), which is the case for most current AQ modeling systems. Meteorology is very important as it influences every aspect of the AQ system, including (a) natural sources of PM such as wind-blown dust and sea salt, (b) plume rise, (c) transport and diffusion, (d) gas-phase and heterogeneous-phase chemistry (via temperature and humidity effects), (e) cloud shading, (f) aqueous-phase chemistry, (g) dry removal, and (h) wet removal. Third, chemical initial conditions must be supplied for all model species for every grid cell, chemical upper boundary conditions must specified for all model species at the top model level, and, for a limited-area AQ model, chemical lateral boundary conditions must also be specified for all model species. And fourth, a number of geophysical fields may also be required, including

terrain height, land-use type, vegetation type, aerodynamic surface roughness, albedo, sea surface temperature, and soil texture. The accuracy and representativeness of all of these input files are a key concern, since even for a perfect model, the well-known aphorism "garbage in, garbage out" still holds.

Infrastructure considerations. As discussed by Reid et al. (2003), AQ modeling is typically resource-intensive in terms of model input data, people, calendar time, and computer power. In order to apply an AQ model for a particular case, the input data sets described above must be prepared for the model configuration selected, including emissions, meteorological, and geophysical files, the model must be run, and then the model results must be processed, analyzed, and interpreted. Typically, a minimum of three highly-trained modellers would be required to contribute, namely an emissions-processing specialist, a meteorological-modeling specialist, and an AQ-modeling specialist. The required calendar time from start to finish, including configuring and testing the model for the application, would likely be a minimum of weeks but more likely months. The minimum computer resources needed would be a high-end PC with multiple processors, large internal memory and disk space, and off-line archiving hardware to save numerous large model output files. Access to emissions data, meteorological data, geophysical data, and AQ measurement data is of course assumed as well.

Model accuracy, sensitivity, and uncertainty. For an AQ model's predictions to be used by policymakers, the model and its predictions should be credible. To be credible, the model should give the right answers for the right reasons. The determination of whether a model is giving the right answers is addressed by model performance evaluations, in which model predictions are compared to measurements. Model performance evaluation is discussed in more detail in the next section. However, there are some fundamental issues related to model accuracy, sensitivity, and uncertainty that need to be kept in mind. For one thing, how is the "right answer" determined? For another, how can model uncertainty be determined?

There are actually a surprising number of issues that arise in comparing AQ model predictions with ambient measurements. The biggest one is incommensurability, which arises due to the fact that model predictions correspond to grid-volume averages whereas measurements are typically made at points in space or along lines (e.g., aircraft flight tracks, DIAL). For example, for a regional-scale AQ model whose smallest grid volume is 20 km by 20 km by 50 meter, how representative would a single point measurement be of the 20 km^3 of air contained in that grid volume? One effort to address this question was made during the 1988-90 EMEFS field experiment in eastern North America. Surface measurements were taken of 24-hour SO_2, particulate SO_4, total NO_3 and hourly O_3 concentrations for two years at six clusters of 3 to 5 stations that fell within the confines of model 80-km by 80-km horizontal grid cells. Cluster or subgrid daily variability was found to be approximately linearly related to mean concentration, with the largest variability associated with SO_2 and minimum O_3 ($\sim \pm 70\%$), intermediate variability associated with p-SO_4 ($\sim \pm 30\%$) and t-NO_3 ($\sim \pm 40\%$), and the smallest variability associated with maximum O_3 ($\sim \pm 20\%$) (Seilkop, 1995a,b). Significantly, uncertainties due to this subgrid-scale variability overwhelmed uncertainties associated with instrument error. McNair et al. (1996) performed a somewhat related analysis in the Los Angeles basin for O_3, NO_2, and CO for two 1987 SCAQS cases. For circular areas with 25-km radius, smaller than those considered during EMEFS, they found that local inhomogeneities for these three species had normalized gross errors in the 25-45% range.

A second issue related to the comparison of model predictions and ambient measurements is the need to compare "apples with apples". For example, for gas-phase species, AQ model predictions correspond to ambient conditions whereas some networks report measurements at STP. For PM comparisons, model PM predictions are calculated based on Stokes diameter whereas PM measurements are reported using aerodynamic diameter, PM measurements unlike models can suffer from artifacts related to volatile species such as nitrate, some organic compounds, and aerosol-bound water, and the distinction between elemental carbon (EC) and organic carbon (OC) is analysis-method-based and can vary from network to network (Seigneur and Moran, 2004). Note that the *definition* of the EC and OC variables predicted by an AQ model also depends on the EC-OC analysis method used to speciate primary PM emissions.

Turning to model uncertainty, the discussions and conclusions of a 1982 workshop on AQ model uncertainty have been described by Fox (1984) and Venkatram (1988). The latter identified three main sources of model uncertainty as "(1) errors in model inputs, (2) errors in model formulation, and (3) inherent uncertainty associated with the stochastic nature of turbulence". The last source constitutes a lower limit on model uncertainty since it cannot be reduced even if all model-related errors are corrected. One aspect of this inherent uncertainty is related to the time and space averaging used in measurements vs. the ensemble averaging that is used to describe atmospheric turbulence. That is, atmospheric measurements correspond to samples from a single flow realization whereas AQ model parameterizations related to diffusion and mixing are based on ensemble averages for (theoretically) an infinite number of flow realizations with identical external conditions (e.g., Moran, 2000).

Reid et al. (2003) have noted that it is not possible to quantify overall model uncertainty because it is dependent on so many factors, some of them dependent on the particular application being considered, but also on the *interactions* of these factors. As already discussed, these contributing factors include errors and uncertainties in input data such as emissions, meteorology, and boundary conditions, uncertainties in our scientific understanding and in process parameterizations, errors associated with numerical methods, and uncertainties associated with required parameters like reaction rates. It is, however, possible to quantify some individual sources of uncertainty, particularly for numerical methods, to identify model sensitivity to various inputs and parameters, and finally to compare results from parameterizations and even entire models in order to try to characterize the range of uncertainty.

Error characterization is generally reported as part of the description of new numerical methods and parameterization techniques. A wide range of sensitivity analysis techniques exist, including DDM, ADIFOR, FAST, variational techniques, perturbation theory techniques, Green's function techniques, and stochastic techniques, that can be used to understand which model parameters and input variables most influence selected model outputs (see Zhang et al. (2005) for a useful literature review). Besides being compared side by side outside of models, the impact of different parameterization schemes can also be compared when embedded in a host model (e.g., Padro et al., 1993; Mallet and Sportisse, 2006). And some studies have compared differences in AQ modeling system results due to the use of different component models. For example, Hogrefe et al. (2003) compared the impact of using emissions files constructed by two different emissions processing systems from an identical emission inventory on the predictions of one AQ model. They found differences on the order of ±20 ppb in predicted daily maximum 1-hour ozone concentration. Another source of uncertainty is meteorological inputs. Smyth et al. (2006b) compared the outputs from one emissions processing system and one regional PM model for two sets of meteorological input files for the same period that were provided by two different meteorological models. An operational evaluation of the two meteorological models suggested that their performance was essentially equivalent, as was the performance of the AQ model for the two sets of meteorological files, but when grid cells were matched for the same time, large variability was observed, particularly in aerosol quantities influenced by relative humidity. And recently, the performance of seven AQ models in predicting ozone was compared for the same period (summer 2004) and region (eastern U.S.). The range of model predictions generally bracketed the measurements, and interestingly none of the models individually could match the skill of a weighted average of the seven forecasts (McKeen et al., 2005).

One other approach to assessing uncertainty is to synthesize expert opinion. Seigneur and Moran (2004) prepared a table that presented qualitative ratings of PM modellers's level of confidence in major aspects of the predictions of current PM AQ models. Only a few model aspects (SO_2, NO_x, and p-SO_4 air concentrations) were judged to have a "high" level of confidence. Most aspects were assigned "medium" or "low" ratings, and a few aspects, such as secondary OC and PM ultrafine mass and number concentrations, were assigned "very low" ratings. These ratings were based on an assessment of all contributing uncertainties, including uncertainties associated with the emissions of different pollutants and with scientific understanding.

3.4.4 Review of best practice for using models for AQ management

Let us now build upon the previous sections and consider that wry but wise epigram by Box (1979): "All models are wrong, but some are useful". That is, in applying models for AQ management, we must accept from the start that no model is perfect. Instead, as discussed in the previous section, AQ model predictions will be affected by numerous sources of error and uncertainty. How then can we account for the resulting uncertainty and apply models in a reasonable and defensible way in order to inform AQ management?

To start, how can we judge whether an AQ model will in fact be useful? For a model to be useful, presumably it must be credible. That is, it must have demonstrated sufficient skill and reliability that its predictions can be used with some confidence by analysts and policymakers in the formulation of AQ management strategies. Confidence can in turn be built in two ways: first, through *model verification* to assess the consistency, completeness, and correctness of the model and through *model performance evaluations* to characterize its performance and quantify its errors; and second, by applying the model in as appropriate, transparent, and defensible a manner as possible for the AQ issues being considered.

Model verification and model evaluation. Model verification and model performance evaluations should always be a required step before a model is applied in the policy arena. According to Fox (1981) and Russell and Dennis (2000), model verification is an assessment of the accuracy, reality, or truth of a model. It does not require a model to be run. Rather, model verification is a "desk check" in which the consistency, completeness, and correctness of a model's design, science, process representations, algorithms and numerical methods, inputs, and source code are examined and

assessed. Peer reviewers should be involved in such an examination, and, ideally, any interested party should have unrestricted access to the model source code for this purpose.

Model performance evaluation is the process of examining and appraising model performance through comparison of model predictions with measured AQ data and/or predictions from other models (e.g., Fox, 1981; Dennis et al., 1990; Russell and Dennis, 2003). There are four main types of model performance evaluation: (i) operational; (ii) diagnostic; (iii) mechanistic; and (iv) comparative (Seigneur and Moran, 2004).

- An *operational* evaluation requires the statistical evaluation of model predictions of a few key pollutants of interest with atmospheric measurements over time and space scales consistent with the intended applications of the model. An operational evaluation is intended to answer the basic question: "Are we getting the right answers?" Examples of operational evaluations include EMEP (2003), Eder and Yu (2006), and Eder et al. (2006). A paper by Fox (1981) reviews a wide range of statistical measures that have been used in operational evaluations, but a recent U.S. EPA report (U.S. EPA, 2005b; Section 15.2) recommends a small number of statistical measures that have been found to be representative and useful in evaluating the performance of photochemical AQ models.
- A *diagnostic* evaluation is more of a research-level evaluation and involves an examination of model performance at the process level for all relevant species. A diagnostic evaluation addresses the basic question: "Are we getting the right answers for the right reasons?" Because diagnostic evaluations are more wide-ranging and comprehensive than operational evaluations and generally make use of non-routine measurement data sets such as those from specialized field campaigns, they can identify the presence of compensating errors or excessive "tuning". Examples of diagnostic evaluations include Dennis et al. (1993), Karamchandani and Venkatram (1992), Sillman et al. (1998), Hogrefe et al. (2001a,b) and Biswas et al. (2001), Heald et al. (2005), and Yu et al. (2005). Also, Seigneur et al. (2000) have described how to optimize the design of field studies that will be used in the evaluation of PM AQ models, a recent U.S. EPA report (U.S. EPA, 2005b; Section 15.3) lists some diagnostic analyses that have been found useful in assessing the ability of photochemical AQ models to predict changes in ozone due to changes in emissions of ozone precursors, and a recent paper by Zhang et al. (2005) examines three diagnostic probing tools that have been used to examine photochemical AQ model performance.
- A *mechanistic* evaluation involves testing individual model components (i.e., process representations) in isolation against field or laboratory measurement. Such evaluations address the question: "Are we using good parameterizations?" Some examples of mechanistic evaluations include Pleim and Xiu (1995), Odum et al. (1996), Geron et al. (1997), and Zhang et al. (2001).
- Finally, a *comparative* evaluation involves a side-by-side comparison with another model or model component for identical or similar inputs. A comparative evaluation addresses the basic question: "Are we getting comparable answers from comparable models?" Examples of comparative evaluations include Alapaty et al. (1997), Hass et al. (1997), Kuhn et al., 1998; Ansari and Pandis (1999), Zhang et al. (2000), Hogrefe et al. (2001a,b), and McKeen et al. (2005).

Note that the term "model evaluation" denotes a process rather than an outcome or conclusion. For the terms "model verification" and "model validation", on the other hand, Oreskes et al. (1994) argued that numerical models of natural systems can never truly be verified or validated, since these terms imply the absolute correctness of a model. Fox (1981) and Russell and Dennis (2000) were careful to restrict their definitions of these terms. Model verification, as described above, refers to an examination process that at best leads to a provisional conclusion. And model validation is a process leading to a judgement on the quality, suitability, and usefulness of a model for a particular application that should be based on evidence from both model verification and multiple model performance evaluations. Such a judgment, however, must always be viewed as provisional, since additional information such as the results of a new evaluation may change the balance of evidence.

It is also important to consider which aspects of model performance need to be evaluated. Most AQ model evaluations involve case studies in which a model is run for a particular period using input emissions and meteorology suitable for that period and then model performance is examined using measurements from that same period. However, as already discussed, the most common AQ model application is to evaluate the impact of emissions *changes* on AQ. The key aspect of model performance in this instance is how well the model predicts the atmospheric *response* to the change in input emissions, and the approach to the corresponding performance evaluation is necessarily somewhat different. For a direct evaluation of model response, AQ measurements are required for *two* different periods so that an atmospheric response can be calculated, which means that the AQ model must be run for the same two periods using different input emissions corresponding to each of the two periods. Obviously, such a model-response evaluation is more demanding than the usual single-period evaluation since considerably more data and more modeling effort are required. Confounding issues include (i) the need to use emissions for two different periods estimated using a consistent

methodology and (ii) the additional variability introduced by interannual meteorological variability. As a consequence, published model-response evaluations are uncommon, but a few are available (e.g., Moran and Zheng, 2006).

Note that so-called "accountability" studies, in which the emissions changes that have occurred are due primarily to legislated control measures and the study goal is to assess the performance of the AQ models used to predict the benefits of those control measures *before* the control measures were enacted and implemented, are also model-response studies. Given the considerable time that will have elapsed, however, between the time the original AQ modeling runs were performed and the time that the AQ measurements were made following implementation of the control measures of interest, it is not likely that the particular version of the AQ model (or even the model itself!) is still being used. On the other hand, current AQ models can also be evaluated in a retrospective mode for the same legislated emission changes (e.g., 1985 Eastern Canada Acid Rain Program, 1990 U.S. Clean Air Act Amendments, 1998 U.S. NO_x SIP Call). Note also that in terms of U.S. regulatory modeling terminology (e.g., U.S. EPA, 2001, 2005b), a model-response evaluation is equivalent to the evaluation of model-predicted *relative reduction factors*.

Finally, in considering the question "How accurate does a model need to be", Reid et al. (2003) suggested that the general answer is that "… the model predictions should be good enough that model uncertainty does not affect the decisions that are based on the predictions". In the real world, of course, this may not always be the case. How then should models be used given such uncertainties?

Model applications. In their review of photochemical models and modeling, Russell and Dennis (2000) discussed the *modeling process* as a separate topic. By this they meant the set of steps required to apply a model, including selection of model domain, grid resolution, and model configuration, preparation of model input files, model execution, and postprocessing and analysis of model predictions. The modeling process itself is worthy of individual attention because, as discussed in the previous section, all of these steps may influence the results provided by the modeling system. It is thus important to work through the modeling process in as reasonable and defensible a way as possible. Some limited guidance on how to do this does exist. For example, the U.S. EPA has prepared several documents to help modellers follow "best practice" when using regional AQ modeling systems for certain regulatory applications (U.S. EPA, 2001, 2005b).

Best practice basically boils down to thoughtful and careful selection, set up, and application of an AQ modeling system accompanied by careful scrutiny and consistency checking of the results by various means, including the use of measurements and results from both alternate configurations of the selected AQ model and from other AQ models. Figure 3.4.4 describes eight steps of best practice for AQ modeling based on guidance from two EPA reports (U.S. EPA, 2001, 2005b). Most of the following steps will be relevant to any AQ model application.

Let us consider each step in turn. Some relevant background material has already been discussed in Section 3.4.3.

1. Formulate a conceptual model.
Both modeling specialists and modeling "clients" should have a conceptual understanding of the AQ issue to be considered with an AQ model. A conceptual model will provide useful guidance for all of the remaining modeling-process steps, including the identification of stakeholders, the selection and configuration of the AQ model, the development of candidate emission control scenarios, and the assessment of model results. As an example of how to formulate a conceptual model, Section 8 of U.S. EPA (2005b) lists a large number of questions about and analyses of emissions data, measurement data, and AQ model results that could be considered in developing a conceptual model for the occurrence of high annual or daily ozone levels in a particular locale or region.

2. Develop a modeling/analysis protocol.
This step may be handled either formally or informally, but if model results are to be provided to interested parties, it is desirable early on to identify all interested parties and to obtain agreement on (a) which questions should be addressed, (b) what assumptions are reasonable to make (e.g., What processes can be neglected? How large does the model domain need to be?), (c) how the modeling work should be performed, (d) what sorts of results should be generated, (e) who should review them and how, (f) what the timetable should be, and so on. The conceptual model from Step 1 can guide this development, for example, by suggesting whether long-range transport is likely to be important or not, which will help to define the geographic extent of the "community" of stakeholders. Section 9.2 of U.S. EPA (2005b) gives a detailed suggested outline of what such a protocol might look like.

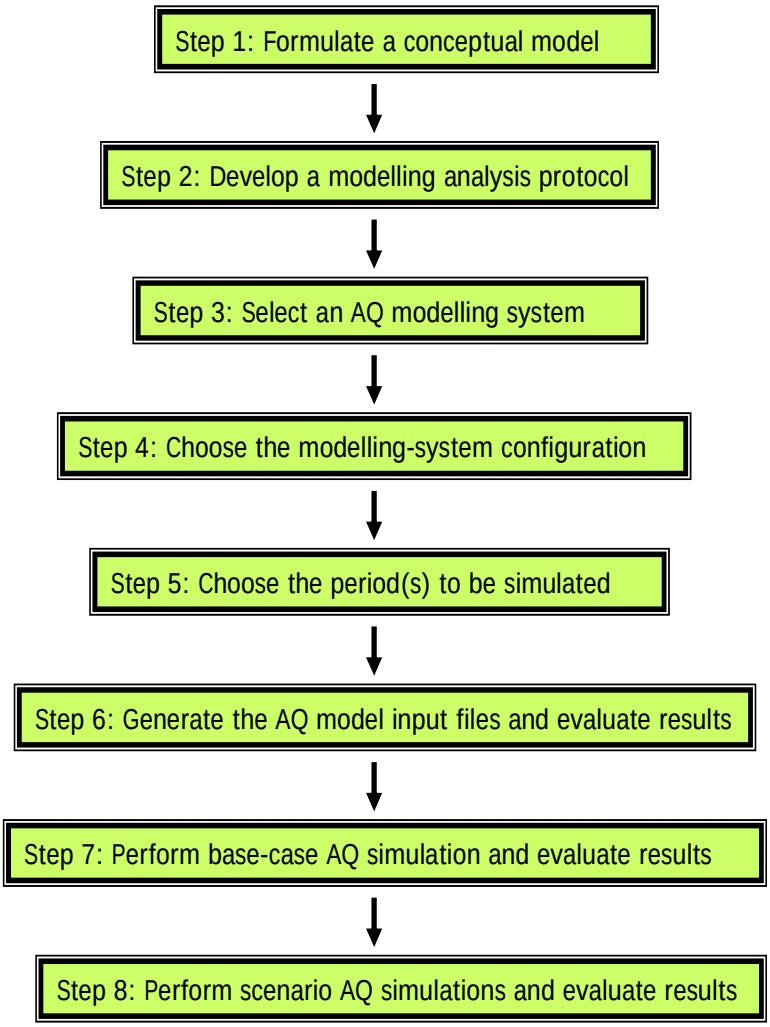

Figure 3.4.4. Eight Step Set of Best Practices for Air Quality Modeling

3. Select an AQ modeling system.
Once the questions of interest have been identified in Step 2, the next step is to select an AQ modeling system that is capable of answering those questions and that has been judged to be credible and skillful through peer review and performance evaluations. Availability of model source code, previous successful applications to similar problems, and advanced science features and tools are useful additional selection criteria. As well, the time and resources available for the model application are practical considerations that may also affect the choice of modeling system. [McKeen et al. (2005) describe seven current PM AQ modeling systems. Other AQ modeling systems are described by EMEP (2005), Heald et al. (2005), Hodzic et al. (2006), Mallet and Sportisse (2005), Meng et al. (1997), and Zunckel et al. (2006)]

4. Choose the modeling-system configuration.
Selection of an AQ modeling system is not sufficient by itself. It is also necessary to choose a modeling domain and map projection, horizontal and vertical grid resolution, level of nesting if any, an integration time step, a "spin-up" or "ramp-up" time, a "refresh" rate for the meteorological model, methods to specify chemical initial and boundary conditions, and, where choices are available, the particular physics and chemistry process parameterizations to be used in the meteorological and AQ models (e.g., convective parameterization, gas-phase chemistry mechanism, secondary organic aerosol scheme). In the case of an "off-line" AQ model, it is also desirable to harmonize to the extent possible the AQ model domain, map projection, and horizontal and vertical resolution with those of the companion meteorological model. Many of these choices will be guided by the conceptual model from Step 1 and the question(s) to be answered from Step 2. For example, the relative importance of long-range transport and the role if any of local

terrain-forced meteorological circulations such as sea-land breezes will need to be considered. Sections 12 and 13.2 of U.S. EPA (2005b) provide some useful discussions about some of these choices.

5. Choose the period(s) to be simulated.
This is one of the most open-ended steps, but it will be strongly constrained by the question(s) to be answered and, if relevant, by the form of the AQ standard (e.g., daily or annual, average or maximum) or the exact wording of the legislation of interest. In the case of short-term effects or AQ standards, the conceptual model should provide useful guidance, particularly related to the meteorological conditions that are associated with AQ exceedances. When choosing short-term simulation periods, Section 11 of U.S. EPA (2005b) recommends choosing a set of periods (a) for which extensive emissions, meteorological, and AQ data sets exist, (b) that correspond to a variety of relevant synoptic conditions, and (c) that provide enough samples to have statistical significance, (d) where each period is long enough to span a full synoptic cycle (~5–15 days) and includes a relevant exceedance. By considering full synoptic cycles, the model is forced to simulate the conditions before and after an exceedance as well, allowing confirmation that the model can forecast non-exceedances as well as exceedances (e.g, Biswas et al., 2001). Additional reasons for choosing specific periods include (e) periods during intensive AQ field experiments, for which more detailed diagnostic evaluations can be performed, and (f) periods that have already been modeled, so that either model performance is already known to be satisfactory or else comparable results are available for comparison from a peer AQ model.

In the case of long-term effects or AQ standards, correspondingly longer simulation periods will be required. Continuing advances in computer technology have meant that running AQ models for periods as long as a year or more has become feasible (e.g., Eder and Yu, 2006), but even so there are still likely to be representativeness issues due to interannual meteorological variability (e.g., Brook and Johnson, 2000). Choosing periods that satisfy short-term selection criteria (a), (e), and (f) is desirable.

6. Generate the AQ model input files and evaluate results.
This step builds upon the previous four steps and will usually require (a) preparing geophysical fields for the selected domain and grid, (b) running a prognostic meteorological model with some type of data assimilation for the simulation periods selected in Step 5 to prepare meteorological input files, and (c) running an emissions processing system for the same simulation periods to prepare emissions input files for a base case and often a number of emission scenarios as well. For any regional (i.e., limited-area) AQ modeling system, it may also be necessary (d) to run both global meteorological models and AQ models or to analyze available chemical climatologies (e.g., Logan, 1999) in order to provide chemical boundary conditions. In preparing the input emission files, the size of the model domain will dictate how many emission inventories will need to be processed. For many North American model domains, it will be necessary to process both Canadian and U.S. or both U.S. and Mexican inventories, or in some cases, all three. For AQ modeling elsewhere in the world, such as East Asia or Europe, it is also likely that multiple national inventories will need to be combined.

As part of this step, it is also important to check the input files produced so as to ensure that the inputs provided to the AQ model are as accurate and credible as possible. As discussed before, meteorology drives the AQ simulation and the AQ model results are very sensitive to the meteorological inputs in complex and nonlinear ways. At a minimum, an operational evaluation should be performed against meteorological measurements: the suite of meteorological parameters considered should include temperature, humidity, wind speed, wind direction, cloud-related fields, precipitation, and, if possible, planetary-boundary-layer depth (e.g., Hogrefe et al., 2001a, Smyth et al., 2006). Evaluation of the processed emissions is not as straightforward, but current emissions processing systems produce a range of log files and summary tables that can be checked for warning and error messages and for continuity, consistency, and plausibility, particularly when data from more than one country or jurisdiction are being combined. Visualization tools can also be applied to check the spatial and temporal patterns contained in the processed emission files. The emission files for various emissions scenarios should probably receive even greater scrutiny since extensive manipulations were likely required to transform current inventories to account for various socio-economic projections and control measures. The inclusion or exclusion (depending upon the modeling/analysis protocol) and the treatment of natural emissions such as wildfires and windblown dust should also be checked. Sections 13 and 14 of U.S. EPA (2005b) provide useful and detailed discussions concerning this step.

7. Perform base-case AQ simulation and evaluate results.
The selected AQ model should have already undergone performance evaluations, but these may have been for other time periods. In this step the AQ model is run for the base case for the time periods selected in Step 5 and its performance is evaluated so as to characterize and quantify the overall modeling system's performance (i.e., including the treatment of emissions and meteorology) and to determine whether that performance is acceptable. Given known model limitations, errors, and uncertainties, Russell and Dennis (2000), Reid et al. (2003), Seigneur and Moran (2004) and the U.S. EPA

(2001, 2005b) have all argued that this performance evaluation for the base case should not be restricted to just a basic operational evaluation against surface measurements of one or two pollutants, but instead should include a broader set of analyses that all feed into a "weight-of-evidence" judgement. Clearly, such an evaluation is somewhat open-ended and not prescriptive, but it should be more likely to lead to a correct judgement.

This broader set of analyses, many of them independent tests, could include any of the following possibilities:

- a more comprehensive operational evaluation, including consideration of a suite of ozone and PM precursors and other related gas-phase species (e.g., NO_x, NO_y, CO, NH_3, H_2O_2, HNO_3, individual VOC species) and PM chemical components both at the surface and aloft (e.g., Biswas et al., 2001);
- sensitivity tests based on alternate configurations of the AQ model, including the use of a different emissions processing system or meteorological model, different rate constants and other model parameters, different grid resolutions, different chemistry mechanisms, and different boundary conditions (e.g., Mallet and Sportisse, 2006);
- bounding tests in which emissions inputs are either increased or decreased to reflect the magnitude of uncertainties related to those inputs;
- comparisons with results from peer AQ models, including operational AQ forecast models, if these have been run for the same region and time period(s) (e.g., Hogrefe et al., 2001b; Biswas et al., 2001; McKeen et al., 2005);
- if appropriate, comparison with receptor-based model results (e.g., Marmur et al., 2006);
- comparison with observation-based models or analyses for chemical regimes, including indicator species ratios and gas ratio (e.g., Sillman et al., 1997; Stein and Lamb, 2002; Martin et al., 2004); and
- use of model probing techniques, including process analysis and direct decoupled method (e.g., Zhang et al., 2005).

8. Perform scenario AQ simulations and evaluate results.

In this last step, once the scenario simulations have been performed and the results analyzed, several additional diagnostic or comparative evaluations can be carried out to examine the reasonableness of the AQ model's response to specified emission changes, particularly if disbenefits as well as benefits are predicted to occur. These include (a) applying observation-based models for chemical regime, model probing techniques, and sensitivity/bounding tests to the scenarios, (b) comparison with the relative response functions predicted by peer AQ models for the same set of scenarios, and (c) retrospective analyses of model response to historical emission changes.

Seigneur and Moran (2004) described one comparative evaluation in which the predictions of two different PM AQ models were compared for the same three emission-change scenarios. Although the magnitudes of the responses for ozone, particulate nitrate, and $PM_{2.5}$ mass were all different between the two models for the three scenarios, the directions (i.e., sign) of the responses were the same, providing support for the general conclusions about atmospheric response. This directional consistency was particularly important in the scenario in which VOC emissions were reduced by half: both models predicted a ~30% decrease in ozone levels (at one station) but an increase in both particulate nitrate and $PM_{2.5}$ mass levels, that is, a $PM_{2.5}$ disbenefit.

The effort required to follow best AQ modeling practice and carry out each of the above eight steps for a model application may seem overwhelming. It is worth noting, however, that this is the worst case. For a jurisdiction with a history of AQ problems, a conceptual model (Step 1) likely already exists, and some AQ modeling may have already been performed. If an active in-house or external AQ modeling team with past experience for that jurisdiction can be accessed and the AQ modeling system that they use is widely recognized and accepted, then Steps 2, 3, and 4 may not be needed and the modellers can begin at Step 5. If a new scenario is similar to a past scenario in terms of the periods to be simulated (Step 5) or most assumptions about emissions (Step 6), then the generation of input data sets likely will not require as much effort as a completely new scenario for a new period and/or new model domain. And if the base scenario has been considered before, then Step 7 may not be required either, so that the completion of Step 8 is effectively the minimum requirement for a new modeling study.

Furthermore, given the open-endedness of some of the above steps and the reality of limited resources, it may not be possible to do as thorough a job as policymakers and modellers would like to do. The penalty for "cutting corners" may not be incorrect predictions but at a minimum it will be a greater degree of uncertainty and lower confidence in those predictions. Application of an AQ modeling system always entails many compromises, and the work that can be performed for the resources that are available is just one more compromise. However, the 8-step set of modeling best practice described above should be viewed as a goal to be approached as closely as possible if AQ modellers are to provide their clients with the best possible guidance.

3.4.5 Conclusions and recommendations

AQ measurement, emissions, and modeling can each provide useful information for AQ risk management, and when they are considered together they can provide additional insights and guidance. However, AQ models depend upon the availability of information about measurements, emissions, and meteorology whereas the converse is not true. AQ modeling should thus follow and not precede the development of measurement and emissions information.

AQ models are able to quantify the links between emissions of primary pollutants or precursors of secondary pollutants and ambient pollutant concentrations and other physiologically, environmentally, and optically important properties. They are also the only AQ tool available that can predict *future* air concentration and deposition patterns based on possible future emission levels. AQ models can account for the impacts of nonlinear processes and are able to predict whether a candidate abatement strategy will lead to benefits or disbenefits or both. They are naturally amenable to a "one-atmosphere" approach and can address multiple pollutants simultaneously and quantify possible co-benefits. They are distinguished from dispersion models by their inclusion and treatment of chemistry.

AQ models have been applied to date in a number of ways, both directly or indirectly, to support AQ management and policy formulation. These applications include: the evaluation of impact of emissions changes, including proposed control measures; source apportionment and source attribution; input to conceptual model formulation and development; emission inventory evaluation; measurement network and field experiment design; AQ forecasting; and testing science.

There are a large number of possible sources of AQ model error and uncertainty ranging from not understanding the underlying science and truncation errors intrinsic to the numerical techniques employed by the model to uncertainties in the input emissions and model-measurement incommensurability. There are even more ways for these numerous sources of error and uncertainty to interact, often nonlinearly and sometimes cancelling out (so-called "compensating errors"). As a consequence, AQ model uncertainty is impossible to quantify but it is possible to characterize through model performance evaluations, model intercomparisons, and sensitivity and bounding tests. There are also varying degrees of uncertainties across pollutants and their components. For example, AQ model predictions of sulphur-based pollutants are thought to have the least uncertainty as compared to predictions of carbonaceous particles or individual VOC species.

Box (1979) wrote that "All models are wrong, but some are useful". For an AQ model to be "useful", it should be credible. For it to be credible, it should give the right answers for the right reasons. AQ model credibility is established through model review, model performance evaluations, and successful model applications. However, model credibility is always provisional, so model evaluation (and model improvement) should be an ongoing process. Any new model application is a new opportunity for a model to fail.

A key question in applying AQ models is: "How accurate does a model need to be?" Reid et al. (2003) suggested that the general answer is that "model predictions should be good enough that model uncertainty does not affect the decisions that are based on the predictions". AQ modeling uncertainty can be managed and limited by following "best practice" at all stages of the modeling process. Best practice basically boils down to thoughtful and careful selection, set up, and application of a credible AQ modeling system accompanied by careful scrutiny and consistency checking of the results by various means, including measurements and results from both alternate configurations of the selected AQ model and from other AQ models. The credibility of the model predictions for a given application is then determined based on a weight-of-evidence judgment that considers all of the evaluation results. This process is not at all "cut and dried" —it is much more in the nature of applied research than of a routine activity.

The AQ modeling process is demanding in terms of requirements for input data, skilled modeling personnel, calendar time, and computer resources. AQ modeling should be viewed as a significant undertaking. Therefore, such programs must be resourced appropriately, including a long-term commitment, in order to obtain credible information useful to AQ risk management.

3.5 Combining Measurements, Emissions and Model Output

Independently, emission inventories, measurement programs and models are essential tools for AQ risk management and for describing the state of the atmosphere. A range of new methods are being explored that combine emissions and measured and modeled concentration fields to expand the capability of daily, routine AQ forecasts and improve estimates of intra-urban and inter-urban variation in long-term or chronic exposure. 'Fusing' these diverse information sources together to support a wide range of health and air quality studies, as well as real time data reporting and analysis,

holds considerable promise. Figure 3.5.1 presents a conceptual picture of the types of multi-scale information that can potentially be assimilated or 'fused' into a complete picture of the spatial variation in air pollutant concentrations. Although they are a source of input to the AQ models, emission inventories may also represent an independent source of spatial information and/or a predictor for use in empirical models.

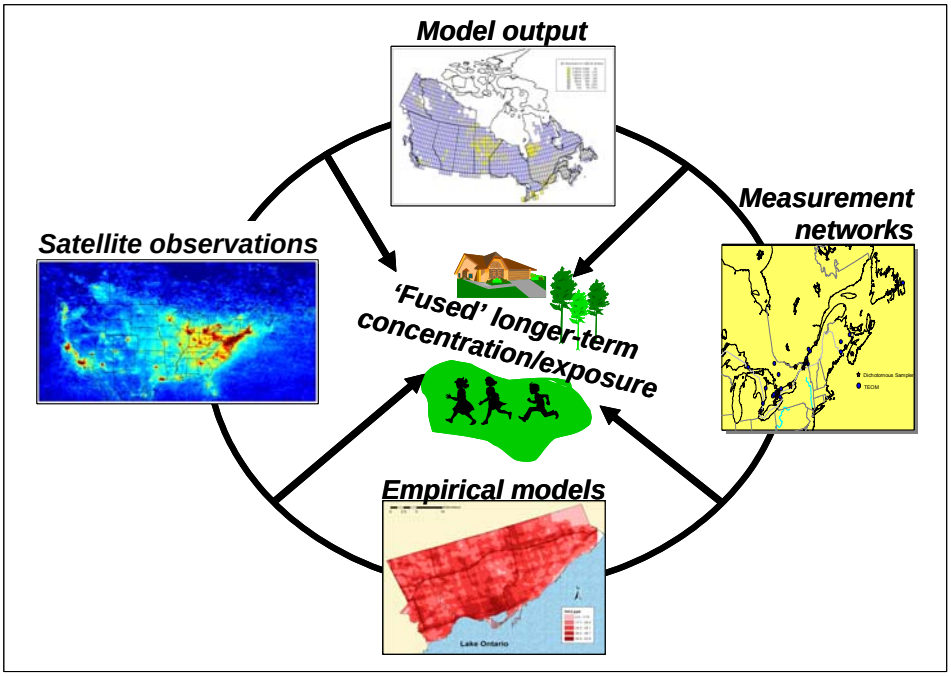

Figure 3.5.1. Conceptual system of fused or assimilated data for estimating longer-term ambient concentration patterns or chronic air pollutant exposure at any geographic location and at any scale, from regional to neighbourhood. The system needs to be flexible in terms of which source(s) of information the estimate relies most heavily upon at each scale of interest and must be capable of providing estimates of location-dependent uncertainty. This uncertainty will vary geographically due to inconsistencies in the amount of information that is available at the finer scales.

Data assimilation routines using real-time data and model output are now being applied on a continuous basis to characterize large scale patterns across North America. The amount and quality of information available varies from pollutant to pollutant and geographically. At present, in North America, ozone is the most advanced, while routines for $PM_{2.5}$ are being developed. Figure 3.5.2 presents ozone concentrations across eastern North America derived from the Canadian Meteorological Centre air quality forecasting system and ozone data compiled under the AirNow program (Menard and Robichaud, 2005). This image is derived from gridded ozone concentrations that are produced by combining observations with model predicted concentrations to 'interpolate' and prepare the data for computing future concentrations using an air quality forecasting model. Hourly concentrations at each grid point were used to compute the maximum 8 hr average concentration at each point on each day. These concentrations were then combined for the five month period known as the "ozone season".

At present AQ models only assimilate surface observations, but approaches for "chemical data assimilation" are undergoing considerable research and development (Menard, 2006). The long term goal is to begin utilizing observations from satellites and possibly other irregular sources of information (e.g., aircraft). The most advanced satellite instrument is OMI (Ozone Monitoring Instrument) on the Aura spacecraft, which was launched in 2004 (Schoeberl et al., 2004). In terms of the common air pollutants, daily, 13x24 km resolution observations for O_3, NO_2, SO_2 and aerosols are being measured. Devising the appropriate procedures for assimilating and/or interpreting such data presents a significant scientific challenge. Even with a satellite such as Aura, observations are, at best, once per day if no clouds obscure the measurements.

The map of NO_2 over the northeast of North America shown in Figure 3.5.3 provides an indication of the capabilities of OMI. It is important to note that satellite observations of trace gases and aerosols are 'total vertical column' amounts (i.e., not necessarily surface conditions). Research is needed to further improve the processing of satellite data from the raw signals and from other supporting data (e.g., correcting for clouds and variations in surface albedo) and then in

deriving boundary-layer and/or surface concentrations, as well as vertical profiles. The latter of which requires, in itself, the combined use of AQ and meteorological models and surface observations. Nonetheless, satellite data represents a valuable source of information because it is freely available and provides global coverage - air pollutant information can be obtained where no monitoring exists. In addition to the initialization of AQ models, the spatial patterns derived from satellite observations (i.e., across days, weeks or months) are well-suited to determining, in an internally-consistent manner, gradients in chronic exposure across large regions and among different countries. Thus far, aerosol observations ($PM_{2.5}$) have received the most attention for this purpose (e.g., Liu et al., 2005).

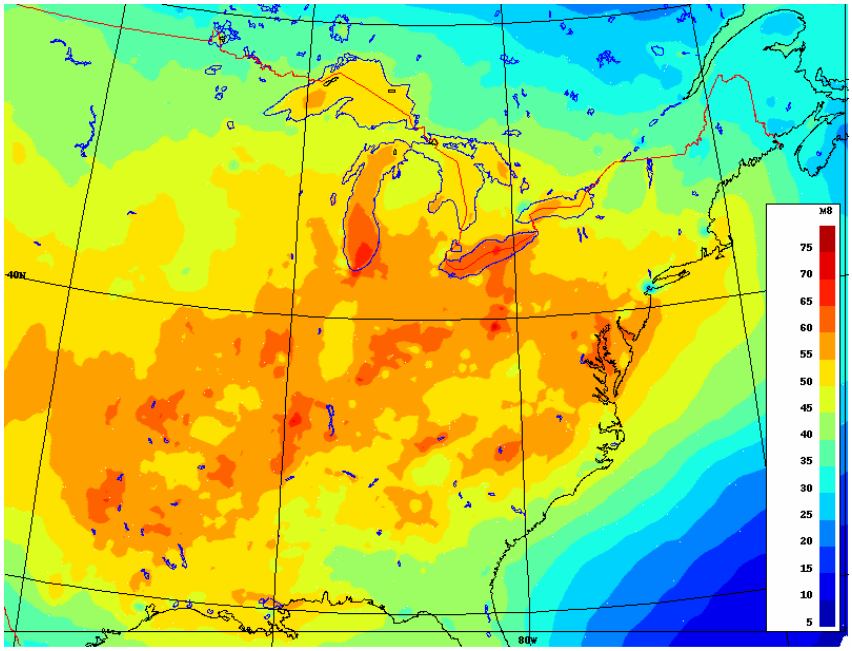

Figure 3.5.2. Average daily 8 hour maximum ozone (ppb) for summer (May-Sept.) 2005.

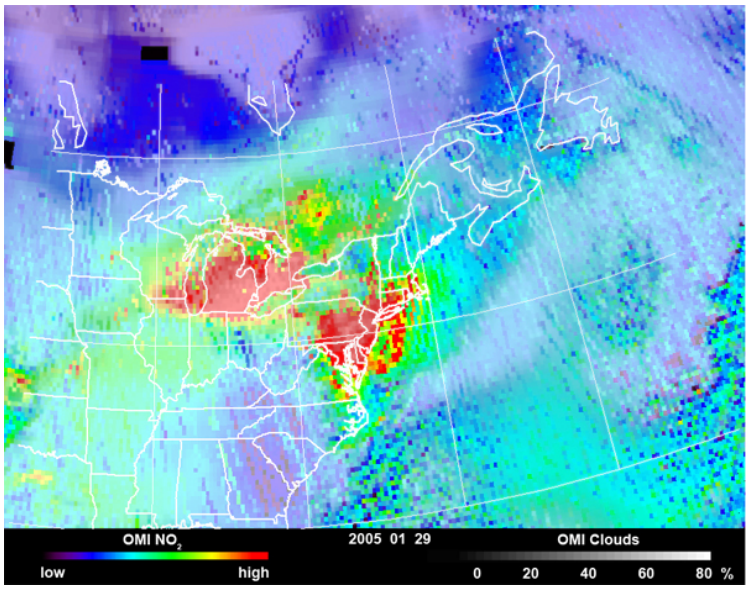

Figure 3.5.3. Nitrogen dioxide (NO_2) observations from the ozone monitoring instrument (OMI) on the AURA satellite. The red areas show high amounts, the purple areas show regions of clouds which shield the sensor from nitrogen dioxide below the clouds. On Jan. 29, 2005, and subsequent days there was a large $PM_{2.5}$ event due to high particle nitrate concentrations. This period represents the first wintertime air quality advisory ever issued in the Province of Ontario. Air quality alerts were also issued for Michigan during this period. The large concentrations over southern Michigan and southwestern Ontario are consistent with the surface conditions observed during the period. The OMI instrument was provided to the Aura Mission by the Dutch and Finnish space and meteorological agencies. Image generated by OMI NO_2 team. (contact James F. Gleason, NASA/GSFC)

Population exposure to ambient air pollution occurs at neighbourhood scales. This is beyond the resolution of all the sources of information discussed above (i.e., Figures 3.5.1, 3.5.2, and 3.5.3). Furthermore, it is unlikely that any of these will be able to resolve such scales in the future and in terms of the types of deterministic AQ models discussed above, it is not reasonable to expect the meteorology to be modeled or for emissions information to provided at a fine enough scale. At best, new parameterization schemes or independent emissions models will be developed to treat sub-grid scale features to enable AQ models to reliably predict at the 1 to 5 km resolution. However, research is needed to determine how best to use such models to predict population exposure changes and their uncertainties within these grid sizes so that the costs and benefits of local scale air quality risk management strategies can be evaluated.

To resolve urban to neighbourhood scale exposure patterns for health studies a variety of approaches are currently being applied. These range from interpolation of monitoring site data (e.g., Jerrett et al., 2005) to exposure surrogates such as distance-to-roadway and traffic counts (e.g. Hoek et al., 2002) to small scale dispersion models and/or combinations of both (Wu et al., 2005; Cyrys et al., 2005). The local scale exposures of interest have generally been associated with traffic since data on road networks are readily available. However, a wider range of emission sources have been included in some ambient air pollutant exposure modeling efforts (e.g., Gram et al. 2003).

Geographic Information Systems (GIS) have proven to be useful for mapping exposure patterns, integrating different sources of information and in developing land-use regression models (Brauer 2006a). Intra-urban chronic exposure estimates have been derived using LUR for several cities (Brauer et al. 2003; Kanaroglou et al, 2005; Sahsuvaroglu et al. 2006; Gilbert et al., 2005; Silva et al. 2006; Setton et al., 2006; Luginaah et al., 2006; Brauer 2006b). Cyrys et al. (2005) compared both LUR and dispersion model estimates for NO_2 and $PM_{2.5}$ and reported that for their cohort of interest in Munich, Germany, the two approaches led to similar exposure classifications. These results and most other LUR efforts have focused on traffic-related pollutants (e.g., NO_2 and sometimes $PM_{2.5}$). However, recent studies in Windsor, Ontario, have expanded the dependent variables in LUR to include SO_2, benzene and toluene (Wheeler et al., 2006).

The empirical models image in Figure 3.5.1 is an example of the NO_2 surface predicted by LUR for Toronto, Ontario (Kanaroglou et al., 2005). The small scale spatial variability (i.e., neighbourhood scale or better) produced by applying the LUR for all points in a GIS database appears more-realistic compared to the pattern obtained using interpolation, with respect to the known distribution of traffic (Jerrett et al., 2004). A LUR model, once developed for the area of interest, also provides the capability of estimating chronic exposures for each member of a health study cohort if their addresses are known. Ideally, such estimates should be spot-checked with independent measurements within residential areas, inside a variety of homes and also in comparison to personal exposure measurements. This could potentially lead to the coupling of LUR models for outdoor, at home, concentrations with individual exposure models that consider home characteristics and time activity.

3.6 Overall Summary, Conclusions and Recommendations

To be written after review by NERAM V attendees and after discussion at NERAM V and any subsequent follow up.

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CHAPTER 4: What Air Quality Management Approaches are Currently in Place and Which Strategies Have Been Shown to be Effective?

Authors: Bart Croes and Jurgen Schneider; Contributor: Alan Krupnick

This chapter presents strategies for improving ambient air quality at the local, regional and global levels. Several case studies provide examples to illustrate some of the approaches and identify factors associated with successful policy development and implementation. Evidence of effectiveness is emphasized. The chapter reviews policy approaches including mobile source, point source and area source emission reduction strategies; standard-setting approaches; market-based approaches; trans-boundary strategies; multi-pollutant strategies; and public education/behavioral approaches.

4.1 What is the range of air quality management approaches in Europe and North America?

Different countries have developed and implemented different approaches to tackle air pollution problems. This chapter focuses on the development within the European Community (EC; former European Economic Community, EEC) and North America. The chapter does not provide a comprehensive overview, but focuses on a few relevant developments.

4.1.1. Historical development

European Union legislation on environmental issues and air pollution

An important starting point for the development of environmental policy was the first United Nations Conference on the Environment in Stockholm in 1972. In 1972 the European Council made a commitment to establish a Community environmental policy. The first so called Environmental Action Programme (EAP[3]) was decided in November 1973, which laid down principles for the environmental policy in the Community. It emphasized inter alia that economic development, prosperity and the protection of the environment are mutually interdependent [ref to follow].

However, environmental considerations were also always linked to other considerations relevant for policy development within the Community, e.g., the setting of uniform emissions standards to avoid distortions to industry competitiveness. Product regulations had to be harmonised in order to avoid non-tariff barriers originating from different national product norms. On the other hand, the economic benefits, especially the positive employment effects to be gained from environmental policies were stressed.

Environment policy was built into the Treaty by the Single European Act of 1987 and its scope was extended by the Treaty on European Union on 1992. This allowed the use of majority voting on environmental legislation. The general objectives formulated now in the Treaty are to:

- Preserve, protect and improve the quality of the environment,
- Protect human health, and
- Utilize natural resources in a prudent and rational way.

For achieving these environmental objectives the Treaty explicitly lists the precautionary principle, the principle of preventive action, the principle of rectifying damage at the source and the polluter pays principle.

In 1992, the EC set itself the objective for achieving sustainable development. The long-term goal, to transform the European economy into one whose development would be sustainable for generations to come, was set out in the 5th Environmental Action Programme 'Towards Sustainability'[4]. In addition, the 5th Environment Action Programme calls for the effective protection of all people against recognized health risks and demands that the guideline values of the Word Health Organization (WHO, 2000) should become mandatory at the European Union (EU) level.

The 6th Environmental Action Programme (covering the period from 2001 to 2010) identifies four environmental areas for priority actions:

[3] http://ec.europa.eu/environment/env-act5/envirpr.htm
[4] http://ec.europa.eu/environment/actionpr.htm

- Climate Change.
- Nature and Biodiversity.
- Environment and Health and Quality of Life.
- Natural Resources and Waste.

The main avenues for action include:

- Effective implementation and enforcement of environmental legislation: necessary to set a common baseline for all EU countries.
- Integration of environmental concerns: environmental problems have to be tackled at their source.
- Use of a blend of different approaches: all types of instruments have to be considered, not just legislation. The essential criteria being optimal efficiency and effectiveness.
- Promoting of participation and involvement across society – business, citizens, NGOs and social partners – through better access to quality information on the environment and co-operating to devise solutions.

In addition, the EAP requires the European Commission to prepare Thematic Strategies covering seven areas including air pollution.

Environmental legislation leaves plenty of scope for national action and allows Member States to take tougher protection measures than those agreed at the EC level. The situation is different for legislation affecting the free movement of goods (e.g., product regulations). Stricter regulations may only be applied in special cases.

Development of air quality legislation in the European Community

The first so-called Directive of the European Community on air quality entered into force in 1980 (Directive 80/779/EEC). This Directive set air quality limit values and guide values for sulphur dioxide and suspended particulates. The Directive specified a date by which the limit values had to be attained, but also allowed for a prolonged period of noncompliance in zones if a Member State could show that plans for the progressive improvement of the quality of the air in those zones were developed. From today's perspective, the limit values were rather high. In 1982 and 1985, new Directives on lead and nitrogen dioxide, respectively, entered into force. These Directives also contained limit values.

In 1992, an ozone Directive was decided. This Directive did include certain threholds for the assessment of air pollution and for the warning of the population, but did not request emission reductions in the case of exceedances of these assessment thresholds [ref to follow].

In 1996, the Environment Council adopted an **Air Quality Framework Directive (FWD) 96/62/EC** on ambient air quality assessment and management. This Directive covers the revision of previously existing legislation, the introduction of new air quality standards for previously unregulated air pollutants and setting the timetable for the development of daughter directives on a range of pollutants. The list of atmospheric pollutants to be considered includes sulphur dioxide, nitrogen dioxide, particulate matter, lead and ozone – pollutants governed by already existing ambient air quality objectives – and benzene, carbon monoxide, polycyclic aromatic hydrocarbons, cadmium, arsenic, nickel and mercury. The so-called daughter directives to the Air Quality FWD are described in more detail in section 4.1.2.

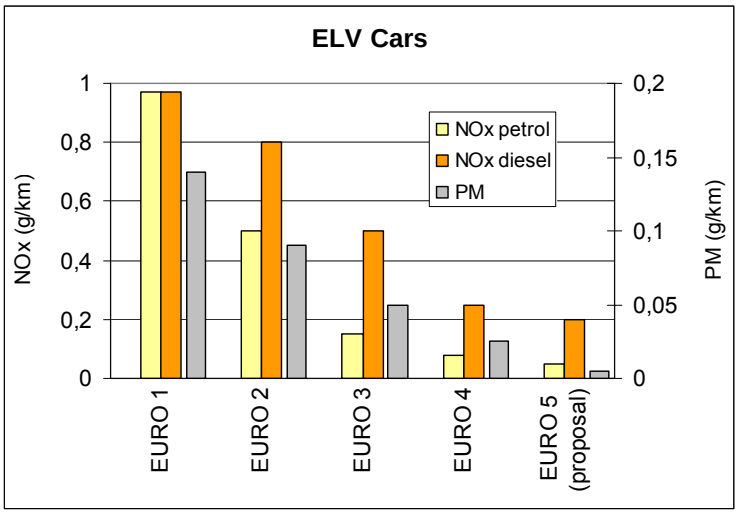
The Air Quality FWD and its daughter directives are only one pillar of the EU air quality legislation. A number of other directives had considerable (and partly larger than the air quality directives) impact on air quality, notably those setting emission standards for mobile and stationary sources. In addition, some directives regulate product standards, a few of them are also particularly important for air quality (such as the Directives on fuel quality, solvents, etc.).

The most important EC Directives which impact air quality include:

- The **EURO standards** have established emission limit values for different pollutants for cars (differentiated between diesel and gasolinel fueled), light- and heavy-duty vehicles. As an example, the development of emission limit values (ELV) for cars (NOx and PM) is given in Figure 1[5].

Figure 1: Development of emission limit values (ELV) for cars in the EU for NOx and PM.

[5] However, it has to be noted that real life emissions can be considerable higher than the ELV.



- The directive on **Large Combustion Plants** (LCP, 2001/80/EC) sets more or less stringent emission limit values for large installations in the power generation sector.
- The Directive concerning **Integrated Pollution Prevention and Control** (IPPC, 1996/61/EC) requires the implementation of the Best Available Technology (BAT) concept to a large number of industrial activities (energy industries, production and processing of metals, mineral and chemical industries, waste management, etc.), for which it lays down general rules for the national permitting systems. The Directive covers both new and existing installations. The basic concept is that operators should go as far as they reasonably can to optimize their environmental performance by applying the best available techniques. Environmental performance is eventually to be measured against meeting the existing environmental quality standards, e.g., for air pollution to comply with the air quality standards of Community legislation. Measures going beyond BAT may be requested if this is necessary to achieve EC environmental objectives. The IPPC Directive covers only larger installations (> 50 MW). However, there is no comparable EU legislation for small (including the domestic sector) and medium installations, even though these source categories may contribute significant to excess air pollution.
- The Directive on **National Emission Ceilings** (NEC, 2001/81/EC)**.** This Directive sets national emission ceilings for the pollutants SO_2, NOx, NMVOC and NH_3.
- The Directive on Volatile Organic Compounds Emissions from Storage and Distribution of Petrol (94/63/EC).
- The Directive on Solvents Use in Industry (99/13/EC).
- The Directive on Sulphur Content of Liquid Fuels.
- The Directive on Emissions from Engines to be Installed in Non-Road Mobile Machinery.
- The Directive on the Quality of Petrol and Diesel Fuels.
- The Directive on emission of VOCs due to use of organic solvent.
- The Directive on the incineration of waste.

Trends in emissions in the European Union

Even though there was continued economic growth in the past decades, emissions in general stabilized or decreased. As an example, the aggregated emission of PM (primary and precursors for secondary PM) are shown in Figure 2.

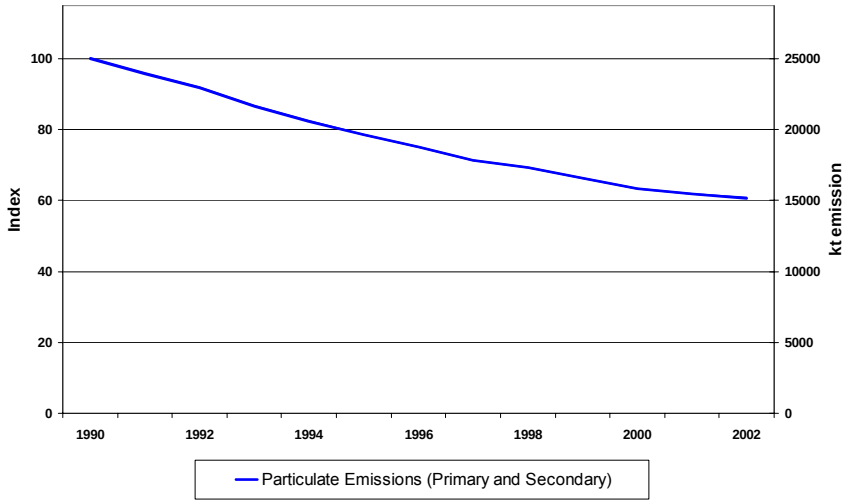


Figure 2: Emissions of primary and secondary fine particles (EU-15), 1990-2002. Source: EEA, 2006.

This decoupling was triggered by stringent legislation, but also by other factors including fuel switch (which was partly influenced by economic considerations). This is illustrated for the power sector and SO_2 emissions in Figure 3.

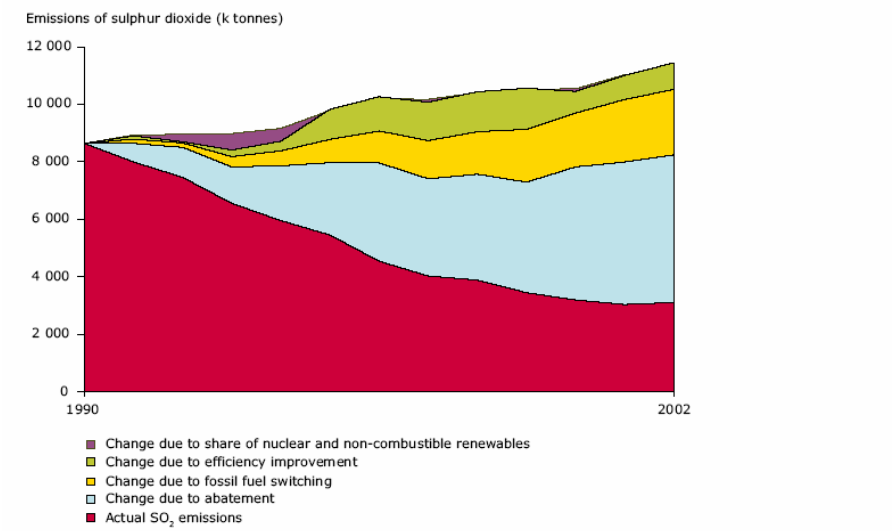


Figure 3: Development of SO_2 emissions from the power generation sector in the EU 15. Source: EEA, 2006

Development of air quality legislation in North America

United States

[To be written – rely on NRC report "Air Quality Management in the United States]

California[6]

California's air pollution control program began in 1959, when the California legislature created the California Motor Vehicle Pollution Control Board, to certify emission control devices for vehicles. Subsequently, under the Federal Air Quality Act of 1967, California was granted a waiver to adopt and enforce its own emission standards for new vehicles, in recognition of California's unique air quality need to set more stringent emission control requirements compared to the rest of the nation. In 1967, the California Air Resources Board (CARB) was formed through the Mulford-Carrel Air Resources Act, signed into law by Governor Ronald Reagen. The Act created CARB by merging the California Motor Vehicle Pollution Control Board and the Bureau of Air Sanitation. CARB has the ability to set mobile source emission standards more stringent than the U.S. Environmental Protection Agency, except sources involved in interstate commerce: trains, planes, ships, and interstate trucking. Other states, like many in the Northeast U.S., have taken advantage of their option to adopt California's mobile source emission standards.

CARB also sets regulations for consumer products, paints and solvents, and identifies and controls toxic air contaminants. It coordinates the efforts of federal, state and local authorities to meet ambient air quality standards, while minimizing the impacts on the economy. While local air quality management districts have the primary authority to control emissions from stationary and areas sources, CARB can assume this authority if local agencies do not develop or implement their air quality plans. Californians support and want air pollution control – 65% put environmental protection over economic growth (although California has accomplished both), and this has created a supportive Legislature. For example, the California Legislature recently passed a bill (signed by Governor Arnold Schwarzenegger) to give CARB the authority to regulate greenhouse gas emissions 1990 levels by 2020, a 25% reduction from business as usual.

The governor of California, with the consent of the State Senate, appoints the 11 members of CARB, five of which are from local air quality management districts. It is an independent board when making regulatory decisions. The Board is required to have a medical doctor and an engineer as members. The first chairman was a respected atmospheric scientist (Professor Arie Haagen-Smit) who discovered how urban smog was created and the latest (Dr. Robert Sawyer) was formerly a mechanical engineering professor at the University of California, Berkeley. Except for the Chairman, the Board only meets once per month and relies on its staff for technical input. The Board oversees a $150 million budget and a staff of over 1,100 employees located in Northern and Southern California. In addition, the board provides financial and technical support to the 35 local districts. CARB is funded by vehicle registration fees and fees on stationary sources and consumer products. It also receives up to $166 million per year in incentive funds from fees on vehicle registration and new tire sales. This goes to diesel engine retrofits, car scrappage, and agricultural, port and locomotive projects.

California has 4,000 air quality professionals at the State and local levels. Most of CARB's workforce are engineers and scientists, and about 20% have Ph.D.'s and Master's degrees. CARB conducts its own vehicle testing programs and funds extramural research at a level of $5 million per year, taking advantage of the strong academic community in California and other states. It also funds a technology demonstration and commercialization program, and the development of state-of-the-art emission, air quality and macroeconomic models. The technology research demonstrates that reduced emissions are feasible, but the use of performance-based standards allows industry to come up with more cost-effective approaches. Enforcement and monitoring programs ensure that the emission standards are met. CARB has a requirement that the scientific underpinnings of all its regulations undergo scientific peer review. This is normally done by the University of California. Underlying this science-based approach is the willingness to move ahead in the face of some uncertainties.

Mexico[7]

The Mexican air quality situation is dominated by the Mexico City Metropolitan Area (MCMA), home to 20 million people, 3.5 million vehicles and 35,000 industries. The governance of the MCMA is split primarily between the Federal District (*Distrito Federal* or DF) and the State of Mexico (*Estado de Mexico* or EM). One of the major obstacles to the implementation of anti-pollution measures in the MCMA is the lack of a powerful metropolitan institutional structure. The Metropolitan Environmental Commission (*Comision Ambiental Metropolitana*, or CAM) was created in 1996 to coordinate the policies and programs that are implemented in the metropolitan area. Permanent members of CAM consist of the federal Secretariat of Environment and Natural Resources, the federal Secretariat of Health, the Chief of Government of the Federal District, and the Governor of the State of Mexico.

[6] Adapted from Susan O'Connor and Robert Cross, "California's Achievements in Mobile Source Emission Control", *EM*, July 2006
[7] From Luisa T. Molina, Mario J. Molina, et al., "Air Quality in Selected Megacities", Critical Review Online Version, *J. Air & Waste Assoc.*, 2004.

Every two years, the responsibility to preside over CAM changes between the DF and the EM governments. Any decision on how to organize the Commission as well as the responsibility for operating costs would go to the jurisdiction in office at the time. Frequently, the side presiding over CAM has to use its own financial resources to manage the commission and its own environmental officials also serve as CAM officials. The local government that is not presiding over CAM, as well as the federal government, contributes human resources and other support to CAM operations, mainly for the specific tasks of its working groups.

The Environmental Trust Fund for the Valley of Mexico (*Fideicomiso Ambiental del Valle de Mexico*) was created exclusively to support CAM projects. Between 1995 and 1997, the Trust Fund received money collected from the application of a surcharge on gasoline sold in the MCMA. The annual renewal of the surcharge required the approval by the Finance Ministry, which did not happen in 1998. Since then, the surcharge has not been reactivated. The Trust Fund has its own organization and rules of operation, and it is managed through an Executive Committee headed by the Finance Ministry. One representative each from CAM, governments of the DF and the State of Mexico and SEMARNAT are included. However, without income, the Trust Fund has been depleted. Other sources of funding for CAM projects include international environmental agencies, national and international financial institutions, international and national academic institutions and foreign governments.

There are serious concerns over its current operation: one of the most important issues is that CAM does not have a specific budget for its own operation, nor does it have a defined operative organizational structure as well as lack of continuity. The Technical Secretariat is appointed by the presiding government, which rotates every two years; in addition, local and federal representatives change in response to political events. These deficiencies in institutional memory cloud an integrated long-term vision of the policy requirements.

The Metropolitan Commission for Transport and Roadways (*Comision Metropolitana de Transporte y Vialidad*, or COMETRAVI) has a mandate similar to that of CAM, but it also lacks financial resources and has no executive or regulatory powers. In 1999, COMETRAVI developed a proposal for the adoption of comprehensive integrated strategies for transportation and air quality in the MCMA. This strategy has not been incorporated into the official programs.

The lack of integration of environmental policies with transportation, urban development and land use planning is one of the most important barriers preventing sustainable environmental improvements. Another important barrier is the incomplete harmonization of environmental policies among the Federal Government, the State of Mexico and the Federal District, which results in unfair practices and inefficiency. Also, at present neither local nor federal environmental agencies have sufficient human and financial resources to efficiently carry out their environmental management activities. Furthermore, the continuing dispersion and growth in the size of the MCMA drive the need for vehicle-miles traveled still higher. The almost totally unregulated establishment of communities on the periphery creates both mobility and environmental problems. The development of a regional planning commission with strong enforcement capability is fundamental to creating a sustainable transportation/environmental system in the MCMA.

As a large source of emissions, the MCMA has the potential to influence air quality over a much wider region than the Valley of Mexico thus exposing larger populations in nearby cities and also affecting forests and crops. Pollutants emitted outside of the MCMA likewise may influence air quality within the Valley of Mexico. Therefore in addition to metropolitan coordination, there is an urgent need for regional coordination and planning. To ensure continuity in the implementation of long-term strategies, it is essential that the CAM be significantly restructured and be empowered to carry out the planning, integration and implementation of metropolitan environmental policies.

Canada

[To be expanded – placeholder information below is borrowed from Molina et al. 2004, see also presentation by Barton and McLean]

The regulation of pollutants in Canada occurs at the federal, provincial, and municipal levels. The Canadian federal government, in consultation with provincial ministers of the environment, recently developed a set of ambient air quality standards known as Canada-wide standards (CWS). Provincial governments are responsible for implementing air quality standards, but are free to design their own implementation plans. The Canadian federal government has responsibility for setting new equipment emission standards; however, provinces are free to set their own more stringent ones. Until fairly recently, Canadian standards were more lax than those in the United States, but in the past few years the gap has closed.

The proximity of the Central Ontario Region to the border makes it vulnerable to long-range transport of pollutants from the United States; this entails the collaboration between the two countries in the development and implementation of cost-effective emission control strategies. The Canadian government, industries and non-governmental organizations are

all taking positive steps to help reduce the level of pollution in Canada. The government has instituted emission caps, emission trading, a "drive clean" program, and other initiatives aimed at reducing pollutant emissions. Many industries are also taking voluntary steps to reduce emissions, while the non-governmental organizations continue to lobby government, industry and the public to adopt practices that will reduce the emission levels.

4.1.2. Air quality standards, guidelines and limit values

European Union

Numerical limit and target values as well as threshold values were set in four so-called daughter directives (DD) to the air quality FWD. The limit values from the first and second DD are listed in Table 1 (the third and fourth DD do only contain target values). The table also contains information on the basis for setting standards. As requested in the 5th EAP, many numerical values are identical to WHO AQG levels as contained on the Air Quality Guidelines for Europe (AQG, WHO, 2000). For some pollutants, the WHO AQG do not contain a numerical level (such as PM and benzene). The respective limit values were based on recommendations from technical working groups.

Table 1: Numerical values of air quality limit values in the EU.

Pollutant	Averaging period	Limit value	Basis for setting standard	Exceedances allowed
SO_2	1 hour	350 µg/m³	Based on WHO AQG level[8]	24
	1 day	125 µg/m³	WHO AQG level	3
NO_2	1 hour	200 µg/m³	WHO AQG level	18
	1 year	40 µg/m³	WHO AQG level	-
PM_{10}	1 day	50 µg/m³	Risk assessment in combination with an assessment of feasibility	35
	1 year	40 µg/m³	Risk assessment in combination with an assessment of feasibility	-
Benzene	1 year	5 µg/m³	Risk assessment in combination with an assessment of feasibility	-
Lead	1 year	0,5 µg/m³	WHO AQG level	-
CO	8 hour	10 mg/m³	WHO AQG level	-

There are some important specifics about the limit value concept of EU legislation. The most important include:

- The LV have to be attained within a given period and not to be exceeded once attained. This definition implies that the limit values are not a weak environmental objective, but a strict requirement (which in principle also constitutes individual rights for citizens).
- The limit values apply in principle everywhere (including hot spot locations) except at workplaces (in reality, compliance monitoring and therefore compliance assessment includes hot spots, but usually focuses on those hot spots where exposure can occur).

North America

The North American countries have each separate ambient air quality standards. Within the United States, California has set its own standards, generally more stringent than those set by the U.S. Environmental Protection Agency. Even though they rely on many of the same human exposure and epidemiological studies, these standards have striking differences. The use of allowable exceedances, spatial averaging of monitoring data, and natural (e.g., dust storms) and exceptional event (e.g., prescribed burn) exceptions can greatly reduce the stringency of these standards.

[8] WHO AQG for SO2 is 500 µg/m³ as 10-minute average.

Table 2: Ambient air quality standards for North America

Pollutant	Averaging period	U.S.	California	Mexico	Canada
SO_2	1 hour	--	655 µg/m³	350 µg/m³	160 µg/m³
	1 day	365 µg/m³	105 µg/m³	80 µg/m³	30 µg/m³
NO_2	1 hour	--	470 µg/m³	400 µg/m³	--
	1 year	100 µg/m³	--	--	60 µg/m³
PM_{10}	1 day	150 µg/m³	50 µg/m³	150 µg/m³	50 µg/m³
	1 year	--	20 µg/m³	50 µg/m³	--
$PM_{2.5}$	1 day	35 µg/m³	--	--	30 µg/m³
	1 year	15 µg/m³	12 µg/m³	--	--
Ozone	1 hour	235 µg/m³	180 µg/m³	216 µg/m³	100 µg/m³
	8-hour	160 µg/m³	150 µg/m³	--	--
CO	1 hour	40 mg/m³	23 mg/m³	--	34 mg/m³
	8-hour	10 mg/m³	10 mg/m³	13 mg/m³	--

4.2. Case study – European Union

Stringent ambient air quality standards by themselves do not provide protection. The main tools to achieve the limit values are so called plans and programmes (if the sum of the limit value and a so called margin of tolerance is exceeded) and, after the attainment date, action plans, which have to be implemented if there is a danger of exceeding limit values. The limit values for PM_{10} and NO_2 are rather stringent, and exceedances are frequent in some parts of Europe [ref to follow]. This trigged the development and implementation of air quality plans to reduce pollution. These plans also have to be reported to the European Commission. The plans are currently scrutinized in a project funded by the European Commission. Figure 4 provides an overview of the pollutants covered by the plans and those sources which have been identified as main source of this pollutant.

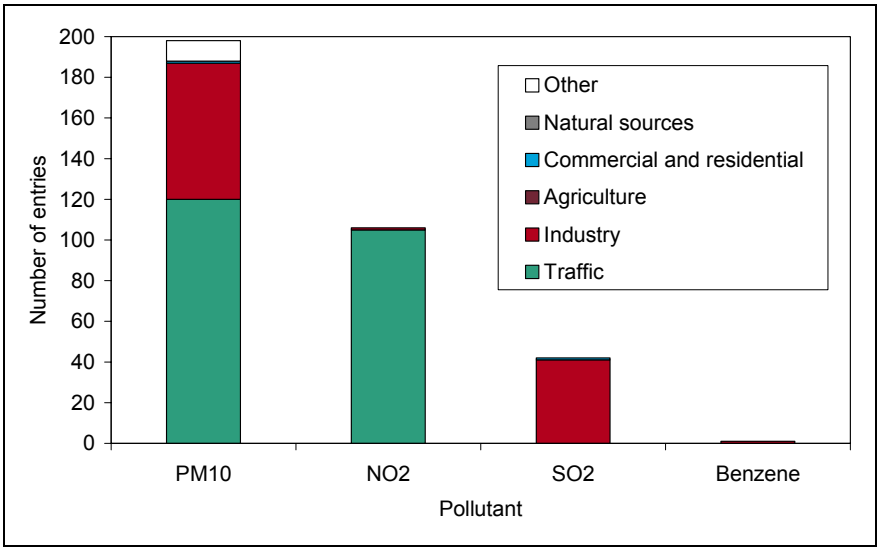


Figure 4: Main sources listed for different pollutants in plans and programmes reported to the European Commission for exceedances of limit values between 2001 and 2003.

There is some flexibility concerning the implementation in Member States, e.g. concerning responsibilities. There are large differences concerning the responsible authorities; in some Member States, local authorities are responsible for air quality assessment and management, while there are also examples where the responsibility lies with regional or national authorities. There is no simple answer to the question which model is most effective.

4.2.1. Exposure reduction target (ERT)

Experience has shown that for non-threshold pollutants, single limit values or standards may not on their own be the most appropriate way of managing air quality, particularly in areas where existing air quality management systems are mature. This has encouraged the European Commission to propose a new, additional concept, the exposure reduction target (ERT) (which has not entered into force yet, even though the concept is in principle supported by the Council and the European Parliament). The following short description of the basics of the concept is derived from a non-paper issued by the European Commission.

The existing legal framework of the Air Quality FWD and its Daughter Directives require complete compliance meaning that limit values must be met everywhere continuously. As such, a conventional air quality management strategy would implement measures according to their cost-effectiveness so as to reduce the areas of exceedance of these limits. Such a strategy would deliver increasingly smaller areas above the limit values. In the remaining areas, it may well be that reaching complete compliance is very difficult and costly. In addition, there would be little incentive to improve air quality where limit values are already respected.

For pollutants with no effect threshold, such as $PM_{2.5}$, it will generally be more beneficial for public health to reduce pollutant concentrations across the whole of an urban area as benefits would accrue from reductions in pollution levels even in relatively "clean" areas.

Therefore, an ERT was proposed for fine particulate matter $PM_{2.5}$ $PM_{2.5}$ is responsible for significant negative impacts on human health. Further, there is as yet no identifiable threshold below which $PM_{2.5}$ would not pose a risk. Advice from the WHO suggests that it is justified to assume a linear response linking exposure to $PM_{2.5}$ to adverse effects. This advice should apply both in "clean" as well as in "polluted" areas. The exposure reduction concept entails a reduction in the exposure of a larger part of the population compared to the limit value approach which affects (as we approach complete compliance) a smaller number of people. As such, the overall improvement in public health comes at a higher cost with limit values. A Commission Working Group has looked at this issue and concluded that exposure reduction would be a more cost-effective way of reducing air pollution. [9]

However, there is also an issue of environmental justice. Therefore, the European Commission stressed that it is necessary to limit the absolute maximum individual risk for European citizens. This is why the Commission proposes to keep in addition to the ERT a limit value. The new approach combines:

- A relative target for the reduction of ambient concentrations averaged over a wide geographical area. The extent of this reduction could be determined by the balance of costs and benefits. Intuitively higher reductions should be required in more polluted areas, without putting disproportional pressure on these areas and taking into account transboundary aspects. Thus, a percentage reduction would seem appropriate.
- A limit value.

The exposure-reduction approach, including any initiative aimed at improving the accuracy of the exposure-response function, embodies a form of environmental justice, although of a different kind from the ambient air quality standards. As long as there are sources of emission in an urban area, then there will always be differences in exposures due to dilution and dispersion, even if there is uniformity in compliance with ambient standards. If the exposure reduction approach is adopted, and if the reduction amount is required to be the same everywhere, then there will be uniformity in the improvement in exposure, in percentage terms, if not in absolute amounts. In addition, when coupled with a concentration "cap" citizens are guaranteed an absolute minimum standard of air quality to protect them against unduly high risks.

The ERT would provide a better air quality management system than one relying solely on ambient air quality standards. The following benefits (in addition to those already mentioned above) have been identified:

[9] See chapter 9 of CAFE Scenario Analysis Report Nr. 4 Target Setting Approaches for Cost-effective Reductions of Population Exposure to Fine Particulate Matter in Europe available at http://europa.eu.int/comm/environment/air/cafe/activities/pdf/cafe_scenario_report_4.pdf.

- Source-related emissions reductions would contribute more effectively and not just in areas where there are exceedences of limit values.
- No need to modify the ambient air quality standard as time elapses as the emphasis is on reducing overall exposure thus saving administrative resources.
- Proposed approach would complement and "fine tune" overall emission ceilings for a Member State or region, which, if implemented, alone would not have the necessary focus on the improvement of public health; i.e. the total emission ceilings might be achieved with a disproportionately small improvement in public health, depending on the spatial relationship between the emission reductions and the populations exposed.

At this stage, no experience with the ERT is available.

4.2.2. Emission reductions

Emissions are generally a function of the underlying emission generating activity and an emission factor, which depends on the applied technology (including any relevant abatement technology). Emission reductions may aim at the reduction of the activity or may be directed to decrease the specific emissions (often through end of pipe technologies).

Due to the uneven distribution of emission sources, pollutants show spatial gradients. These gradients vary also as a function of the atmospheric lifetime of pollutants. There are therefore considerable differences in the scale of relevant sources. Broadly spoken, for pollutants with short atmospheric lifetimes such as ultrafine particles, NO and NO_2, local sources may dominate the ambient levels. Longer lived species such as $PM_{2.5}$ and CO may have considerable regional and even continental and hemispheric background levels. This has important implications for control options. The contributions from emissions at different scales are shown schematically in Figure 5 for fine PM.

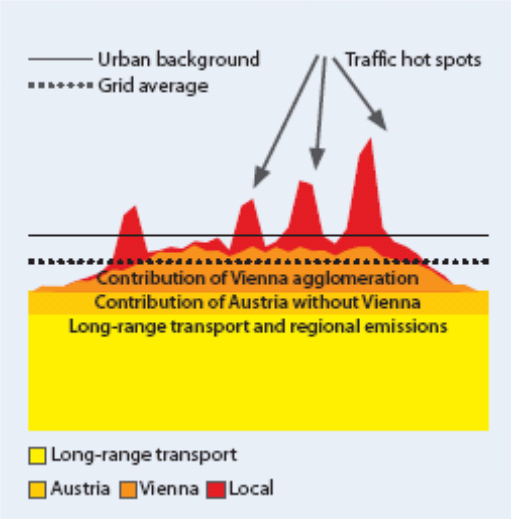


Figure 5: Schematic illustration of different PM_{10} levels in different locations for Vienna (Source: WHO, 2006).

This implies that reduction strategies at a local scale have only a limited scope.

4.2.3. Local emission reduction approaches

Point sources and transportation sources

The most important source related regulations on the EC level (which have also a considerable impact on local air quality) are the continuous tightened EURO standards for mobile sources and the IPPC and LCP for power generation and industrial installations. However, the EURO standards and the LCP are applicable irrespective of air pollution levels. According to the IPPC Directive, measures going beyond BAT may be requested if this is necessary to achieve EC environmental objectives (such as limit values).

Measures to comply with limit values are usually in addition to these regulations. There are numerous possible additional measures for all relevant sectors. Databases containing lists of possible measures to reduce air pollution at a local scale are now available [ref to follow]. These databases often contain estimates for reduction potentials and costs.

The measures reported in plans and programmes under the AQ FWD are summarized in Table 3 and Figure 6.

Table 3: Categories of measures reported from Member States within plans and programmes.

Category: Traffic	
Sub-category	**Measures**
Technical	Emission reduction of cars, buses, trucks, motorcycles, railways, ships, airplanes
Traffic management	Traffic flow management, parking charges, congestion charges, improved cargo logistics, airport traffic management
Public transport	Improvement and promotion of public transport, promotion of bicycle and pedestrian traffic
Traffic restrictions	Measures which restrict traffic in certain areas
Road construction	Construction of by-pass roads, constructive measures which improve traffic flow
Speed reduction	Area or road specific speed limits
Street cleaning	Improved street cleaning, alternative winter sanding
Other	alternative traffic concept, bicycle sharing, car sharing, car pooling, efficient driving training, labelling of low emission vehicles, low emission road surface, promotion of methane fuel stations, mobility planning, promotion of railway cargo transport, restrictions to maintain engines running, restrictions to studded tyres, truck toll, tunnel exhaust cleaning
Category: Stationary sources	
Sub-category	**Measures**
Agriculture	Measures in the area of manure handling and feeding
Construction	Measures to reduce emissions on construction sites
Heating	Improvement of heaters, building insulation, district heat
Industrial	Measures to reduce industrial and power plant emissions
Other	Restriction of open fires, removal of sand surfaces
Category: Regulation and information	
Sub-category	**Measures**
Financial incentives	Fiscal stimulation, emission certificates, financial support of low-emission technology
Information of the public	Information and awareness of employees, pupils and the general public
Change to emission standards	Improvement of emission standards on the European level
Other	---
Category: Other measures	
Sub-category	**Measures**
Energy	Support of alternative energy production, measures to reduce energy consumption
Fuel improvement	Propagation of low-sulphur and low-VOC fuels
Urban planning	Integration of mobility and air quality aspects in urban planning.
Other	Combination of information, incentives and traffic restrictions; procedure of regularly taking and evaluating new measures. reduction of transboundary pollution; planting of trees; construction of a protective wall.
Category: Other activities	
Sub-category	**Measures**
Air quality monitoring	Monitoring of pollutant concentrations
Studies	Emission inventory, emission monitoring, emission study, energy consumption research, exposure study, research program, study on regional transport
Not specified	Measures with unspecified emission reduction, measures which are in the stage of planning
Other	Definition of plans to reduce emissions, resettlement of population.

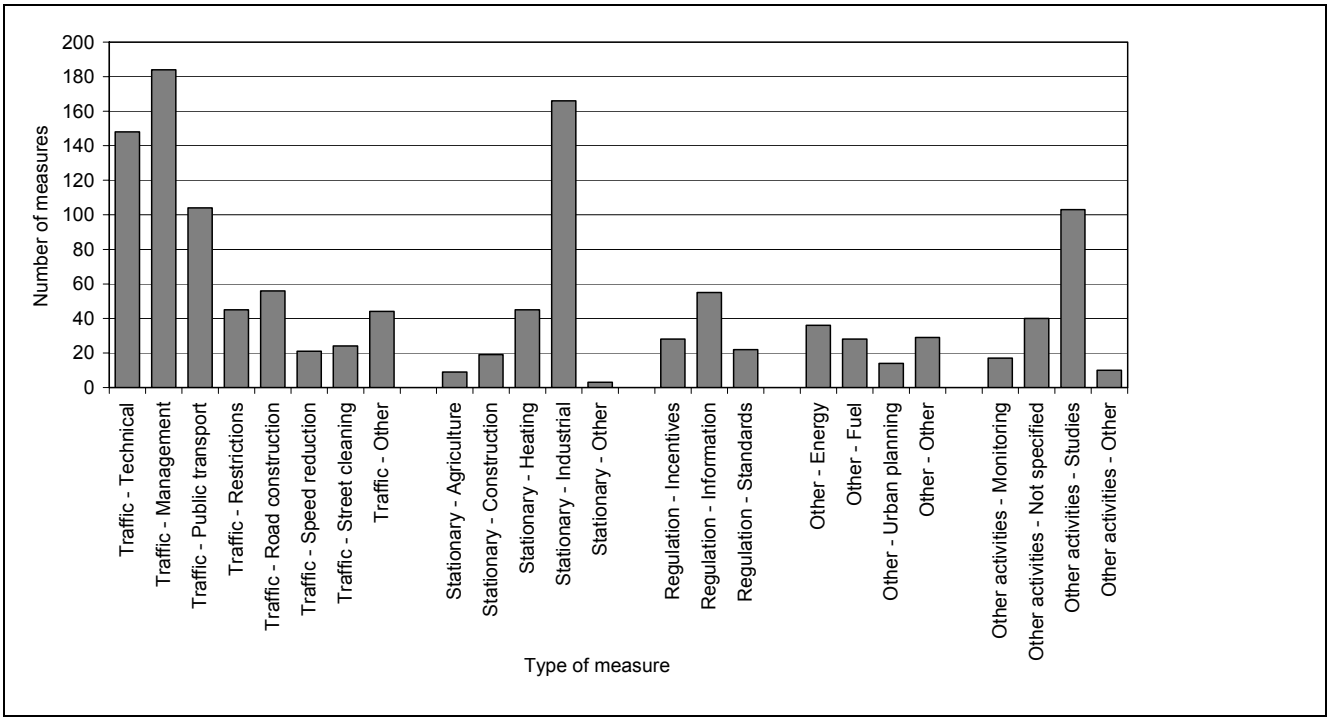


Figure 6: Categories of measures reported by all Member States.

Management of hot spots

As stated previously, limit values apply throughout the territory of Member States. Therefore, efforts to comply with limit values is often focused on hot spot locations (locations in the vicinity of emission sources with the highest pollution levels). As part of the information transmitted by Member States on plans and programmes under the air quality FWD, the authorities have to quantify the area of exceedance (and for traffic hot spots the length of roads). This information is shown in Figure 7, indicating that some plans aim at the reduction of pollution in rather limited areas [ref to follow].

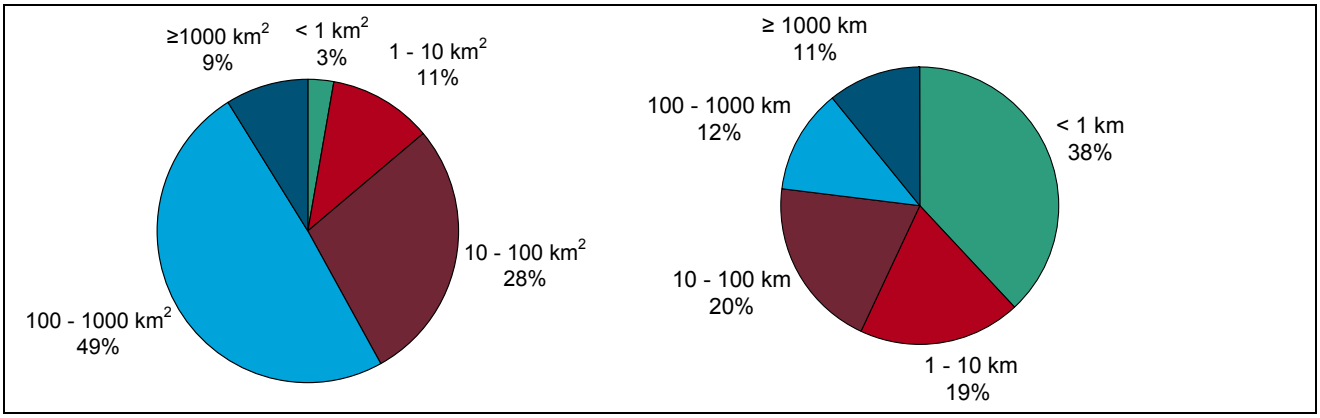


Figure 7: Estimate of the surface area where the level was above the limit value + margin of tolerance in the reference year. Right: Estimate of the length of road where the level was above the limit value + margin of tolerance in the reference year. For 27 % of exceedance situations, no surface area and no length of road was reported.

A recent study on the ex post evaluation of local measures in the EU [ref to follow] concluded that

- Specifically targeted local measures do appear effective in terms of local emissions reductions, air quality improvement and progress towards legally binding air quality limit values, particularly when these schemes tend to be targeted at air quality hot spots (such as low emission zones; fuel bans; traffic flow controls). They also have good benefit to cost ratios, which are similar to or better than for the introduction of European level air quality policies. This provides some initial support for these measures as a complement to further European based legislation.
- The effectiveness of all local measures is very site-specific. It is not possible to simply transfer schemes between locations without consideration of local conditions. Location-specific characteristics of the following key factors determine this effectiveness: background pollutant levels, pollutant formation and transport mechanisms, cultural and economic factors influencing the scale and frequency of emissions from various sectors, legal and informational limitations on the ability of responsible authorities to act.
- The most effective schemes, in reducing emissions and reducing air quality hot spots appear to be those schemes directly focused on air quality improvements. This includes measures such as low emission zones, motorway flow management, smoky vehicles bans, etc in urban areas. Many traditional local transport schemes appear less effective in achieving emissions or air quality improvements, though this is not surprising when these schemes are aimed at other problems (e.g. congestion). However, these latter schemes have other benefits (e.g. travel time benefits, reduced accidents, etc) that are often their primary objective.

4.2.4. Regional and national level approaches

Some measures are usually most effective at a regional or national level. These include many of the source related regulations listed in Table 3 (such as limit values for installations, national speed limits, etc.). In addition, taxes are usually applied on a national level. This includes fuel taxes or a specific tax on NOx, which e.g. is applied in Sweden for stationary sources (see below). These instruments have the potential to affect both the emission generating activities (e.g., mileage of road transport) and the introduction of abatement technologies. Changes in activity usually has ancillary effects (e.g., for road traffic emissions of GHG and noise).

Within the European Community, there is a minimum fuel tax for diesel and petrol. However, the real taxes are often higher and differ by Member States (see Figure 8).

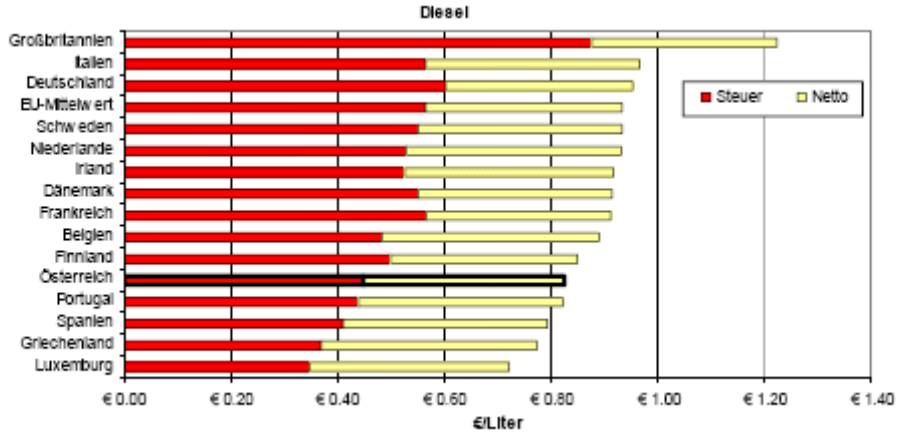


Figure 8: Fuel prices and taxes in different EC Member States; data from autumn 2004.

For Austria, the effects of economic instruments for road transport has been assessed and compared to technical measures (such as retrofitting programmes; speed limits; traffic restrictions for high emitting vehicles) to reduce NOx and PM. Notably, a general road pricing scheme for cars and an increase of fuel taxes were among those measures which brought the largest emission reductions [ref to follow].

There are some other successful examples for the application of economic instruments within the European Community. For example, Sweden has implemented a charge on NOx emissions in order to reduce these emissions cost effectively [ref to follow]. According to the NOx Act the charge is paid for emissions of NOx from boilers, stationary combustion

engines and gas turbines with a useful energy production with at least 25 GWh per annum. The charge is based on actual recorded emissions and is imposed irrespective of fuel used. It is levied at a rate of SEK 40 (about € 4,3) per kg of emitted NOx. To avoid distorting the pattern of competition between those plants which are subject to the NOx charge and those that are not, the system is designed so that all revenue except the cost of administration is returned to the participating plants, in proportion to their production of useful energy. Boilers with high emissions relative to their energy output are net payers to the system, and sources with low emissions relative to energy output are net recipients. This feature of the system encourages the targeted plants to reduce their emissions of nitrogen oxides per unit of energy to the lowest possible level. Since the Swedish Parliament passed legislation introducing the NOx charge in June 1990 the specific emissions have dropped from an average of about 160 milligrams of NOX per megajoule (mg/MJ) of energy input to about 55 mg/MJ, equivalent to 65 per cent.

4.2.5. Transboundary/hemispheric approaches

It has been recognized decades ago that some of the environmental problems linked to air pollution have a strong transboundary component. These problems include acidification (caused by the deposition of oxidized sulfur and nitrogen compounds), eutrophication and ground level ozone. Also fine PM may have a significant transboundary component. This has important consequences for abatement strategies. Since sources and receptors are often located in different countries, multilateral agreements are necessary to combat these effects effectively.

In Europe, the UN ECE Convention on Long Range Transboundary Air Pollution (CLRTAP) provides a framework for emission reduction agreements. Eight Protocols have been signed and entered into force in the last decades. Notably, the CLRTAP played also an important role in promoting research to investigate the sources, transboundary transport and effects of air pollution [ref to follow]. The prime objective of these activities was to substantiate cost effective emission reduction strategies on a European scale. The most recent Protocol, the Gothenburg Protocol, set national emission ceilings for four pollutants (SO2, NOx, NMVOC and NH3) to combat acidification, eutrophication and ground level ozone (all this substances are also precursors for secondary PM). While in the past concrete obligations for controlling emissions were derived solely based on technical and economic aspects or equal emission reduction percentages, this Protocol attempts to quantify specific reduction requirements for the Parties with the aim of achieving certain targets for acidification, eutrophication and ground level ozone. The starting point for negotiating emission ceilings were results form a so called integrated assessment model, which was used to investigate cost effective emission reductions [ref to follow]. The first step was to construct a baseline scenario (including the effect of already decided emissions reduction measures) based on the projected development of emission generating activities. The emissions are translated into ambient concentrations (using the results of a European dispersion model) and effects in the same modeling framework. In a second step, cost effective emission reduction strategies (expressed e.g. as national emission totals in a specific year) can be identified to achieve different environmental improvements. However, the agreed ceilings were in case of the Gothenburg Protocol the result of subsequent political negotiations and did not necessarily reflect cost effective emission reductions from a European perspective.

In the European Community, the Directive on National Emission Ceilings (NEC) has similar objectives and was based on the same integrated assessment model which was also used for the Gothenburg Protocol. The NEC Directive itself does not contain any concrete requirements for sources. It is up to the Member States do identify those sectors where cost effective measures should contribute to achieving the ceilings.

4.2.6. Public education/behavioral/stakeholder engagement approaches

Emissions of air pollutants are often linked to the individual life style of citizens. This includes the choice of the transport mode, and also the use of energy. Many campaigns were launched to influence individual behavior of citizens. It is not easy to find published ex-post reviews of the effectiveness of such campaigns. A small survey among experts in Austria (mainly based on expert judgment) suggests that such campaigns are often limited in their effectiveness. However, there was also a consensus that PR is an important element in increasing the acceptance of the public for new measures.

4.2.7. Strategies for enforcement of regulatory measures

In general, the European Community has a relatively strict system to enforce implementation of Community legislation.

EU legislation has to be transposed and implemented in EU Member States. Transposition and implementation is scrutinized by the European Commission, the 'safeguard' of the Treaty (and secondary legislation). If legislation is not implemented sufficiently, the European Commission may start a so called infringement procedures, which has several steps. At the end, there is the possibility that Member States are condemned by the European Court of Justice, which can also result in considerable fines (which have to be paid by the Member State).

Non-compliance with limit values has also lead to national court cases in different Member States. In Austria, there has been a ruling by an appealing court implying that the authorities might be liable to damages compensation if there is health damage due to access air pollution [ref to follow].

In addition, licensing of new (usually quite clean) plants in areas with air quality in the range or above limit values is usually only possible if this installations have small contributions to air pollution or if there emissions are compensated by other measures. Therefore, there is often a clear interest by industry to promote emission reductions in other sectors in order to avoid non technical barriers in licensing of new plants.

4.2.8. Case study – California

In the post-World War II boom period, California developed severe air quality problems. By the mid-1960s, total oxidant (ozone plus NO_2) levels approached 800 ppb in Los Angeles, and 24-hour-average PM_{10} concentrations exceeded 1800 $\mu g/m^3$ in desert areas and 600 $\mu g/m^3$ in Los Angeles. Although California made significant progress by attaining air quality standards for lead, SO_2, sulfates, and NO_2, and reducing peak ozone levels and PM, there are still many days of unacceptable ozone and particle levels across most of the State. In fact, over 90% of Californians continue to breathe unhealthy air at times.

Mobile sources such as gasoline-fueled vehicles (24 million cars and light trucks for 34.5 million people) and diesel-powered vehicles (1.25 million trucks and buses) play a major role in California's air quality problems. Because of California's proximity to the Pacific Ocean and geography, the meteorology is particularly conductive to generating poor air quality. Los Angeles' pollutant formation potential is the worst in the U.S. due to its unique combination of recirculation patterns, stagnation, inversions, and topography. The Los Angeles Air Basin's carrying capacity (an estimate of the maximum atmospheric burden a region can have and still attain air quality standards) per capita is five times less than Houston's (36 versus 181 lbs VOC and NOx/person/year), which has similar ozone peaks. As a result of the State's poor air quality and large population, California residents receive more than 40% of the nation's population-weighted exposure to ozone values above the national 8-hour standard of 0.08 ppm, and more than 60% of the population-weighted exposure to $PM_{2.5}$ values above the annual standard of 15 $\mu g/m^3$.

California's $PM_{2.5}$ nonattainment areas are dominated by ammonium nitrate and carbonaceous species, derived primarily from mobile sources. Unlike the East Coast of the U.S. and Eastern Canada, California has greatly reduced sulfate levels. This is due to essentially removing sulfur from diesel fuel and gasoline, and the use of natural gas for electrical generation.

PM is California's greatest challenge, as is responsible for over 6500 premature deaths per year (about 10 times greater than ozone and 20 times greater than cancer cases from know toxic air contaminants). Air pollution is estimated to cost Californians $51 billion per year – $4 billion per year in direct medical costs, with the remainder the value assigned to premature death. CARB calculates that California gains $3 in health benefits for every $1 it currently invests in air pollution control.

The policy that people of all races and incomes need equal protection from the detrimental effects of pollution is known as environmental justice, and has emerged as an important issue in California over the past five years. The debate focus on the need for community controls in addition to statewide measures. In California, people who live near busy roads are disproportionately Hispanic, Asian, and black, and from low-income families. Several Dutch studies found reduced lung function and higher asthma, hayfever, and wheezing rates for children living near heavy truck traffic. A study by Ralph Delfino found that Hispanic children with asthma symptoms had higher breath levels of benzene, a marker for traffic.

California is also concerned about indoor sources of air pollution. Kirk Smith has calculated that a typical pollutant release is a thousand time more likely to go down someone's throat if it occurs indoors rather than outdoors because people are usually indoors, near the sources. While the sources and risk reduction measures are known, CARB and other agencies have very little authority in this area.

California has adopted many emission standards more stringent than the U.S. standards. This includes light- and medium-duty vehicles – exhaust and evaporative standards, handheld and non-handheld small off-road equipment, personal watercraft, in-board motors for boats, and portable engines. Planned regulations for light-duty vehicles include a parts replacement program and improvements to the Smog Check program (i.e., more vehicles to test only, loaded mode testing for gasoline trucks, evaporative emission control test to detect liquid leakers). For forklifts and other large spark-ignited equipment, CARB is working on lower emission standards for new equipment as well as in-use reductions through catalyst retrofits. For heavy-duty vehicles, CARB has a broad range of controls to reduce emissions from both new and in-use vehicles (i.e., OBD, reduced idling, chip reflash, gasoline tanker vapor recovery, in-use inspections in EJ areas) and must go beyond those strategies to get additional reductions. For off-road compression ignition equipment, although California is preempted from controlling a significant (~80%) of this equipment, it is a huge source of emissions and large reductions are needed. California will work with the U.S. EPA to establish more stringent nationwide standards for HC, NOx, and PM from off-road compression ignition engines, and implement in-use strategies to get additional reductions. For marine engines, California plans to get reductions from existing harbor craft through cleaner engines and fuels. For the ports, reductions from land-based port emissions are planned, including cargo handling equipment and locomotives, heavy trucks, and dredges. CARB will set standards for additives to control engine deposits.

California has a goal of reducing diesel PM by 75% during this decade and 85% by 2020. This is being achieved with new emission standards, cleaner fuels, retrofits of existing engines, and enforcement programs. CARB and the U.S. EPA have adopted new vehicle standards that reduce emissions by 90% beginning in 2007. CARB will require aftertreatment on every diesel source where it is technically feasible. Low-sulfur fuel is required, as well as cleaner fuels like CNG and measures to reduce or eliminate idling. Enforcement programs are used to minimize the effects of tampering and wear, especially in environmental justice communities.

California considers greenhouse gases to be ozone and particle precursors as climate change can affect urban air pollution. In 2004, CARB adopted regulations that reduce greenhouse gases emitted by passenger vehicles and light trucks, although this measure is being litigated by the automotive industry. Reductions in greenhouse gases on the order of 30% can be achieved for all vehicle types using technologies already deployed in production vehicles. The costs are on the order of a few hundred to a thousand dollars and are more than offset by reduced operating costs of up to $5000. Gas-electric hybrid vehicles and other technologies can achieve greater reductions.

California set the bar for U.S. EPA and European Union emission standards that are now being adopted by many developing countries, particularly in Asia. Most of the world's population benefits from the fact that over 70% of the vehicles worldwide must comply with cleaner emissions standards.

These policies have resulted in significant emission reductions and air quality improvements over the years. At least 50% reductions have been achieved in both the stationary and mobile source emission categories over the past 20 years, and will continue their downward trend (see figure 9). These emission reductions have been achieved despite a doubling in vehicle miles traveled and a 50% increase in population. California's economy grew by 75% despite the $10 billion cost per year for air pollution measures adopted since 1990.

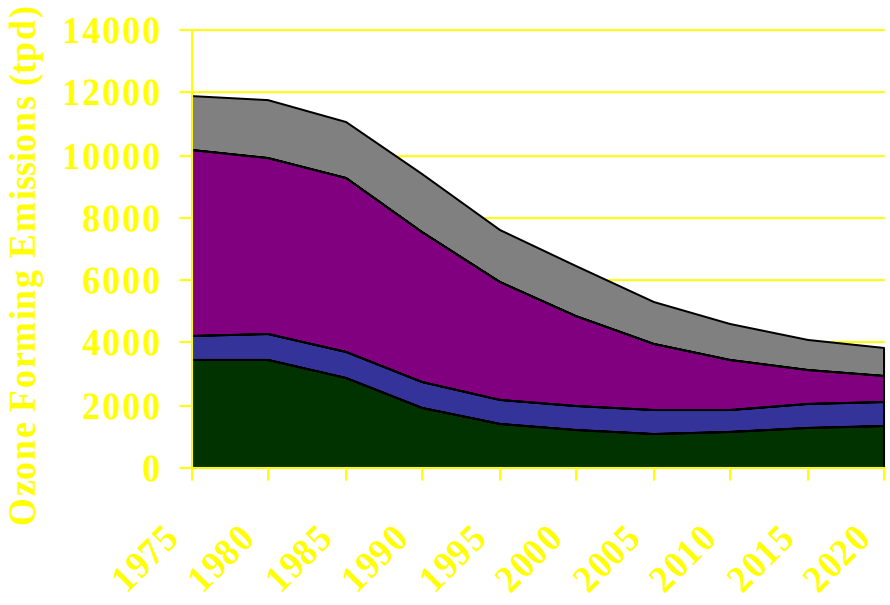


Figure 9:

Air pollution levels have improved dramatically. The health-based standards for lead, NO_2, SO_2, and sulfates have all been attained, CO is very close, and peak ozone levels have dropped 75% relative to levels in the mid-1960s. California has also had success with PM_{10} and air toxics.

CARB's technology-forcing emission standards have resulted in major advancements in emission control technologies. Today's cleanest passenger car emits less than one percent of ozone precursor emissions compared to the emissions from a car produced in 1960. California's successful introduction of many emission control programs has served as the basis for many similar U.S. programs. Through decades of emission control success, these programs have significantly improved California's air quality, despite more than doubling the number of people and tripling the number of vehicles over the last four decades.

Two of the keys to CARB's success are the technical evaluations that go into its regulation development and the very open public process. CARB develops new emission test methods, and in some cases, proves that more stringent emission standards are achievable by funding or conducting technology demonstrations. It encourages participation by all stakeholders, including the public, industry and communities that may be impacted by air pollution disproportionately from others. CARB meets with many stakeholders to hear concerns and to provide a mechanism for addressing their issues. It holds workshops that solicit suggestions and comments on initial issues. The technical data and assumptions are published in advance of the workshops. Regulations are first proposed in an initial report and additional workshops are held for public comment. CARB change its proposal once significant issues are raised that warrant a revision. Once the regulation is adopted, it issues a formal response to all issues raised. The public has a chance to air their concerns directly to our Board members who are appointed by the Governor and represent different professions and regions in California. The Board reviews the technology and enforceability of regulations when necessary to make sure that the regulations meet the expectation held at the time of adoption.

Figure 10 shows a 29-year timeline of the cost-effectiveness of various vehicle and fuel regulations, in dollars per pound of ozone precursor. Most measures have cost less than $2 per pound, which is considered to be quite reasonable in comparison to a benchmark of $5 per pound for stationary and area source control measures. Due to technology advancements, these costs have stayed fairly steady. CARB considers economic impacts of its regulations on California businesses and individuals, and regulations do not advantage or disadvantage California manufactured products over products manufactured elsewhere in the U.S. or in the world.

A recent study by EBI concluded that the air pollution control industry in California generated $6.2 billion in revenues and employed 32,000 people in 2001. The U.S. figures are $27 billion in revenues and employment of 178,000 people.

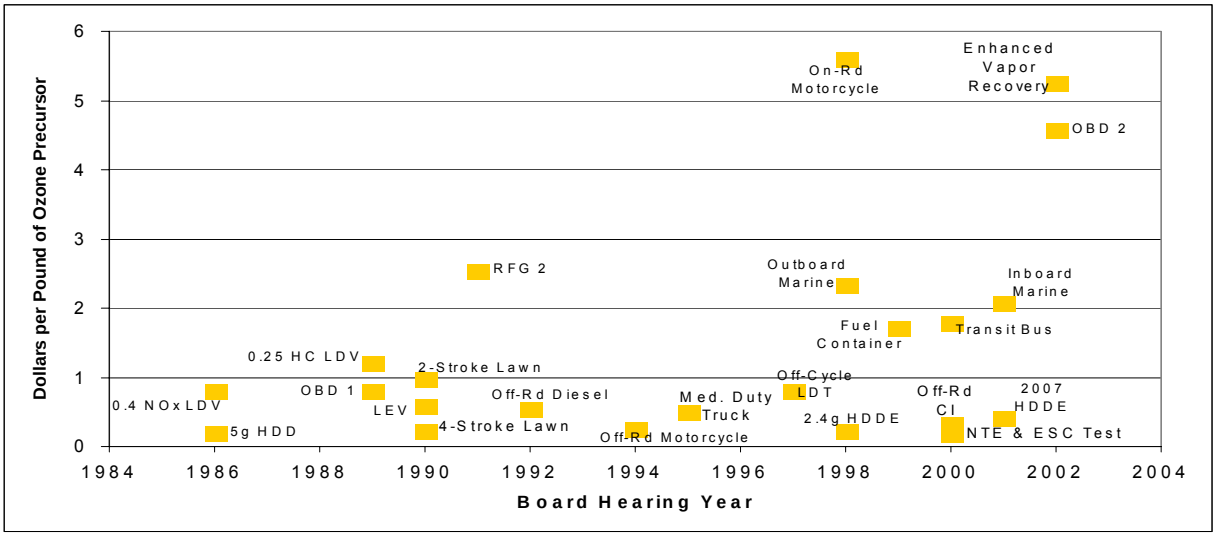


Figure 10:

4.2.9. Case study – Mexico City, Mexico

[To be written – use Molina et al. 2004 as a starting point and focus on success of lead and ozone reduction measures, unintended consequences of "No Driving Day" program, and efforts to improve public transport.]

4.2.10. Case study – Toronto, Canada

[To be written – use Molina et al. 2004 as a starting point and focus on SO_2 and NO_2 success story, ozone and transboundary control agreement, stationary source emissions trading, and newly recognized role of local traffic pollution from Jerrett studies.]

4.3. Which air quality policy measures have been shown to provide cost-effective improvements in air quality and public health? How transferable are these strategies to other locations? What lessons have been learned from control strategies in various sectors?

4.3.1 Europe

A UK consultant investigated the effects of short term and local measures to reduce air pollution (AEAT, 2005). The study concluded that .. *it is extremely difficult to find reliable and consistent data on the ex post costs, and the ex post benefits (particularly in relation to emissions and air quality), of local measures, they were able to draw some general conclusions*.

Probably it is even more difficult to assess and compare consistently the ex post costs, and the ex post benefits for local, regional and national measures. However, a few conclusions can be drawn.

- It is generally accepted that air quality management has been a success story in the EC. Member States of the European Union spend large sums for air quality protection, mainly triggered by source related legislation.
- The EURO standards are seen as an essential element of AQ protection which ensured that the continuously increasing road transport is emitting less pollution than a few years ago even though the EURO standards proofed to be less efficient in practice than expected. However, there is still no legislation in force forcing external costs of road traffic to be internalized.
- The concept of integrated pollution prevention for industrial installations including the application of BAT for new and existing plants is also widely accepted. However, there is still a debate if legally binding emission limit values are warranted for those installations.

- Air quality limit values are one important element of air quality policy. The inherent focus on the most polluted sites has been discussed recently and lead to the proposal to supplement the limit value approach by an exposure reduction target (ERT), which sets objectives for relative improvements (more or less irrespective of absolute pollution levels) for urban background locations.
- Energy efficiency will become increasingly important (primarily due to concerns about energy prices, security of supply and climate change).

There is robust evidence indicating that air pollution still causes severe health and environmental damage [ref to follow]. Since many measures to reduce air pollution are already in force, this leads to a situation where additional measures are getting increasingly expensive, while the reduction potentials get smaller and smaller. As a consequence, additional measures need to be well justified. This implies that any additional measures need to be based on robust science. This includes a profound knowledge on

- the sources of air pollution,
- the atmospheric dispersion,
- ambient levels,
- effects of air pollutants
- as well as costs and reduction potential for abatement measures,

Therefore, recent legislative proposals in the EC have been accompanied by impact assessments comparing the cost and benefits of these proposals.

4.3.2 U.S., California, Canada and Mexico (to follow)

4.4. Conclusions

In response to severe air quality problems, Canada, Mexico and the United States imposed comprehensive emission controls, but growth in population, vehicle miles traved and industrial activity prevented attainment of health-based ambient air quality standards. Emissions of VOC and CO (and to a lesser extent NO_X) from new passenger vehicles were reduced by a factor of a hundred in comparison to pre-control vehicles. The United States adopted emission standards for 2007 and subsequent model year heavy-duty engines that represent 90% reductions of NO_X and PM compared to 2004 model year emission standards. Implementation of reformulated gasoline and diesel fuels resulted in further reductions. Stationary source NO_X and SO_X emission standards were reduced by at least a factor of ten since 1980. Small off-road engines, architectural coatings, consumer products and solvents are also targeted for large emission reductions. While some air quality problems have been eliminated or greatly reduced (i.e., lead, NO_2, SO_2), particulate matter and ozone levels remain high in many North American cities, resulting in thousands of deaths per year and increased disease rates. In response, air quality management agencies are developing innovative approaches, including regulation of in-use emissions, reactivity-based VOC controls and exposure-based prioritization of PM controls. Several cooperative, multi-national efforts (e.g., NARSTO) have advanced the science and technology of air pollution control, and begun to address transboundary issues. Newly recognized challenges also need to be integrated into air quality management programs, ranging from the microscale (e.g., air pollution “hotspots”, ultrafine particles, indoor air quality) to global scales (e.g., climate change mitigation, international goods movement).

CHAPTER 5. Emerging Challenges and Opportunities in the Development of Clean Air Policy Strategies

Martin Williams and Quentin Chiotti

This chapter will discuss future challenges for air quality management on local urban scales, and will also extend the discussion to wider spatial scales and consider the important links between air quality and climate change policies.

5.1 Urban Air Quality Management

In many areas of the developed world air quality management is a fairly mature subject. There have been some recent developments which are of significance to these areas, as well as to developing countries where air quality management systems may be at an earlier stage. A particularly significant development in this context has been the publication of a global update of the World Health Organisation guidelines for air quality (http://www.euro.who.int/air). The significant feature of this publication in the current context is that it explicitly addresses the problems of air quality management in developing countries. An example of this is the way the WHO has dealt with particulate matter (PM). In its previous publication of air quality guidelines in 2000, the WHO recommended guidance for risk management in the form of exposure-response relationships and suggested that air quality managers quantify the risks relevant to local levels of PM and make the risk management decisions on control policies appropriate to whatever balance of risks and benefits was felt appropriate. While this approach is used in some of the developed countries and regions, in the 2006 update there was a body of opinion presented to the WHO, largely from the developing countries, which felt that this level of detail was not particularly helpful. Accordingly, the 2006 update has now returned to the older approach of recommending a guideline expressed as a concentration and averaging time, together with a series of three successively more stringent 'Interim Targets'. WHO recognizes this approach as particularly helpful for developing countries whose levels of PM are currently quite some way above the guidelines themselves.

There are several significant implications of this approach for air quality and risk managers. Firstly, the move away from recommending an exposure-response coefficient or function for a non-threshold pollutant such as PM could be viewed in some quarters as not allowing any scope for national or regional air quality managers to undertake their own risk management and to formulate their policy targets considering local prevailing levels of pollution as well as the predominant socio-economic climate. This latter approach is the way policies for the management of PM levels are handled in the UK. For example, expert advisory groups under the umbrella of the Department of Health have devoted considerable intellectual resource to assessing the literature and recommending exposure-response coefficients, together with likely uncertainty ranges, so that in formulating policy measures, the Environment Department can undertake the appropriate quantification of the effects on public health and where possible, cost-benefit analysis and proceed with policies which respect the three 'pillars' of sustainable development. This process can still be undertaken of course, but there is now no international body recommending an exposure-response relationship to underpin such risk management analyses. (It is worth noting however that the UNECE/WHO Task Force on Health within the Convention on Long Range Transboundary Air Pollution did assess the then current literature and recommended exposure-response relationships in 2004).

To carry out risk management analyses of the type described above requires a fairly well developed scientific and economic infrastructure in the air quality management system. The assessment of potential new control policies proceeds according to the following chain:

Possible policy scenario→projected emissions→atmospheric concentrations/exposures→health and/or environmental effects→monetised damage costs vs control costs

The first stage requires estimates of the effect on emissions of the proposed policies, the second requires, at the very least, a robust dispersion/chemical model of atmospheric transport, the third requires one or more exposure-response functions and the final stage requires economic analysis of the possible monetary values associated with the effects of the policies. This chain therefore requires a certain maturity of development in the air quality management system. The

discussions in the WHO forum which produced the global update felt that this was not appropriate to less well developed systems and therefore chose to recommend the Guideline and Interim Targets for PM referred to above. The possible criticism that these concentration levels were somewhat arbitrary (in the sense that they were not derived from a balance of costs and benefits as might be thought appropriate for a non-threshold pollutant) was addressed by choosing the guideline for $PM_{2.5}$ as the lowest level at which total, cardiopulmonary and lung cancer mortality were shown to increase with more than 95% confidence in the ACS study of Pope et al. (1995) The highest value (i.e. least stringent) Interim Target, denoted IT-1, was chosen as the highest observed value in the studies on long-term health effects. The two values in between, IT-2 and IT-3 were essentially arbitrary values chosen between the other two extremes, the change in risk on moving from one level to the next was quoted, using essentially the exposure-response relationships derived by Pope et al in the ACS study.

While in the short term this approach will no doubt be helpful to developing countries in formulating policies, in the longer term it may well need revision as the policy process matures and both sides of the debate become more knowledgeable and sophisticated. There will be further challenges for developed countries in future too, as they approach the lower end of the IT/guideline scale produced by WHO. The potential difficulty of deriving conclusions of policy relevance from epidemiological studies as levels of PM decrease has been noted in Chapter 2 and this is likely to complicate the policy process. For example, let us suppose a country or region approaches the guideline for $PM_{2.5}$ ($10\mu g/m^3$ annual mean). Does it stop once the level is reached or will there still be robust epidemiological evidence for the absence of a threshold that suggest further improvements might be warranted? If so, what does one make of the guideline? Questions of this kind will make the management of PM a subject of some difficulty for years to come.

5.2 Novel Approaches to Air Quality Management

The final paragraph above serves as a useful introduction to the topic discussed in this section, which was also prefaced in Chapter 2. This concerns the problem of how one manages air quality in areas where standards or guidelines are already met. This is particularly important for non-threshold pollutants like PM, and highlights the problems associated with managing air quality via a single air quality standard or guideline. This problem becomes increasingly apparent as ambient levels are successively reduced and the easier control measures have been taken.

Experience in the UK and elsewhere in Europe has shown that for non-threshold pollutants, single limit values or standards may not on their own be the most appropriate way of managing air quality, particularly in areas where existing air quality management systems are mature. This has encouraged the European Commission to propose a new, additional concept, the exposure reduction target (ERT) (which has not entered into force yet, even though the concept is in principle supported by the Council and the European Parliament). The following short description of the basics of the concept is derived from a non-paper issued by the European Commission.

The existing legal framework of the Air Quality Framework Directive and its Daughter Directives require complete compliance, meaning that limit values must be met everywhere. As such, a conventional air quality management strategy would implement measures according to their cost-effectiveness so as to reduce the areas of exceedence of these limits. Such a strategy would deliver increasingly smaller areas above the limit values. In the remaining areas, it may well be that reaching complete compliance is very difficult and costly. In addition, there would be little incentive to improve air quality where limit values are already respected.

For pollutants with no effect threshold, such as $PM_{2.5}$, it will generally be more beneficial for public health to reduce pollutant concentrations across the whole of an urban area as benefits would accrue from reductions in pollution levels even in relatively “clean” areas.

Therefore, an ERT was proposed for fine particulate matter $PM_{2.5}$. $PM_{2.5}$ is responsible for significant negative impacts on human health. Further, there is as yet no identifiable threshold below which $PM_{2.5}$ would not pose a risk. Advice from the WHO suggests that it is justified to assume a linear response linking exposure to $PM_{2.5}$ to adverse effects. This advice should apply both in “clean” as well as in “polluted” areas. The exposure reduction concept entails a reduction in the exposure of a larger part of the population compared to the limit value approach which affects (as we approach complete compliance) a smaller number of people. As such, the overall improvement in public health comes at a higher

cost with limit values. A Commission Working Group has looked at this issue and concluded that exposure reduction would be a more cost-effective way of reducing air pollution (Amann et al., 2005). [10]

Environmental Justice

There is also an issue of environmental justice. The European Commission has stressed that it is necessary to limit the absolute maximum individual risk for European citizens. This is why the Commission proposes to keep in addition to the ERT a limit value. The new approach combines:

- A relative target for the reduction of ambient concentrations averaged over a wide geographical area. The extent of this reduction could be determined by the balance of costs and benefits. Intuitively higher reductions should be required in more polluted areas, without putting disproportional pressure on these areas and taking into account transboundary aspects. Thus, a percentage reduction would seem appropriate.
- A limit value.

The exposure-reduction approach, including any initiative aimed at improving the accuracy of the exposure-response function, embodies a form of environmental justice, although of a different kind from the ambient air quality standards. As long as there are sources of emission in an urban area, then there will always be differences in exposures due to dilution and dispersion, even if there is uniformity in compliance with ambient standards. If the exposure reduction approach is adopted, and if the reduction amount is required to be the same everywhere, then there will be uniformity in the improvement in exposure, in percentage terms, if not in absolute amounts. In addition, when coupled with a concentration "cap" citizens are guaranteed an absolute minimum standard of air quality to protect them against unduly high risks.

The ERT would provide a better air quality management system than one relying solely on ambient air quality standards. The following benefits (in addition to those already mentioned above) have been identified:

- Source-related emissions reductions would contribute more effectively and not just in areas where there are exceedences of limit values.
- No need to modify the ambient air quality standard as time elapses as the emphasis is on reducing overall exposure thus saving administrative resources.
- Proposed approach would complement and "fine tune" overall emission ceilings for a Member State or region, which, if implemented, alone would not have the necessary focus on the improvement of public health; i.e. the total emission ceilings might be achieved with a disproportionately small improvement in public health, depending on the spatial relationship between the emission reductions and the populations exposed.

Hemispheric Air Pollution Transport and Policy Challenges

In the last few years there has been a growing recognition that transport of air pollutants (as opposed to long-lived greenhouse gases) can occur between continents, particularly in the northern hemisphere. This presents a challenge to the scientific community but also to the policy maker. One step forward in understanding this problem, initially from a scientific point of view was the establishment in December 2004 of the Task Force on Hemispheric Air Pollution within the Convention on Long Range Transboundary Air Pollution (CLRTAP) of the United Nations Economic Commission for Europe. The discussion in this section is based largely on a paper produced by the co-chairs of the Task Force, Terry Keating (US EPA) and Andre Zuber (European Commission) (Zuber and Keating, 2005)[11].

There is well-documented evidence for the intercontinental transport of ozone, particles, and their precursors, as well as mercury and some persistent organic pollutants. Emissions from one continent can influence air quality in another through an increase in the overall hemispheric burden of pollution and through discrete episodic flows of enhanced levels of pollution. This latter contribution can clearly vary with location, season and the pollutant concerned.

[10] See chapter 9 of CAFE Scenario Analysis Report Nr. 4 Target Setting Approaches for Cost-effective Reductions of Population Exposure to Fine Particulate Matter in Europe available at http://europa.eu.int/comm/environment/air/cafe/activities/pdf/cafe_scenario_report_4.pdf.

[11] Available at http://www.unece.org/env/documents/2005/eb/EB/Inf.Doc.05.Hemispheric%20Transport.pdf

There have been observations of discrete intercontinental ozone transport events made at mountain top sites and by aircraft, but the more important influence on ground-level ozone concentrations have been an apparent increase in the hemispheric burden in the troposphere. This background level is controlled primarily by global NO_x emissions and to a lesser extent by methane and carbon monoxide. For aerosols and their precursors, episodic flows appear to be the most important influence. The episodes are often natural events such as dust storms, volcanic activity or fires. For mercury, the export of emissions into the free troposphere contributes to a hemispheric or global pool of mercury. Influenced by this global pool, mercury deposition patterns are thought to be more related to patterns of emissions and precipitation than to transport events. Some persistent organic pollutants may be transported long distances as aerosols and remain where they are deposited, while others may revolatilize or become re-entrained and travel further in successive hops to reach environments far from the sources. A classic example is the presence of such pollutants in the Arctic.

Although intercontinental transport can be demonstrated using observations and measurements, it is important, not least for future policy assessment reasons, to understand how changes in emissions in one continent influence air quality in another. This requires a predictive model that can account for the non-linearity and complexities of the atmospheric system. Ideally this simulation should be performed with an integrated system of models capable of linking the local, regional, hemispheric and global scales. Models exist which are capable of simulating these phenomena but more evaluation and development are required before they are applicable with confidence to policy questions.

Although current models are uncertain, the magnitudes of impacts are significant enough to suggest that further investments should be made to better characterize intercontinental transport and the potential impacts of emission changes. For ozone, it has been estimated that such transport contributes between 1 and 10 ppb to average surface ozone concentrations in Europe, North America and Asia. Even at the low end of this range, this contribution may offset the benefits of local air pollution controls, and, depending on whether or not ozone is thought to have a threshold for adverse human health effects, it could also have important public health impacts.

For aerosols, current models and observations suggest small but significant contributions (up to $2 \mu g/m^3$ $PM_{2.5}$ annual mean) of intercontinental transport of anthropogenic pollution. Impacts on an episodic basis can be even larger (10 to 100 $\mu g/m^3$ PM_{10} maximum daily concentrations). However, such episodic events are primarily associated with wildfires or wind-blown dust. For mercury, it has been estimated that intercontinental transport may contribute between 10 and 75% of the total deposition to the different continents in the northern hemisphere, with individual continents or regions providing between 1 and 40% of observed deposition. It should be stressed that these estimates are uncertain and are based on preliminary work. Further developments are under way to attempt to improve our understanding via improved descriptions of the transport processes, emission inventories and additional observational data.

Intercontinental transport will pose significant problems not just for the scientific community but also for the policy makers. It is reassuring that the first steps in addressing the scientific issues in an institutional framework were made by the CLRTAP which has a very strong track record in the use of science in a policy context. The Task Force on Hemispheric Transport of Air Pollution will look more broadly than the current scope and coverage of the Convention however. It will attempt to build upon a wide variety of existing research efforts in individual countries and internationally, including IGBP/IGAC, UNEP, the World Bank, GEOSS, WMO, and ACCENT within the EU. The EMEP Centres of CLRTAP will also contribute to this effort. Of particular interest is the establishment of contacts and co-operation from scientists and experts outside the UNECE regions including those from Asia, Africa and Central America.

In the longer term, there may well be a need to integrate this scientific endeavour with policy needs. While the Task Force is at present engaged in addressing the scientific questions, thought is being given elsewhere to the policy context via the work of the Global Atmospheric Forum of IUPPA. The Forum recognizes that a number of regional agreements already exist and rather than attempting to set up some form of global instrument (for example along the lines of the Kyoto Protocol) it may be more productive, at least in the early stages of development of thinking on intercontinental transport of air pollution, to attempt instead to foster collaboration and interaction between these regional instruments. The instruments of particular importance here are the UNECE, the EU, the network of the Central Asian Republics, the Male Declaration in South Asia, the Asian Brown Cloud network, the Clean Air Initiative Asia, EANET in East Asia, APINA in Africa, IANABIS the Inter-America network for atmospheric/biospheric studies and the Canada/US Air Quality Agreement. Two of the principal aims of the Forum are to (i) provide a framework for dialogue and co-operation between these networks and related organizations on the practical challenges facing them and for developing joint projects and (ii) to encourage harmonization of systems and approaches in key areas to facilitate co-operation at intercontinental, hemispheric and global scales, to encourage the establishment of new networks in areas where none exist and to encourage capacity-building in areas where resources currently constrain action.

An important area of the Forum's work will also be to provide a forum for debate on common issues and important here of course is the interaction between climate change and air pollution and the policies and institutions needed to tackle pollution at the hemispheric and global scales. This will pose challenges to policy makers and to institutions in the future. Some aspects of the issues surrounding the interaction between climate change and air quality are addressed in the next section.

Air Quality and Climate Change

There are a number of emerging challenges and opportunities that are important to consider in the development of clean air policy strategies. Much of the integrated focus to date has been in the area of atmospheric chemistry, exploring the integration of air quality and climate change, with less emphasis on the specific emission reduction technologies and measures that will reduce emissions of both air pollutants (including air toxics) and greenhouse gases. There are also important linkages to explore between mitigation and adaptation measures, although in the latter context this is probably more for climate change than for air quality.

In North America, the integration of air issues has been somewhat constrained by a nagging ongoing debate around climate change science and whether actions to reduce GHG emissions are actually necessary, despite the findings of the IPCC, the National Academy of Sciences, among other reputable science-based agencies and initiatives. The fact that the United States has yet to ratify the Kyoto Protocol, and Canada's relatively poor record in reducing greenhouse gas emissions (currently 25 percent above 1990 levels) demonstrates the lack of commitment for taking meaningful actions. Nonetheless, there is growing evidence and concern that climate change is real, is already happening (in some cases/areas occurring at a pace more quickly than previously projected), and that the impacts and effects will be severe. Unlike the "co-benefits" research that occurred in the years leading up to the ratification of the Kyoto Protocol, the linkages between climate change and air quality will begin to move from the theoretical (discussion) to the practical (and applied) level more quickly than many anticipate. It is already happening in some countries. For example, the UK has analysed the effects on carbon emissions of measures to achieve air quality objectives beyond 'business as usual'in a recent review of the UK Air Quality Strategy (DEFRA, 2006) and the recent publication of the UK Climate Change programme considered the impacts on pollutant emissions of the climate change measures. Some of these measures will be synergistic, leading to cumulative benefits for both air quality and climate change, but some may be contradictory, leading to conflicting outcomes.

In proposing emission standards for vehicles which will probably require aftertreatment with a small fuel economy penalty, the European Commission has tacitly accepted the small disbenefit to GHG emissions for the large improvements in particulate emissions that will result. The EU, including the UK, have been talking about reducing GHG emissions by 25 per cent below 1990 levels by 2020, and there is growing acceptance by environmentalists, governments and even industry that a global reduction of 60-80 per cent is necessary by 2050 if we are going to keep CO2e concentrations in the atmosphere below levels that would cause a 2C degrees increase in global temperatures (and avoid dangerously interference with the earth's climate). With such a huge emission reduction challenge, the need to link air issues together is both necessary and unavoidable. The benefits to air quality of these longer term plans were explored in the recent UK Air Quality Strategy Review referred to above (see DEFRA, 2006, Chapter 6). In Canada, the current Federal Government is planning to introduce a Clean Air Act, and Made-in-Canada plan to address climate change largely through air quality measures. However, in anticipation of these initiatives, environmentalists are concerned that the proposed measures may obfuscate public attention away from taking meaningful actions on climate change. Indeed, the introduction of a tax credit for transit passes and proposed commitments to a 10 percent ethanol requirement in gasoline suggests that much more aggressive measures are necessary if Canada is to meet its Kyoto commitments. Furthermore, given past experience with environmental legislation (e.g. Canadian Environmental Protection Act), there are legitimate concerns that the new Clean Air Act may take up to 5 years before it is passed, an unacceptable period of time in terms of the urgency to deal with climate change today, rather than tomorrow.

From a broad context it is important to consider at least four aspects of the linkages between climate change and air quality:

1. the chemical/atmospheric interactions between climate change and air quality (how climate change will impact local air quality, and how air quality and emissions of particulates/aerosols affects climate change at a regional level);
2. actions that directly reduce emissions of GHGs and other air pollutants (e.g. fuel switching, best available technologies, renewables – however some BATs that reduce air pollutants may actually increase GHG emissions);

3. actions that indirectly reduce energy use and emissions (e.g. efficiency, conservation, pollution prevention, land use and transportation planning); and
4. actions that are both mitigation and adaptation; that is measures that reduce emissions and reduce vulnerability by enhancing adaptive capacity (this issue has received very little attention in the climate change literature). One example is the adoption of community-based energy systems such as combined heat and power, and wind power projects that both reduce emissions and reduce vulnerability to a catastrophic system-wide failure of the energy grid.

There are also other air issues to consider, e.g. HAPS, acid rain, particularly in terms of the cumulative impacts and effects, but the primary focus for this discussion are the links between air quality and climate change. It should be noted that some sectors or regions may also be subject to emission reduction controls or targets due to non-human health effects, such as emissions causing acid rain which continue to have significant impacts on aquatic and terrestrial ecosystems.

The Air Waste Management Association recently dedicated an entire issue to the linkages between air and climate change, but it was almost exclusively tied to the science and chemistry rather than focusing on specific locations where it needs to be adopted, which sectors are more suitable for such measures, or what technologies are likely to generate the greatest benefits (EM, Oct. 2005). A quick scan of the issue illustrates what we know so far about the science and the challenges that remain.

Of note, Pennell et al. (2005) identify the following significant interactions from a scientific perspective: emissions – which are highly dependent upon future economic growth, technological change, and energy use, recognizing that we need to be moving towards a zero net carbon emissions future; atmospheric processes and effects; modeling and simulation; monitoring; and policy making. The latter is obviously most germane to this discussion, but the authors do not go beyond concluding that the issues need to be addressed together, that some policy analyses are beginning to focus on "co-benefits", and that research activities in both areas are beginning to converge, but are relatively absent in day-to-day environmental policy and technology management decisions. To complicate matters, Prinn and Darling (2005) argue that managing air quality in a way that supports a climate-stabilization policy could be more difficult than one would think. While this is a valid concern (and indeed a challenge), the authors provide few insights towards resolving the basic conundrum of addressing these issues together: does addressing air quality issues through actions that reduce greenhouse gas emissions produce a broader suite of benefits and outcomes, than addressing climate change by reducing emissions of other air pollutants?

Ten years ago the challenge of addressing air quality and climate change issues together was noted by Pearce et al. (1996) in the IPCC Second Assessment Report. They noted that the question of secondary benefits from carbon abatement should be distinguished from the more comprehensive issue of the optimal abatement mix with respect to all pollutants. In the case of the Kyoto Protocol, the argument has largely been driven by the implicit primacy of the greenhouse problem, with improvements in air quality viewed as welcomed side effects, rather than considered in their own right. One suspects that at this meeting, most would agree that this approach is not necessarily the best way to proceed, and perhaps each pollutant (and air issue) should be assessed (and measures adopted to reduce emissions) in proportion to the environmental damage that it causes. Interdependencies matter, as does location, and greenhouse gas emission reduction measures should be concentrated in places where the joint benefits of reducing all emissions is highest.

Similar observations are found in (Air Quality Expert, 2005) where 6 key questions are raised that address 4 different areas related to air quality and climate change . Not surprisingly, most of the focus is on atmospheric science, and similar to the AWMA issue only gives cursory treatment to the implications for emissions and control options and concludes with the ominous observation that synergies and trade-offs exist in technical control measures, and that there is a need for integrated assessments across sectors and across effects. These last observations are precisely the issues which policy makers have to grapple with in the real world. They need to be recognized, explored, analysed and managed. There can be no hiding from them, nor denying their existence simply because they are inconvenient. As noted above, we are already facing some of them. With respect to the main points, the first three questions are worth closer consideration, dealing with the impact of climate change on air quality (question 1) and the impact of air quality on climate change (questions 2 and 3).

Question 1: How could the likely impact of climate change on the general weather patterns and emissions of air pollutants and their precursors affect atmospheric dispersion and chemistry processes in general, and air quality in particular? For example, might an increase in heatwaves affect air pollution episodes? Might the frequency and intensity of winter inversions decrease? If so, how will this affect air quality?

Several issues arise here. Unless there are any new non-linearities introduced by enhanced climate change, the effects on policy measures are probably minor. It should simply mean that we might need to do more than we thought (more of the same) if say, climate change leads to more frequent and more intense summer smog episodes. We might actually get more improvement than we thought, for the same emission reductions, in 'winter' episodes due to less frequent and less intense winter stagnation periods.

There are some estimates of climate change impacts on air quality that apply to Canada. On a broad scale, the Intergovernmental Panel on Climate Change (2001) has projected that, based on some scenarios, background levels of ground-level ozone will increase by more than 40 parts per billion over most mid-latitudes of the Northern Hemisphere. On a finer scale, Cheng *et al.* (2005) provides projections for air quality affecting Windsor, Toronto, Ottawa and Montreal, and estimated that for 3 different emissions scenarios the number of low ground-level ozone days would generally decrease and the number of high ground-level ozone days would generally increase. In the worst case scenario of air pollutant emissions increasing by 20 per cent by 2050 and 32 per cent by 2080, the study estimated that the annual total number of poor ozone days (one-hour maximum $O_3 \geq 81$ ppb) could increase by 4-11 days by the 2050s, and by 10-20 days by the 2080s. The number of good days (one-hour maximum $O_3 \leq 50$ ppb) could decrease by 24-40 days by the 2050s, and by 42-52 days by the 2080s."

Health Canada and Environment Canada are developing some new scenarios of climate change impacts on air quality for the 2007 national assessment on climate change and health, but it remains uncertain if these efforts will improve our understanding substantially. It is probably prudent to agree with the Air Quality Expert Group (2005) and the point that different models show quite a wide range of responses, and that there are large uncertainties in the modelling output. In Canada, or Ontario specifically, there is little doubt that air quality will get worse with climate change; however, by exactly how much is less certain, but the conclusion is nonetheless clear --- that we need to do even more on reducing emissions causing air pollution, and ideally do so without adding more greenhouse gas emissions.

A logical extension of this work would be to project health effects. Research in Canada has looked at the synergistic impacts of temperature change and air quality under three climate change emission scenarios (Cheng *et al.*, 2005) and projected under the worst case conditions that mortality due to poorer air quality would increase 15-25 per cent by 2050 and from 20-40 per cent by 2080. If you add to these numbers that heat-related deaths would double and triple by 2050 and 2080 respectively. These are not trivial numbers in terms of human health, but the synergistic and cumulative impacts on managed and unmanaged ecosystems could also be substantial, whether in concert with air quality, acid deposition or air toxics.

Undoubtedly, a key to the assessment of future ozone effects is the issue of whether or not there is a threshold for effect. If there is, then the projected effects could be large; if there isn't, then they will be small and potentially significantly less than those due to PM.

Ozone is also difficult because the behaviour of future trends depends on the metric one is examining. Peak hourly ozone will behave differently from the annual mean (in general in urban areas the former will decrease with decreasing NOx and VOCs but the latter will increase, being dominated by the titration in urban areas), and metrics between these two extremes will differ in their behaviour too. In fact the behaviour is controlled by three factors (i) the local NOx environment and the titration effects (ii) the behaviour of the NOx/VOC smog reactions in future scenarios and (iii) the influence of the global tropospheric background. Because of these (especially (i) and (ii)) future ozone trends will be strongly location specific and this means that one has to do urban scale modelling – not something the ozone modelling community has addressed very much as yet. This conclusion is consistent with conditions experienced in Ontario and Quebec, a region that is subject to considerable transboundary pollution from the U.S. Ohio Valley. In Toronto, for example, during smog episodes driven by ozone, more than 90 percent of the pollution comes from the U.S.; whereas during PM episodes, the percentage is closer to 50 percent. This implies that local actions to reduce emissions, especially during ozone events, will have little impact on ambient conditions, and that for measures to be effective, they either have to be international in scope, or at a much finer spatial resolution. In the latter case, emission reductions may need to be neighbourhood (or even site) specific, in addition to micro-modelling of individual risk exposure..
The other important issue in terms of climate change and ozone is the influence of biogenic emissions which needs fuller assessments in considering control scenarios for future years.

Question 2: What are the links between the sources of emissions responsible for climate change and air quality?
What are the main scientific issues associated with the interactions of GHGs and air pollutants in the atmosphere and their impacts on climate change and air quality?

Question 3: What do future trends in UK air pollutant emissions tell us about the potential impact on climate for the UK and Europe? Given that some air pollutants cause air quality concerns on a regional scale, over what scale will their impact on climate be felt?

The answers to these questions are even more complex than the previous question and relationship. The role of aerosols in off-setting climate change on a local scale is still very much an emerging science, and policy makers risk venturing into the debate of having decision makers ponder polluting more SO_2, NO_X and PM in order to off-set climate change.

More specifically, the challenges, complexities and trade-offs between air quality and climate change can be illustrated by considering three key sectors: energy, transportation and agriculture. Although it ultimately comes down to how society generates, produces and uses energy, policy-makers also need to consider the need to move towards a net zero carbon future, where we need to decouple the global economy from fossil fuels and rely upon non-carbon sources of energy.

Both climate change and air quality policies deal essentially with the same emission sources, so it is clearly sensible to ensure they are considered together by policy makers. Some points to make are as follows, but these are not comprehensive or complete regarding challenges and opportunities.

ENERGY:

* Any policy measure which reduces the use of fossil fuels in existing applications will be co-beneficial for air quality and climate change. Such measures include energy efficiency in buildings and households, which could also have co-benefits in the form of improved indoor air quality.

*Measures to increase the proportion of carbon-free energy generation in the portfolio will be co-beneficial. Sources would include, wind, solar, hydro, tidal, wave, and nuclear, although some of these have their own associated problems and challenges (i.e. nuclear waste issues and public acceptance of wind farms). Community-based systems enhance local adaptive capacity, create and retain jobs in the local community, and potentially reduce a wide range of pollutants contributing to air pollution and climate change, in addition to CFCs, although these outcomes are necessarily clear (see below).

*Potential trade-offs could arise where energy generating sources fit aftertreatment of the flue gases, a practice which usually leads to a small fuel consumption penalty. Historically this penalty has been considered worth paying due to the significant air quality benefits for public health and the wider environment which can accrue. Reductions of air pollutants like sulphur dioxide has resulted in a decrease in aerosol sulphate concentrations which, on the basis of current knowledge, has lead to an increase in radiative forcing. Despite this, it is unlikely that policy measures would be considered to increase sulphur emissions as a means of alleviating radiative forcing.

*Measures to increase the efficiency of fossil fuel use by replacing remote, central energy generation from fossil fuels by local small scale combined heat and power sources in urban areas, running on fossil fuels could lead to climate change/air quality trade offs which should be quantified and assessed.

TRANSPORT

* The classic problem here is the diesel vehicle. This has more efficient use of fossil fuel energy than petrol and hence smaller carbon emissions per kilometre travelled, all other things being equal. However, there are potentially significant public health disbenefits arising from the higher emissions of particulate matter which have arisen to date from diesel vehicles compared with petrol equivalents. Technology is available to reduce significantly these emissions of particulate matter and proposals for emission standards for light duty vehicles which it is thought will require such technologies have been proposed by the European Commission (see Chapter 4). Concern has been expressed that these devices lead to increased fuel consumption, of the order of a few percent, and this has been cited as a reason not to proceed with these controls. It seems likely, at the time of writing, that the EU will go ahead with agreeing such standards so that by implication the U has tacitly accepted that the small fuel penalty is outweighed by the relatively large (potentially of the order of about 90%) reductions in particulate matter. One factor which has not been included in analysing these trade-offs is the benefit to radiative forcing which may arise from the reduction in black carbon emissions from diesel vehicles. The reason this has not been done is the uncertainty in the science in this area, and that of the wider issue of aerosols and climate change as a whole where more research is clearly needed. However, measures such as particulate filters/traps which reduce particulate matter emissions by significant amounts will clearly be effective in reducing these trade-offs.

* As with energy use in fixed sources, any policies which lead to reduced travel and/or fuel use will be a win-win for climate change and air quality. Such measures are usually fiscal and could involve such policies as road user charging, fuel duty measures, tax/duty measures on high-emitting vehicles (although until the primary particulate emissions from diesel vehicles are reduced significantly-as discussed above-there are potential perversities in applying such measures to the current fleet). Measures on aviation are probably of wider interest in the climate change context but reductions in NO_x emissions from aircraft engines in the cruise and take-off engine modes will benefit both local air quality and climate change.

* In the medium to longer term, low carbon vehicles (hybrids, fuel cell vehicles etc) will also be win-wins, providing the primary energy generation is also low or zero carbon.

AGRICULTURE

* The common issues linking climate change and agriculture are mainly related to methane emissions and its impact on tropospheric ozone levels, and ammonia emissions which can affect ecosystems directly in the vicinity of sources, and at longer range through the formation of secondary particles which can affect health and can be deposited on ecosystems where they contribute to acidification and eutrophication problems.

* The commonality of air quality and climate change issues and tropospheric ozone is clear and there will clearly be co-benefits arising from any measures to reduce methane emissions from agricultural sources world wide.

* The link between ammonia, air quality and climate change is less obvious. The problem arises through the so-called 'pollution swapping' concept and the management of nutrient nitrogen in the agricultural context. Nitrogen releases in this sector arise from fertiliser use and the excretion of nitrogen by animals. Depending on the specific local practices, residence times of manure and slurry in containers and soils etc, this nitrogen can potentially enter the environment as nitrate in streams and rivers where it can cause water quality problems, or it can be released to the atmosphere as ammonia and contribute to the problems outlined above, or it can be released as nitrous oxide, a powerful greenhouse gas. Abatement methods and policies to reduce the effects of nitrogen on the environment need to recognise these potential problems and seek to find optimal solutions. This is an area of developing science.

5.3 Future Research Requirements

What are the current gaps in our knowledge? Where should future research focus to provide appropriate scientific information to inform decisions about the comparative benefits of air quality and climate change mitigation measures? Are the currently available scientific tools sufficient to answer these gaps in our knowledge, and if not, what further developments are required?

Perhaps these questions are best answered after crticial discussion at the conference, but at this point we can begin to at least map out the challenge. Certainly technology will play a big role in determining how successful we are in reducing emissions of GHGs and other air pollutants. In Canada, there are large dollars being invested by government and industry in sustainable technologies, and a review of the Canadian situation illustrates which sectors are attracting the most attention from an investment perspective. According to Sustainable Development Technology Canada (2005) the two sectors receiving the most funding are energy exploration and production (24%), and energy utilization (25%), followed by power generation (16%), transportation (12%), forestry and wood products (9%), waste management (9%), and agriculture (5%).

The three examples discussed above illustrate rather than comprehensively inform decision makers about the challenges of making connections between air quality and climate change. These include energy – specifically the feasibility of less polluting alternatives such as "Clean Coal", green renewables (e.g. river run hydro, wind, solar, biomass), and other less polluting fossil fuels (distributed energy systems, co-generation natural gas for example). The National Roundtable on the Economy and Environment recently came out with their own climate change and energy strategy for 2050, and this included the adoption of clean coal technologies in western Canada (in Alberta and Saskatchewan where it may be geologically feasible to sequester CO_2 underground, and at the same time make it easier to extract oil and natural gas from the tar sands), but this would not be suitable to Ontario where the geological conditions do not support underground storage. There may be some potential in the U.S. where coal plants are more likely to be located closer to coal mines, but under the Clean Air Interstate Rule there seems to be almost exclusive commitment to improving air

quality rather than dealing with climate change (undoubtedly a reflection of the current Bush administration's attitude towards climate change), where significant reductions in NOx, SO_2, and even mercury are possible through end of pipe technology. Of course these technologies do nothing about CO_2 and in some cases can even add GHG to the atmosphere.

This point is well known in the non-ferrous smelting sector, and INCO and Falconbridge in particular. In the case of INCO's superstack in Copper Cliff (Sudbury) Ontario, their sulphur extraction process results in a lowering of temperature in the plume that otherwise contributes to acid rain hundreds and thousands of kilometers away. The lower temperature, unless addressed, would result in the plume falling almost immediately, thereby placing INCO in a non-compliance position with respect to local air quality standards. As a result, they have to burn propane in order to heat up the superstack sufficiently for the SO_2/NOx laden plume to rise sufficiently high enough to be dispersed more broadly. Ironically, the superstack represents outmoded 1970s technological solution to reduce pollution by dilution, and address years of non-compliance with local air standards. However, the addition of burning propane clearly puts the companies operating the smelters in conflict with any regulatory requirements or expectations to reduce GHG emissions. Furthermore, in the recent Pollution Prevention plan posted on the Canada Gazette for the regulation of SO_2 emissions from non-ferrous smelters, community-scale air standards and monitoring were included to protect human health.

Non-carbon alternatives pose challenges in terms of pricing and intermittency, and also open the door to include the nuclear option (which has resurfaced in Ontario, and in parts of Europe, despite wishes to retire and decommission nuclear plants). In many countries the best places to develop renewables, including large scale hydro, tend to be located far away from where the demand is, which poses additional challenges in terms of transmission and distribution. Nonetheless, there is great untapped potential for renewables in many countries, including Canada. In northern Ontario, for example, untapped river-run hydro and wind power in the James Bay and Hudson's Bay lowlands could easily supplant the power currently generated by nuclear plants.

It is also important to recognize that energy efficiency and energy conservation may in fact be two of our most important measures to lower emissions (by reducing the problem at the source, simply reducing our use of energy). Canada's use of coal-fired electricity and emissions causing air pollution and climate change would be much higher today, if we hadn't been so successful at improving energy intensity and efficiency. That being said, Ontario is about 50 per cent less efficient than neighbouring New York State suggesting that there is much room for improvement.

At the residential scale, energy efficiency/conservation options such as improved insulation, heat and air exchange systems, green roofs, etc. can lead to reduced energy and electricity use, and also provide co-benefits for urban biodiversity, and improved indoor air quality.

In terms of transportation, improved fuel efficiency standards are essential to reduce GHG emissions, but do not necessarily equate to reduced air pollutant emissions (generally they do, but sometimes they don't). Alternative fuels is also an option, as many governments are now moving towards the expansion and promotion of ethanol and biofuels, but if you apply a life-cycle assessment then the overall benefits to the environment and health in terms of GHGs and air pollutants are not that large, if at all, depending upon the pollutant that you are considering.

Using less fuel or moving to less emitting vehicles on a km-passenger basis is another option, such as modal shifts from single occupant vehicles to public transit, car pooling, telecommuting, or active commuting. The latter has huge implications for children and youth, in terms of combating obesity and diabetes, but runs into the problem of promoting physical activity during smog episodes. Land use and transportation planning is also essential – specifically the problem of sprawl, as North American cities know only too well (to a lesser degree in Europe and South East Asia). In Toronto, BAU projections are for an additional 3 million people by 2030, an equal number of passenger vehicles, and a 30-40 per cent increase in GHG emissions from transportation sources. Building more efficient vehicles, installing better emission control technologies, and using alternative fuels are all good measures, but we also need to go beyond and consider not using cars period. Similar challenges exist for the movement of commercial goods and freight, involving air, rail, shipping, and intercity and local trucking. Incorporating intermodal use into a sustainable transportation strategy remains the unsolvable problem, as does addressing "just-in-time" delivery systems (the equivalent to sprawl as a huge structural problem).

In the end, it is important to recognize the need to look at these problems and challenges more closely, accept that these challenges are significant, and that there is no silver bullet that is going to solve the problem of both air quality and climate change. A wide suite of measures will be required, and we need to move quickly and effectively.

Acknowledgements

The authors would like to acknowledge the contributions of the following individuals to the preparation of this chapter:

- Andre Zuber and Terry Keating for providing material on hemispheric air pollution
- Juergen Schneider for providing material on the Exposure Reduction Target approach of the European Commission
- Geoff Granville for contributing to the development of the scope and direction of the chapter

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