Strategies for Clean Air and Health

Proceedings of

AIRNET Annual Conference/ NERAM International Colloquium
November 5 – 7, 2003
Rome, Italy

July 2005

Guest Editors

Lorraine Craig
Network for Environmental Risk Assessment and Management

Daniel Krewski
Professor and Director, R. Samuel McLaughlin Centre for Population
Health Risk Assessment, University of Ottawa

John Shortreed
Executive Director, Network for Environmental Risk Assessment and Management

Jonathan Samet
Chair, Department of Epidemiology, School of Public Health
Johns Hopkins University

Institute for Risk Research
University of Waterloo
Waterloo, Ontario, Canada N2L 3G1
Tel: (519) 888-4567, ext. 3355
Fax: (519) 725-4834

Guest Editors: Lorraine Craig, Daniel Krewski, John Shortreed, Jonathan Samet

Managing Editor: Addy Mitchell

Printed and bound at Graphic Services, University of Waterloo

ISBN 0-9684982-4-8

Table of Contents

PART III - Mechanism of Effect

PART IV - Policy Tools and Approaches

PART V - Science-Policy Issues

PREFACE

L. Craig[1] , D. Krewski [2], J. Samet [3], J. Shortreed[1]

The task of identifying sound air quality management strategies to improve human health and the environment involves a number of scientific, technical, and economic considerations embedded within a social and political context. While lack of full scientific certainty limits our ability to identify optimal policy interventions, research findings on ambient air quality, source apportionment, and health effects can help to guide the development of local, regional and continent wide air pollution management strategies. Policy analysis tools incorporating air quality modeling further support decision-makers in the development and evaluation of air quality management policies by estimating impacts of policies on ambient air quality, health, the environment, and the economy.

"Strategies for Clean Air and Health," the fourth in a series of five NERAM Colloquia, was organized jointly with the AIRNET European Network on Air Pollution and Health and the Rome E Health Authority to identify directions for air quality policy development and research priorities to improve population health. Two hundred air quality scientists, policymakers, and representatives from industry and non-governmental organizations from 22 countries convened in Rome, Italy on November 5-7, 2003 to exchange perspectives on the interface between science and policy relating to air pollution health effects, air quality modeling, clean air technology, and risk management policy tools. The conference was sponsored by US EPA, Health Canada, Ontario Ministry of Environment, Shell International, the Canadian Petroleum Products Institute, the European Commission, and the Rome E Health Authority.

This volume includes twenty-one peer reviewed conference papers from North America and Europe addressing the following critical topics in air quality risk management: i) ambient air quality, ii) health effects, iii) mechanism of effect, iv) policy tools and approaches and v) science-policy issues. In addition, a Conference Statement based on the discussions that took place at the meeting provides guidance from the perspective of an international group of scientists, regulators, industries and interest groups on a path forward to improve the interface between science and clean air policy strategies to protect population health. The Statement emphasizes that air pollution in an important local, national, and global population health concern that requires further attention in specific airsheds throughout the world, despite uncertainty in our knowledge of the most harmful components. Policies strategies most likely to be successful are those that are aimed at achieving broad population health co-benefits, including those that integrate clean air goals within urban planning, health promotion, and climate change/energy demand management initiatives. Emission reduction strategies should focus on the toxicologically most important sources of ambient air pollution and consider predicted effectiveness, benefits and costs, as well as implementation time and feasibility. Public and stakeholder outreach is important in building support for policy changes, including the public health community and those at high risk of adverse health impacts. Continued communication among scientists, policy makers, stakeholders and the public is critical for improving the science-policy interface. The Statement identifies the need for innovative approaches to further support policy development including cross-disciplinary health effects research and studies to assess the effectiveness of control strategies. Finally the Statement recommends an international forum for evaluation of scientific evidence on health effects and a methodology for integration of evidence for air quality risk management policy development.

Part I provides a Canadian and European perspective on ambient concentrations of air pollutants in major cities and characterizes the role of local transportation and industrial emission sources.

Brook et al. and Jerrett et al. present a Canadian perspective on sources of $PM_{2.5}$ impacting southern Ontario and its major city of Toronto. Brook et al. use monitoring data to estimate that 30-45% of $PM_{2.5}$ is locally generated, which implies that 55-70% are transported into the area. Using two different receptor-based analyses it was determined that motor vehicle related emissions (exhaust and road dust), most likely of local origin, were responsible for about 20% of the $PM_{2.5}$. Secondary $PM_{2.5}$ from coal-fired power plan emissions was a significant contributor and also played a

[1] Institute for Risk Research, University of Waterloo, Waterloo, ON, Canada.
[2] McLaughlin Centre for Population Health Risk Assessment, University of Ottawa, Ottawa, ON, Canada.
[3] Department of Epidemiology, Johns Hopkins Bloomberg School of Public Health, Baltimore, MD, USA.

role in enhancing production of secondary organic carbon mass on fine particles. The paper cautions that while the results of receptor-based analyses can be used to inform potential strategies for reducing ambient $PM_{2.5}$, it is important to also consider the relative toxicity of the mix of emissions from the various sources in order to target reduction strategies to those which will provide the greatest public health benefit.

Jerrett et al. report on the first North American study to characterize intra-urban traffic pollution with a land use regression model. Land use regression seeks to predict pollution pollution concentrations at a given site based on surrounding land use and traffic characteristics. This work is important in addressing some of the uncertainties in exposure assessment methodology in order to guide effective traffic pollution control strategies. The results indicate that predictive maps from the land use regression method appear to capture small area variations in traffic pollution (NO_2). These variations are likely important to the exposure experience of the population and may detect health effects that would have gone unnoticed with other exposure estimates. The paper suggests that further empirical work in North American be conducted to assess the adequacy of different predictive variables.

Marconi et al. discuss the results of two years of fine and ultrafine particle measurements in a traffic-related site, and an urban background site in Rome Italy. The study was conducted to establish validated and consistent data sets of particle number concentrations in Rome to be used in epidemiological analyses of cardiovascular health effects associated with exposure to ultrafine particles. Peak particles number count (PNC) events were found during winter and during the morning and evening rush hours. CO, NO and NOx were all highly correlated with the particle number concentrations. Consistent with other studies, daily $PM_{2.5}$ and PM_{10} levels were found to be poorly correlated with the daily particle number count, suggesting that independent measurement systems are required if the relationships between health outcomes and ultrafine number concentrations are to be assessed. The results emphasize the importance of primary particles originating from road traffic and the influence of meteorological conditions on particle number concentrations.

Fierens et al. assess the impact of pollutant emissions from two iron and steel plants and two municipal solid waste incinerators (MSWIs) on the exposure of residents of Wallonia, Belgium. The concentrations of dioxins, PCBs, cadmium, mercury and lead in blood and urine of subjects living in the vicinity of two sinter plants and two MSWIs were compared with levels found in referent subjects recruited in a rural area with no industrial source of pollution. After adjusting for covariates, the results show that dioxins and coplanar PCBs emitted by MSWIs located in the rural area can accumulate in nearby residents, however the accumulation requires a regular consumption of local animal products contaminated by relatively high emissions of dioxins. The authors note that it is unlikely that the increased dioxin body burden occurs around MSWIs complying with emission standards currently in force in most countries. Emissions from the sinter plants were not associated with an increased dioxin body burden of residents. This was likely due to the lower dioxin emissions and lower local animal fat consumption of residents around these facilities.

Part II provides an overview of health effects associated with exposure to ambient air pollution based on North American and European studies and explores uncertainties in their estimation using Canadian data. The impact of urban air pollution on less severe morbidity outcomes among children and the elderly in Italy is described. Evidence of early airways inflammation among children exposed to high ozone levels during smog episodes is presented.

Samet and Krewski provide a review of the current state of science on the human health impacts of air pollution. The review is based on the framework and findings of the U.S. National Research Committee (NRC) on Research Priorities for Airborne Particulate Matter. The paper addresses key questions underlying air quality risk management policy decisions.

Krewski and Rainham summarize more than 35 research posters presented at the conference addressing exposure, toxicological and epidemiological studies of air pollution. The studies provide important new findings from more than 41 cities and metropolitan areas across Europe and further support previous evidence that both short and long-term exposures to particulate air pollution have adverse population health impacts, including effects on children. Cellular studies suggest that air pollution can cause mutagenic and oxidative effects, raising concerns about carcinogenicity and cellular regeneration. Studies of biomarkers provide further evidence of air pollution effects at the cellular level. Improved exposure assessment models support the close association between traffic patterns and

air pollution concentrations. Collectively, the findings present opportunities and challenges for the development of policies to improve air quality and human health.

Sahsuvaroglu and Jerrett assess the sensitivity of air pollution health effects estimates in Hamilton, Canada to a wide range of possible uncertainties including baseline pollution levels, single versus multipollutant effects, local or pooled estimates, and chronic effects. Given the wide ranging estimates of mortality and hospital admissions, the authors suggest the need to reach consensus on formats for reporting study results and burden of illness and mortality estimation conventions. To improve the completeness of the health effects estimates, the paper identifies the need for chronic health effects studies, multipollutant studies and studies on health outcomes that are likely to have large population health impacts.

Bedeschi et al. and Vigotti et al. explore the association between exposure to urban air pollution and daily emergency room visits for respiratory disorders among children and elderly in Italy. Using single pollutant models, significant associations were found between road traffic pollutants (PM_{10} or TSP and NO_2) and emergency room visits for respiratory disorders among children residing in a small city of Northern Italy (Reggio-Emilia). These associations were independent of the effect of temperature, humidity and pollen trend. Using single pollutant analyses, Vigotti et al. found statistical associations between PM_{10} concentrations and emergency room visits for respiratory complaints among children and the elderly in Pisa, Italy. Evidence of these less severe health outcomes suggests the need for further action to prevent deterioration of respiratory health among susceptible populations.

Nickmilder et al. assess the effect of ambient ozone on lung function and airways inflammation among healthy children attending summer camps in rural areas in the South of Belgium. While lung function tests did not show any consistent pattern of decrease at ozone concentrations ranging from 48 to 221 $\mu g/m^3$ (1 hour maximal concentration), a highly significant increase in exhaled nitric oxide levels was found in all subjects from an ambient 1 hour ozone level of 167 $\mu g/m^3$. The authors note that the increased exhaled nitric oxide is a marker for inflammation and oxidative stress in the lung that passes undetected with spirometric tests. These impacts were observed at levels that are slightly below US and EU air quality standards, suggesting the need for greater use of this non-invasive test in the assessment of the health risks of ozone and the subsequent derivation of health-based air quality standards.

Part III includes two papers investigating biomarker methods to assess exposure levels and mechanisms of effect. de Burbure et al. assess the protective role of selenium dietary supplementation with respect to nitrogen dioxide lung toxicity in an inhalation study on Wistar rats. The protective role of selenium status was evident for both long term and acute exposures, however its negative impact on Clara cell protein (a natural anti-inflammatory and immunosuppressor) indicates that caution should be used prior to advocating selenium supplementation.

Beskid et al. studied the impact of air pollution containing higher concentrations of carcinogenic polycyclic aromatic hydrocarbons in three European cities using the fluorescence in situ hybridization (FISH) technique as a biomarker indicator of chromosomal changes related to cancer. The findings suggest that police officers in Prague (Czech Republic), Kosice (Slovakia) and Sofia (Bulgaria) as well as bus drivers in Sofia represent a group with increased genotoxic risk. The differences in genetic effects observed in the occupationally exposed and control populations reflect the longer exposure to polluted air experienced by the former population. Further work is needed to analyze factors affecting the genomic frequency of translocations and to establish how to translate the findings for risk assessment.

Part IV describes tools and approaches to guide the development of air quality policy. Case studies of policy development strategies in Europe point to the challenges that policy makers face in devising sound risk reduction strategies. Reid et al. present an overview of atmospheric models and their application to the development of air quality policy. The paper reviews modeling case studies to identify issues associated with model validity and accuracy and provides guidance on the requirements for credible modeling. Rabl et al. offer a brief review of the need for cost-benefit analysis and the available policy instruments for assessing externality costs associated with air pollution. The paper describes the ExternE (External Costs of Energy) project series of the European Commission (EC) and the Life Quality Index as examples of approaches to assess the costs and benefits of pollution control strategies as well as the inherent uncertainties in their estimation. Davidson describes the Environmental Benefits Mapping and Analysis Program (BenMAP) developed by the U.S. Environmental Protection Agency (U.S. EPA) to

assist in the benefit-cost analysis of air pollution control policies. Two $PM_{2.5}$ pollution control policy scenarios are reviewed to demonstrate BenMAP's capabilities. Brody et al. describe a project aimed at building capacity to set environmental priorities in the Ukraine through the use of comparative risk assessment and economic analysis. The project is a partnership between the U.S. EPA and Ukraine's Ministry of Environmental Protection. The paper describes the Russian experience with risk assessment and progress achieved in applying risk assessment to setting environmental priorities in Ukraine. Periera et al. report on trends in annual sulphur dioxide (SO_2) concentrations in the Oporto Metropolitan Area of Portugal before and after the implementation of European Union legislation in 2000 to reduce the sulphur content in fuels (Auto-Oil Directives). Annual SO_2 emissions and concentrations had decreased since implementation of the policies as well as a significant decrease in the number of exceedances of SO_2 recommended limit values. Improvement in fuel quality, the application of emission control programmes and recent technological innovations are believed to be responsible for these improvements. Glorennec and Monroux estimate the health impacts of acute and chronic exposure to PM_{10} in Caen, a city in northwestern France. The authors estimate that a 10% daily decrease in pollution would reduce the number of expected deaths from short-term exposure by 19%, while achieving compliance with European Union regulations (daily mean in 2010: 50 $\mu g/m^3$) would reduce them by less than 3%. The paper concludes that pollution peaks do not contribute significantly to long-term exposure, the impact of which is greater than acute exposure and suggests that local risk management policy aimed exclusively at avoiding exposures exceeding regulatory levels will have only a marginal impact on public health.

In Part V, issues arising during the interface between science and the development of robust policy strategies are discussed from a European stakeholder perspective. Maas describes uncertainties in the source-effect chain and identifies policy strategies for particulate matter abatement that are consistent with a certain set of assumptions, as well as the risks that are associated with such strategies. The paper offers a systematic approach to developing sound PM control policy that considers the appropriate balance between environmental and economic precaution. Wallis offers a critical appraisal of the air quality science-policy interface in the UK. The paper identifies shortcomings in the systems for reporting on progress in reducing air pollution levels and advising the public on precautionary actions. Weaknesses in standard setting on PM and the lack of attention in the policy setting process to ultrafine particles and health effects on the unborn are discussed. The paper emphasizes the vital need for ENGO participation within a stakeholder process that incorporates effective communication and accountability in public information and policy setting. van Bree discusses critical issues underlying the interface between air quality science, stakeholder participation and policy development within the context of the European AIRNET Network multi-stakeholder project. The paper argues that the process and mechanisms by which the science-policy interface operates are as important as the content of the air pollution and health science and offers lessons learned in establishing a multi-stakeholder air quality network to bridge the gap between the policy, stakeholder, public and scientific communities.

This unique collection of international perspectives on air quality science and risk management policy issues serves as a comprehensive resource to inform the development of future air quality policy strategies. A special thank you is extended to Addy Mitchell for formatting the papers and co-ordinating the preparation of this volume, to host Francesco Forestiere of the Rome E Health Authority, to Addy Mitchell and Laurie MacDonald of NERAM, Suzanne Therien of the McLaughlin Centre for Population Health Risk Assessment, Eef van Otterloo and Marjan Tewis of the AIRNET management team and Ingrid Dahmen of the Institute for Risk Assessment Science (IRAS) for their roles in the planning and execution of a successful meeting in Rome.

STRATEGIES FOR CLEAN AIR AND HEALTH: CONFERENCE STATEMENT

L. Craig[1], D. Krewski[2], J. Samet [3], J. Shortreed[1], L. van Bree[4]

ABSTRACT

An international Colloquium "Strategies for Clean Air and Health" was organized by the Network of Environmental Risk Assessment and Management (NERAM) and the AIRNET European Network on Air Pollution and Health to identify directions for air quality policy development and research priorities to improve public health. A Conference Statement was prepared to provide guidance from the perspective of an international group of scientists, regulators, industries and interest groups on a path forward to improve the interface between science and clean air policy strategies to protect public health. The Statement represents the main findings of two breakout group discussion sessions, supported by perspectives of keynote speakers from North America and Europe on science-policy integration and views of the delegates expressed in plenary discussions. NERAM undertook a carefully considered process to try to ensure that the Statement would accurately reflect the conference discussions, including documentation of supporting comments from the proceedings and inviting delegates' comments on two draft versions of the Statement.[5]

[1] Institute for Risk Research, University of Waterloo, Waterloo, ON N2L 3G1 Canada.
[2] McLaughlin Centre for Population Health Risk Assessment, University of Ottawa, Ottawa, Ontario, Canada.
[3] Department of Epidemiology, Johns Hopkins Bloomberg School of Public Health, Baltimore, Maryland, USA.
[4] Netherlands Environmental Assessment Agency (MNP), Bilthoven, The Netherlands.
[5] The Statement is not a consensus document and may not reflect the views of all conference delegates. All comments received and responses to comments are available at www.irr-neram.ca. This documentation provides insight into stakeholder perspectives on issues underlying the development of strategies for clean air and health.

STATEMENT SUMMARY

1. **Air Pollution is an Important Local, National and Global Public Health Concern**
2. **Implement Further Clean Air Policies Based on Current Scientific Knowledge**
3. **Initiate Innovative Research Approaches to Support Air Quality Policy Development**
4. **Improve Communication among Scientists, Policy Makers, Stakeholders and the Public**
5. **Use Exposure and Health Impact Assessments to Assess Benefits of Implemented and Future Regulations and to Develop Interventions**
6. **Initiate International Forum for Evaluation of Principles for Action and Integration of Air Quality Evidence for Policy**

INTRODUCTION

An international Colloquium ***"Strategies for Clean Air and Health"*** was organized by the Network of Environmental Risk Assessment and Management (NERAM) and the AIRNET European Network on Air Pollution and Health to identify directions for air quality policy development and research priorities to improve public health. Two hundred air quality scientists, policymakers, industry representatives and non-governmental organizations from 22 countries convened in Rome, Italy on November 5-7, 2003 to exchange perspectives on the interface between policy and the science on air pollution health effects, air quality modeling, clean air technology, and policy tools. The conference was sponsored by US EPA, Health Canada, Ontario Ministry of Environment, Shell International, the Canadian Petroleum Products Institute, the European Commission and the Rome E Health Authority.

The Statement will provide the focus for future NERAM Colloquia and related initiatives to engage broader participation of government policymakers, corporate decision makers, non-governmental organizations, and policy-focused scientists at the international level in identifying and implementing best practice in air quality management. After having brought together scientists, stakeholders and policy makers at its first Annual AIRNET Conference (London 2002), the Statement of this 2003 second conference will provide a focus for AIRNET to better function as a platform and forum for the science-policy-stakeholder interplay. This new focus on end-user needs will also set the stage for the final Annual AIRNET 2004 Conference (Prague) addressing science and policy communication issues on air pollution and health.

1. Air Pollution is an Important Local, National and Global Public Health Concern

Epidemiologic studies conducted over the past decade to examine the link between community ambient PM concentrations and health continue to show that current levels of air pollution are a significant local, (inter)national and even global public health concern. 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.[6] There is consistent and convincing evidence to show that both short term and long term exposures are associated with premature mortality and a range of cardiovascular and respiratory illnesses. Air pollution is a complex mixture of particulate and gaseous co-pollutants originating from both local and distant sources. Much of the recent research findings and policy development efforts address the health effects of ambient particulate matter (PM) and while delegates identified the important role of gaseous co-pollutants and air toxics, particulate matter was the predominant focus of discussion. PM is a very complex mixture and its chemical and physical composition varies over time and seems dependent upon meteorological and long-range transport conditions and specific source

[6] Cohen, A.J., Anderson, H.R., Ostro, B., Pandey, K.D., Krzyzanowski, M., Kuenzli, N., Gutschmidt, K., Pope, C.A., Romieu, I., Samet, J.M., Smith, K. 2003. Mortality Impacts of Urban Air Pollution. In: Ezzati M, Lopez AD, Rodgers A, Murray CJL (Eds). Comparative Quantification of Health Risks: Global and Regional Burden of Disease Attributable to Selected Major Risk Factors. Geneva: World Health Organization. In Press.

contributions. However, there is now a substantial body of evidence to show that PM_{10} and $PM_{2.5}$ are associated with adverse health effects in airways and lungs and the cardiovascular system. These particles, to which $PM_{2.5}$ forms a major constituent, originate i) directly from combustion and industrial processes, such as from large point sources like coal-fired power plants and steel mills, and from area and mobile sources such as vehicles and ii) indirectly through the complex atmospheric formation of secondary particles from precursor gases. These source emissions should be a priority for cost-effective risk reduction in affected airsheds. For the gaseous co-pollutants (e.g. CO, NO_x, SO_2, O_3) health effects of O_3, and perhaps also CO, are considered important, whereas others may serve as an indicator (surrogate) for the mixture without being a causal health effect agent themselves under the current (low) exposure levels. The health effects associated with exposure to ultrafine (UF) particles (<0.1 μm) continue to be a high research priority because bio-medical toxicology has discovered potentially important adverse reaction pathways, although specific UF epidemiology studies have been rather limited. Ultrafine particles from combustion processes, particularly vehicle engines, reach levels in urban streets, homes and workplaces and may have adverse effects for public health.

2. Implement Further Clean Air Policies Based on Current Scientific Knowledge

Although there is some scientific uncertainty about the specific components of air pollution that are most responsible for health effects and the exact mechanisms of these effects, there is sufficient confidence in the information available for policymakers in affected airsheds to take further immediate actions. In North America and Europe, a wide range of clean air strategies have already been implemented (i.e. Tier 2 motor vehicle emission standards[7], ambient air quality standards and limit values, emission limits for industrial facilities, emission ceilings for countries), while other policy strategies and emerging new technologies for emission reduction will be implemented over the long term (fuel cells, gasification of coal, electricity generation by gas turbine (already has widespread applications in California and elsewhere), etc.). While these strategies are designed and expected to improve air quality and public health, additional policies are still needed in specific airsheds throughout the world to lower air pollution to healthy or acceptable levels. Air quality monitoring has demonstrated that historical air pollution abatement programs have been effective in reducing ambient levels of air pollution. Studies in southern California, Hong Kong, Ireland, Erfurt, Utah Valley and Atlanta have demonstrated health benefits from emission controls and other interventions. Further monitoring and research is needed to ensure that existing and future policy approaches indeed lower population exposure and improve public health.

The presentations and discussions at the Conference identified the following strategies and guidelines for the development of clean air policies:

- Focus on policies that are likely to achieve broad population health co-benefits. For example, integrate clean air objectives within urban planning and community design (green spaces, public transport, traffic demand management), no-regrets climate change policies, energy conservation and energy efficiency programs, and health promotion planning, such as in the areas of obesity, diabetes and substance abuse (e.g. tobacco).
- Emission reduction strategies should target all relevant emission sources which contribute to pollution levels in an affected airshed(s). With respect to ambient PM there is a need to focus on those fractions and sources that are suggested to be toxicologically most important (if sound data exist).
- Reduce pollutants that are likely to result in multiple benefits for air quality, for example, precursors that form both fine PM and ozone such as NOx.
- There are potentially cost-effective control measures for reduction of emissions from small scale combustion sources including domestic heating.
- Adopt a risk-based approach to quantitative impact assessment and policy development considering predicted effectiveness and its uncertainties, estimated benefits and costs, and implementation time and feasibility.
- Engage the public and other stakeholders early in the process to help design, focus and build support for policy changes that directly affect them (e.g. urban transportation solutions, energy conservation and sustainable development).

7 Emission standards set by USEPA in 1999 for all cars, light trucks, and larger passenger vehicles, including sport utility vehicles (SUVs) and passenger vans to reduce NOx and non-methane organic gases. Full compliance should be achieved by 2009 with phase in based on vehicle class. Also establishes limits on sulfur concentrations in gasoline.

- Improve linkages with the medical and patient communities to promote their roles in providing an early warning on adverse health effects of air pollution, in credible communication of information, and advocates for solutions.

3. Initiate Innovative Research Approaches to Support Air Quality Policy Development

Continued support for research to improve the scientific basis for the development of air quality policies and strategies is important; however this research should be targeted at areas that will yield information useful in improving public health and contribute to a sustainable living environment. It is likely that the scope of future research needs to be broadened beyond narrow disciplines in order to provide evidence required to support specific policy decisions. There is a need and an opportunity to initiate innovative approaches to health effects research including international cross-disciplinary research to integrate epidemiology, toxicology, and clinical studies; risk-based approaches and health impact assessment methodologies; collaboration among scientists with opposing views; and testing of plausible untested hypotheses. For example, improving access to existing health datasets while assuring patient confidentiality will assist scientists in providing timely research results to support policy recommendations such as standard setting or determining the effectiveness of solutions.

Priority areas for research to guide policy and to demonstrate the exposure and health-effectiveness of control regulations include further studies to better understand the causal agent(s) involved in the air pollution mixture that are responsible for the associations with adverse health outcomes, and to characterize the health effects associated with various air pollution sources. Health effects studies should continue to focus on identifying susceptible subgroups and understanding social and other determinants that may be associated with increased risk.

4. Improve Communication among Scientists, Policy Makers, Stakeholders and the Public

The key to an effective science-policy interface is through interactive dialogue among the scientific community, policy-makers, stakeholders, and the public. Informed public opinion can bring about rapid policy changes. The scientific community has a central role to play in engaging policy-makers and NGOs to ensure that science is understood and presented in an appropriate manner to the public. It is necessary to accurately communicate and openly debate the health effects of air pollution to raise public awareness of the relative importance of the health risks and to create the momentum and support for appropriate policy changes. It is also necessary to include a better participatory approach for policymakers, stakeholders, and the public to effectively communicate end-user needs to the scientific community.

Policymakers need information on the current state of science on air pollution health risks. They also need information on the degree and significance of scientific uncertainty in drawing conclusions in specific areas. More dialogue between policymakers and scientists is necessary, particularly to identify information needs for policy decisions and to determine the degree of certainty required in the science in order to take action. Such dialogue should be used to assist in targeting research expenditures towards critical information needed for policy development. The pace and extent of actions associated with reducing ambient air pollution levels vs other public health interventions also needs to be identified; to address this will require additional technical and process skills. The scientific community is challenged to play a larger role in communicating with policymakers the importance of their findings and implications for policy options. The following approaches to improving the communication of science for policymakers are suggested:

- Establish guidelines for publishing epidemiological studies, including a consistent format for reporting risks, uncertainties and a requirement to identify key policy implications of the results. The standard reporting requirements of the Journal of Epidemiology and Community Health, the CONSORT process[8], and the

8 The CONSORT statement facilitates critical appraisal and interpretation of randomized, controlled trials (RCT) by providing guidance to authors about how to improve the reporting of their trials. The statement consists of a checklist and flow diagram that authors can use for reporting an RCT. Many leading medical journals and major international editorial groups have adopted the CONSORT statement.

"AIRNET Alert" non-specialist summary approach are examples to follow. This would provide information in a format would facilitate systematic review of the literature in support of policy decisions.

- Establish communication frameworks between scientists, policy makers, stakeholders, and interested parties to create a widely accepted basis for public health policy to improve air quality and to communicate and understand each other's needs.
- Establish a common terminology through which scientists communicate messages with policymakers to summarize what is known based on the balance of evidence and to describe in qualitative terms the associated levels of certainty.
- Establish a common language around the concept of "associated effects" and the interpretation of "causality".

5. Use Exposure and Health Impact Assessments to Assess Benefits of Implemented and Future Regulations and to Develop Interventions

Quantitative health impact assessments and assessments of possible exposure and health benefits from air pollution abatement actions are challenged with large uncertainties regarding health effects, concentration-response relationships, and identification of causal pollutants which may become a suitable target for control. Nevertheless, techniques for assessing the possible effectiveness of complex emission reduction scenarios such as Cost Benefit Analysis or Cost Effectiveness Analysis are useful in identifying pollutant reduction strategies with multiple benefits on human health and the environment. The use of these *ex ante* policy analysis tools and broader Health Impact Assessment methodology should assist in the fair and reasonable treatment of risk factors, including exposure, measurement of PM and gaseous pollutants, special susceptibility of population sub groups, and degree of certainty. However, predictions about future benefits of air pollution abatement strategies should be treated with caution and should always include an analysis of uncertainties.

6. Initiate International Forum for Evaluation of Principles for Action and Integration of Air Quality Evidence for Policy

There is a need for an international independent, systematic and regular evaluation of the scientific evidence on air quality health effects for purposes of assisting policymakers. Such tasks have been carried out previously by the UK Committee on Medical Effects of Air Pollutants (COMEAP) for the United Kingdom on a national level, by WHO Euro e.g. for the European region of WHO and the European Commission, and the WHO International Agency for Research on Cancer (IARC). A well-defined methodology for assessing the evidence systematically (e.g., weighting studies according to research design criteria) has been applied in such exercises to pool and integrate the results of the international literature on toxicology, epidemiology, socio-economic analysis and other policy analysis tools. An international review of principles for air quality policy development (socio-economic analysis, precautionary principle, health effects, ethical considerations etc.) should be undertaken to exchange perspectives on how these principles are implemented under various circumstances (e.g. different legal systems) and to identify strengths and weakness associated with each of the approaches.

PART I - AMBIENT AIR QUALITY

ASSESSING SOURCES OF $PM_{2.5}$ IN CITIES INFLUENCED BY REGIONAL TRANSPORT

Jeffrey R. Brook,[1] Richard L. Poirot,[2] Tom F. Dann,[3] Patrick K.H. Lee, Carrie D. Lillyman, Thera Ip[1]

ABSTRACT

The human health effects of fine particulate matter ($PM_{2.5}$) have provided impetus for the establishment of new air quality standards or guidelines in many countries. This has led to the need for information on the main sources responsible for $PM_{2.5}$. In urban locations being impacted by regional-scale transport, source-receptor relationships for $PM_{2.5}$ are complex and require the application of multiple receptor-based analysis methods to gain a better understanding. This approach is being followed to study the sources of $PM_{2.5}$ impacting southern Ontario, Canada, and its major city of Toronto. Existing monitoring data in the region around Toronto and within Toronto itself is utilized to estimate that 30-45% of the $PM_{2.5}$ is from local sources, which implies that 55-70% are transported into the area. In addition, there are locations in the city that can be shown to experience a greater impact from local sources such as motor vehicle traffic. Detailed $PM_{2.5}$ chemical characterization data were collected in Toronto in order to apply two different multivariate receptor models to determine the main sources of the $PM_{2.5}$. Both approaches produced similar results indicating that motor-vehicle-related emissions, most likely of local origin, are directly responsible for about 20% of the $PM_{2.5}$. Gasoline engine vehicles were found to be a greater overall contributor (13%) compared to diesel vehicles (8%). Secondary $PM_{2.5}$ from coal-fired power plants continues to be a significant contributor (20-25%) and also played a role in enhancing production of secondary organic carbon mass (15%) on fine particles. Secondary fine particle nitrate was the single-most important source (35%) with a large fraction of this likely related to motor vehicle emissions. Independent use of different receptor models helps provide more confidence in the source apportionment as does comparison of results among complementary receptor-based data analysis approaches.

[1] Environment Canada, Downsview, ON.
[2] Vermont Department of Environmental Conservation, Waterbury, VT.
[3] Environment Canada, Gloucester, ON.

INTRODUCTION

The long lifetime (>5 days) of ambient fine particles ($PM_{2.5}$) enhances their transport over large distances (≥1000 km) leading to relatively large areas with elevated concentrations (regional haze), including rural locations (NARSTO, 2003; Brook et al., 1999), and to complex source-receptor relationships. The importance of secondary formation from ambient gases such as sulphur dioxide, nitrogen oxides, ammonia, volatile organic compounds and semi-volatile organic compounds (MSC, 2001), further complicates these relationships. Consequently, identifying sources responsible for high concentrations of $PM_{2.5}$ over populated areas (i.e., cities) is difficult. Clearly, this makes development of specific control strategies aimed at reducing the human health risk challenging.

Southern Ontario and its largest city, Toronto, experience elevated $PM_{2.5}$ related to regional-scale transport. Development of control strategies is further complicated by the fact that some of the sources are in the U.S. and thus, cannot be managed through federal or provincial regulations. Southern Ontario is not unique in this respect. $PM_{2.5}$ moves across international boundaries throughout Europe and among states within the U.S. This paper describes the current situation in southern Ontario and presents results of receptor-based data analyses geared towards obtaining a better understanding of the regional and local sources of $PM_{2.5}$.

Receptor-Based Methods

Receptor methods for source apportionment or attribution include a variety of approaches for interpreting measurements of the physical and chemical properties of ambient particles to infer their possible or probable sources and to quantify the contributions from these sources (Brook et al., 2003). In areas with a $PM_{2.5}$ or PM_{10} problem, receptor methods help identify possible solutions and thus are an important tool for analysis and development of policy and/or specific PM management strategies. Receptor methods are often referred to as receptor models. However, receptor models are actually one type or class of receptor methods that provide the theoretical and mathematical framework for quantifying source contributions. Receptor models contrast with source-oriented chemical transport models in that receptor models start with observations at a given location and work backwards using as much information as is practical to determine the sources contributing to the observations and to quantify the contribution. Chemical transport models combine detailed emission rates with meteorological transport, chemical changes and deposition rates to estimate concentrations and their temporal variations at a receptor and/or at an array of grid points distributed geographically. Chemical transport models can be used to predict how atmospheric concentrations could change if emission rates are changed, while receptor models are not developed for predictive purposes.

Several reviews of receptor modeling methods and capabilities have been published in the past (e.g., Watson, 1984; Gordon, 1988; Hopke, 1985; Henry, 1997; Watson and Chow, 2002; Brook et al., 2003). Examples of semi-quantitative observationally-based data analyses that are considered receptor methods are: (1) Time series plots of single day, average or median hourly $PM_{2.5}$ or PM_{10} concentrations; (2) Averaging by wind speed; (3) Comparisons among source-oriented, neighborhood-scale, urban-scale and regional-scale PM mass and chemical concentrations; (4) Concentration directionality. A summary of the types of receptor models that have been used for source apportionment along with their data requirements, strengths, and weakness is given in Brook et al. (2003). Most of the models are statistically-based, but some physically-based or combined physical-statistical methods have been applied.

In this paper, southern Ontario $PM_{2.5}$ monitoring data (Ontario Ministry of the Environment Tapered Element Oscillating Microbalance – TEOM) are analyzed using several semi-quantitative receptor methods. In addition, results from quantitative source apportionment of detailed $PM_{2.5}$ composition data (Lee et al., 2003) from downtown Toronto, Ontario, are presented. Two relatively new approaches, which have become popular in the research community were used. These are Positive Matrix Factorization (PMF) (Paatero, 1997) and the UNMIX model (Kim and Henry, 2000; Henry, 2002). PMF and UNMIX provide a physical basis for estimating source contributions and profiles when fairly stringent assumptions are met. PMF and UNMIX have undergone a series of tests with a variety of simulated and real data sets to better understand their strengths, weaknesses and similarities (Willis, 2000) and have been applied for source apportionment in a variety of locations (e.g., Polissar et al., 2001; Poirot et al., 2001; Chen et al., 2002; Lee et al., 2002).

RESULTS

Background

The $PM_{2.5}$ standard in Canada, referred to as the Canada-wide Standard (CWS), has been set at 30 μg m^{-3}, 24 hour averaging time, with achievement based on the three-year average of the 98th percentile. Formal reporting on achievement of the CWS does not begin until 2010 and, as a minimum, provinces are to report on all communities with more than 100,000 population. Communities for which jurisdictions demonstrate (i) that continued exceedance of the CWS levels is primarily due to transboundary flow of PM and ozone or their precursor pollutants from the U.S. or from another province/territory, and (ii) that "best efforts" have been made to reduce contributions to the excess levels from pollution sources within the jurisdiction, will be identified in reporting as "transboundary influenced communities" that are unable to reach attainment of the CWSs until further reduction in transboundary air pollution flow occurs. The CWS is clearly lower than the current U.S. daily standard of 65 μg m^{-3}. It also tends to be lower than the U.S. annual standard of 15 μg m^{-3} (i.e., areas can be above the CWS and still have an annual mean less than 15 μg m^{-3}).

An important issue with respect to a nationally consistent approach to achievement determination is the $PM_{2.5}$ measurement methodology. It is well known that different technologies can lead to different concentration values and that $PM_{2.5}$ is "defined" according to how it is measured. At present, Canada does not have a national standard for this method, which is in contrast to the U.S., where a Federal Reference Method (FRM) has been established. Many of the new $PM_{2.5}$ monitors across Canada are "continuous" measurement instruments such as the Tapered Element Oscillating Microbalance (TEOM) or the Beta Attenuation Monitor (BAM). These have been installed for practical purposes and to enable near real time data reporting for air quality index reporting and forecasting. However, current information indicates that the operating conditions of the TEOMs across Canada lead to significant loss of nitrate and for the determination of 24 hour average $PM_{2.5}$ concentrations this is most significant in the months from October to April (cold season) (Brook et al., 1999). Methods are being proposed to account for this low bias in cold season TEOM results, but at the present time these measurements are reported "as is" and often the warm season and cold season are investigated separately.

Figure 1 shows the current $PM_{2.5}$ levels, expressed in terms of the CWS metric, for all stations across the country (from west to east). These measurements are from TEOMs and are assumed to be biased low, as discussed above. Almost all measurements west of Ontario, have 98th percentile $PM_{2.5}$ concentrations of less than 30 μg m^{-3}. Starting in extreme SW Ontario and extending eastward into Quebec a number of monitoring sites are above the 30 μg m^{-3} value. This area, referred to as the Windsor-Quebec Corridor (WQC), includes several large population centres such as Windsor, London, Hamilton, Toronto and Montréal. Mean $PM_{2.5}$ levels are shown in Figure 2 on a map of the WQC. Some rural monitoring sites, such as Simcoe, Ontario, have relatively high $PM_{2.5}$ levels, indicating regional-scale transport is occurring. Sources in the U.S. are also implicated since in some locations, such as Simcoe, high $PM_{2.5}$ is observed under southerly flow conditions (see below) and there are no major sources between the site and the border. The highest concentrations shown in the figures are at Shawinigan, Quebec. This reflects the measurement site's industrial setting, being located 3.5 km south of a large aluminum smelter.

Mean $PM_{2.5}$ concentrations are higher in the urban areas since particles from local emissions are superimposed upon the regional background. In Hamilton and Toronto, the average $PM_{2.5}$ levels are about 45% and 20% higher, respectively, than at Simcoe. Thus, while urban activities clearly contribute to the $PM_{2.5}$ levels, a significant portion of the $PM_{2.5}$ observed in the cities is from upwind sources. Regional-scale concentrations over the WQC and the contribution from both local and upwind sources are strongly dependent upon meteorological conditions. Precipitation, wind direction, local and large scale stagnation and vertical mixing are some of the most important factors and many of the high $PM_{2.5}$ episodes leading to levels above the CWS are a result of specific weather patterns. Development of air quality management plans to reduce the risk to human health needs information on the relative impact of local versus upwind sources and on the main sources contributing to primary (i.e., directly emitted) and secondary (i.e., formed in the atmosphere from precursor gas emissions) $PM_{2.5}$. Although regional-scale air quality models are expected to be able to support development of management plans and to be able to address issues of local versus regional and primary versus secondary $PM_{2.5}$, validated models capable of operating on both regional

and urban scales are not yet available for widespread application. Fortunately, receptor-based methods utilizing observations can also be applied to provide policy-relevant information.

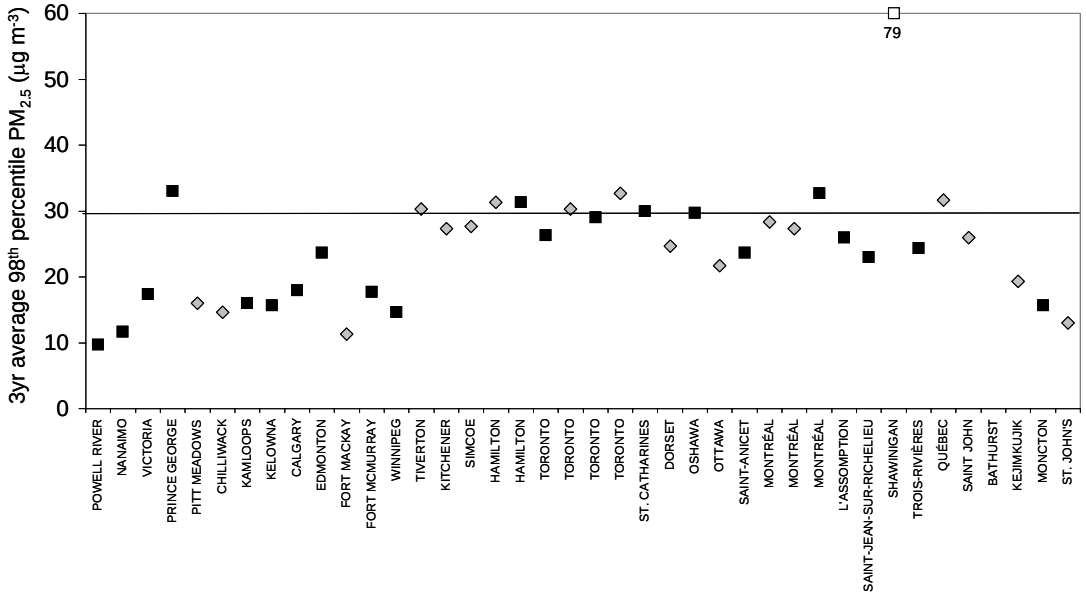


Figure 1. Canadian $PM_{2.5}$ levels expressed as the three-year average 98th percentile concentration. Squares are for sites with data for the three most recent years (2000-2002). Diamonds are for sites with at least three-years of measurements, but not all of these years were during 2000-2002. The solid line shows the value of the Canada-wide Standard, which is to be achieved by 2010. $PM_{2.5}$ was measured using a Tapered Element Oscillating Microbalance (TEOM).

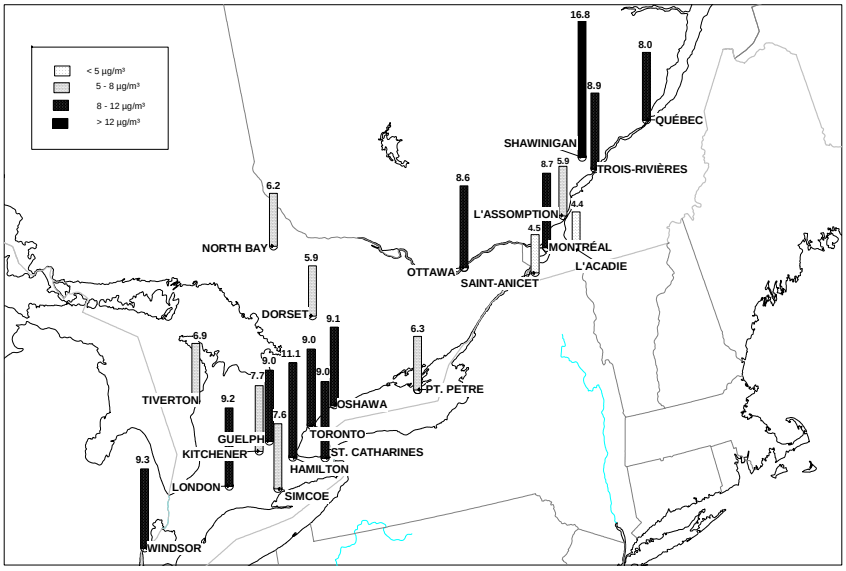


Figure 2. Annual average $PM_{2.5}$ concentrations in the Windsor-Quebec corridor for 2001.

APPLICATION OF RECEPTOR METHODS IN SOUTHERN ONTARIO

Concentration Directionality and Analysis of Spatial Patterns

Comparison of concentrations occurring with different prevailing wind directions provides a clear indication of where the main $PM_{2.5}$ sources are located. For example, in Brook et al. (2002) the median concentration at Simcoe under southerly flow was reported to be 20.3 μg m^{-3} and the frequency of 6 h observations above 30 μg m^{-3} was 18.5 %. In contrast, for northerly transport the median and frequency were 3.8 and 0.0, respectively. Clearly, reduction of regional $PM_{2.5}$ in southern Ontario will need to consider sources located to the south. These results are expanded upon in this paper by including another year of data, different measurement sites and by excluding periods with precipitation. Figure 3 shows the median six-hour average $PM_{2.5}$ for a southwest to northeast transect of sites running from Simcoe through the southern part of Hamilton (Hamilton Mtn.) and into south Toronto (Etobicoke) and north Toronto. The values shown are based upon the same observation periods at each site (i.e., equal sample size from the same 6 hr periods) and are based upon observations when there was no precipitation in the region. This approach provides a clearer picture of the differences between locations. Southerly and northerly flows were determined using three-day back-trajectories as described in Brook et al. (2002). It is important to note that the directionality of air masses is rarely straight and linear, and the actual trajectory often follows a more circuitous route between source and receptor. However, the trajectory sorting approach utilized here and in Brook et al. (2002) minimizes misclassification of transport direction. Median $PM_{2.5}$ was around 4-6 times higher under southerly transport conditions compared to northerly flow, depending upon location. While some differences in meteorological conditions (e.g., wind speed) may have contributed to this directional dependence of $PM_{2.5}$, the major cause was the difference in emissions to north of the region compared to south of the region.

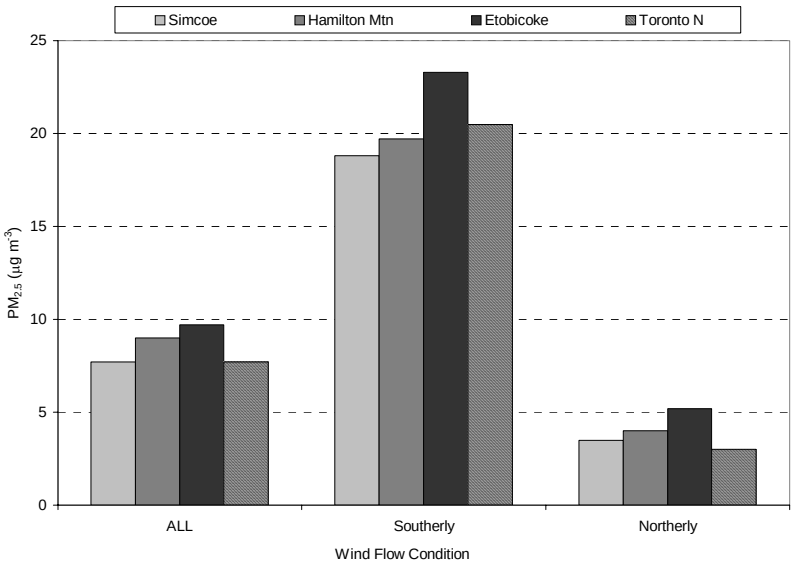


Figure 3. Median 6 hr $PM_{2.5}$ concentrations for the warm seasons (May-September) of 1998-2001 (non-precipitating days). "All" represents all measurements in the period. The sites shown represent a southwest to northeast transect from a rural location (Simcoe) to the outskirts of Hamilton and two sites in Toronto. Etobicoke is in the southwestern part of Toronto and is impacted by traffic.

Among the sites shown in Figure 3, $PM_{2.5}$ was highest at Etobicoke. This was due to the proximity of the site to traffic emissions and highlights the amount of intra-urban variability in Toronto. Site to site differences also varied by wind direction. Median $PM_{2.5}$ was 12% higher at Etobicoke compared to Toronto North during high-concentration southerly flow periods. In contrast, it was 66% higher under northerly flow conditions. This is due to the amount of $PM_{2.5}$ in the regional background. Not surprisingly, when the background levels are low, the within-city variation in $PM_{2.5}$ is more pronounced.

Comparing concentrations between the sites during the same prevailing wind directions can provide an indication of the local contribution to $PM_{2.5}$, at the urban sites. Determination of the magnitude of this contribution is crucial for effective risk management. For example, a small local contribution relative to the regional background suggests that risk management options based upon local emission control policies will not likely be effective. To estimate the local contribution to urban $PM_{2.5}$ the location of the rural site(s) with respect to the urban centre of interest is important (e.g., the rural site should not be downwind of the city). Simcoe is too far to the southwest to directly compare with Toronto concentrations since regional $PM_{2.5}$ levels gradually decrease from south to north. Therefore, in Brook et al. (2002), $PM_{2.5}$ from Simcoe and a rural site northwest of Toronto (Egbert) were used to estimate the average spatial gradient in regional $PM_{2.5}$ and hence the likely regional concentration relevant for Toronto. The urban concentrations were then compared to this estimated regional background to estimate how much of the $PM_{2.5}$ in Toronto was due to local emissions. Furthermore, this comparison, which was only done for the warm season (May-Sept.), was done separately for different wind flow directions. This was because regional background concentrations vary depending upon direction (Figure 3). For high concentration, southerly-transport-periods 30-38 % of the $PM_{2.5}$, on a total mass basis, in Toronto was estimated to be due to local sources. During westerly flow conditions the local contribution was 30-45%. The city was responsible for a larger percentage, up to 52%, when the flow was northerly because $PM_{2.5}$ levels in the air mass entering the city were relatively low.

Multivariate Receptor Models

Analysis of the mass concentration data among sites and for different pollutant transport directions led to better quantification of the amount of $PM_{2.5}$ coming from outside the city. However, this analysis provided little information on the actual sources responsible for the local or the upwind fractions. Therefore, receptor modelling was undertaken using one year of daily $PM_{2.5}$ mass and chemical speciation measurements. Applying two techniques, PMF and UNMIX, was expected to provide more confidence in the results. Agreement between techniques, especially when applied independently, will enhance the international credibility of the results, which may provide scientific support to the development of new Canada/U.S. air quality agreements.

Successful application of all receptor-modelling approaches for $PM_{2.5}$ depends upon detailed chemical characterization. For the Toronto analysis, 15 trace metals (e.g., iron, nickel, zinc, vanadium, selenium), 7 inorganic ions (e.g., sulfate, nitrate, ammonium), 5 water-soluble organic acids (e.g., oxalic acid, malic acid), total black carbon (BC) and four separate fractions of total organic carbon (OC1-4), were quantified (Lee et al., 2003). Both PMF and UNMIX also require a relatively large number of samples, preferably >300, and a high frequency of above detection limit concentrations. The PMF results are reported in detail in Lee et al. (2003). In this paper we focus on some of the UNMIX results and on a comparison of the apportionment obtained by the two approaches. We also only present the annual average apportionment, while seasonal results are included in Lee et al. (2003).

Five major sources and three minor sources were found by PMF to be contributing to the Toronto $PM_{2.5}$ (Lee et al., 2003). The UNMIX analysis, which was undertaken in two stages, uncovered a possibility of nine different sources. Again, five major sources were found and the other four were relatively minor. Both sets of results are summarized in the pie charts in Figure 4. Secondary ammonium nitrate was found to be the most important source of $PM_{2.5}$ mass. This $PM_{2.5}$ constituent forms during relatively cool and humid conditions from the oxidation products of nitrogen oxides (NO_x). Gas phase ammonia (NH_3) is also required. On a province-wide basis, 55-60% of the NO_x is emitted from motor vehicles and industrial and power generation emissions are equally responsible for another 30% (Environment Canada, 2003). Thus, the NO_x emissions inventory suggests that motor vehicles are the major source of the secondary ammonium nitrate. Vehicles operating in Toronto and upwind are likely both responsible, but their relative importance cannot be determined solely from receptor-based methods.

Another secondary constituent, associated with sulphate forming from emissions from coal combustion, was found by both PMF and UNMIX to be the second most important source of $PM_{2.5}$ in Toronto. The Ontario inventory (Environment Canada, 2003) indicates that 70% of the SO_2 emissions are from power plants and metal smelters. Although there is a small power plant in Toronto, most of the emissions from these sources are not released locally (i.e., not from within Toronto). Thus, reducing the “secondary coal” fraction of $PM_{2.5}$ in Toronto will require emission reductions upwind of the city. Both PMF and UNMIX attributed another ~20% of the $PM_{2.5}$ to secondary

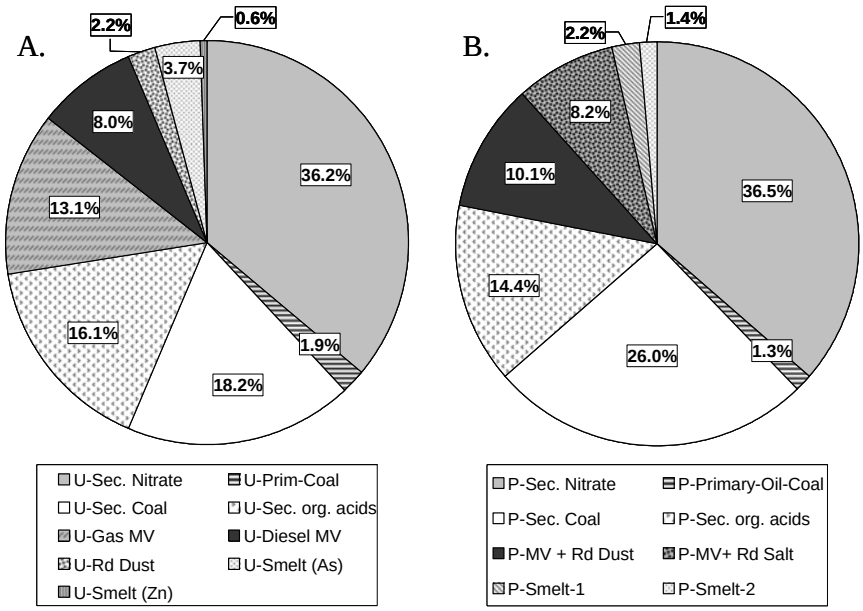


Figure 4. UNMIX (A) and PMF (B) receptor modelling results for 24 hr $PM_{2.5}$ collected in downtown Toronto from Feb. 14, 2000 to Feb. 14, 2001. Percentages indicate the estimated contribution to the annual $PM_{2.5}$ from the source indicated by the shading pattern.

organic carbon forming from gas phase volatile (VOC) and semi-volatile organic carbon (SVOC) emissions. Formation of this component of the $PM_{2.5}$ appeared to be enhanced by the presence of inorganic acids, predominantly acidic sulphate (Lee et al., 2003). The acidic sulphate is likely from upwind power plant emissions, while the main sources of the SVOC and VOC gases, could be located upwind and/or within Toronto. Natural and anthropogenic emissions are both potentially involved. Thus, approaches for reducing this component of the Toronto $PM_{2.5}$ are not clear based upon receptor method results.

The seasonal PMF results in Lee et al. (2003) revealed that secondary ammonium nitrate was responsible for 50% vs. 21% of the $PM_{2.5}$ in the cold vs. the warm season. Conversely, secondary coal and secondary organic carbon were greater contributors in the warm season. These sources were two and four times more important in the warm compared to the cold season, respectively. These differences, which are due to seasonal differences in meteorological conditions including more intense sunlight (i.e., photochemical activity), are important to be aware of when considering strategies to reduce $PM_{2.5}$ concentrations.

A number of local sources, linked to motor vehicle related emissions, were identified by PMF and UNMIX. In total, these sources were responsible for about 20% of the $PM_{2.5}$ with reasonable agreement between the two separate analyses (18% from PMF and 22% from UNMIX). However, while both models identified a distinct influence from motor vehicles (MV), PMF split this influence into two components, interpreted as “MV exhaust + road dust” and “MV exhaust + road salt,” and the UNMIX analysis split the MV contribution into three components, interpreted as “gasoline MV exhaust,” “diesel MV exhaust” and “road dust.” These differences were due to differences in how the PMF and UNMIX analyses were conducted. All of the measured $PM_{2.5}$ chemical constituents were used for PMF, while some were excluded in the UNMIX modelling in order to simplify interpretation. Most notably, sodium and chloride were not used and consequently, the influence of road salt was not identified. Instead, $PM_{2.5}$ mass associated with road salt was likely included as part of the “road dust” source. Another difference was that the UNMIX modelling was undertaken in two steps. The first step focused on apportioning the $PM_{2.5}$ mass using the inorganic ions, BC, total OC and selected trace elements. The second step focused on explaining the remaining mass based upon variations in OC1-4, BC, and inorganic and organic ions.

The two-step UNMIX modelling, with the second step emphasizing the OC fractions and BC, was advantageous because the gasoline and diesel emission sources were separated in step two. These sources were initially identified based upon the ratio of total OC (OC1 +OC2 + OC3 + OC4) to BC. The "Gasoline MV" source was found to be 80% carbon with an OC:BC ratio of 50:1. The "Diesel MV" source was 84% carbon with a ratio of 2.8:1 (i.e., much more BC). This identification was further supported by examining the day of week variation in the mass concentration of each of the UNMIX MV sources. Figure 5 shows that the diesel fraction of $PM_{2.5}$ declined substantially on weekends, which is consistent with urban traffic behaviour. In contrast, the gasoline MV fraction showed less day of week dependence, with maximum impact on Friday and Saturday. Both of these days typically have considerable automobile traffic spread throughout the day and into the night as opposed to typical workday rush hour patterns. Interestingly, the road dust contribution derived from UNMIX exhibited a day of week pattern representing a combination of the diesel and gasoline pattern. This characteristic is logical since road dust is presumably re-suspended by both types of vehicles. The strong day of week patterns exhibited by the MV-related sources implies that local traffic emissions were more important, as a day-of-week preference for emissions from more distant sources would be expected to be significantly attenuated by variable regional-scale transport times.

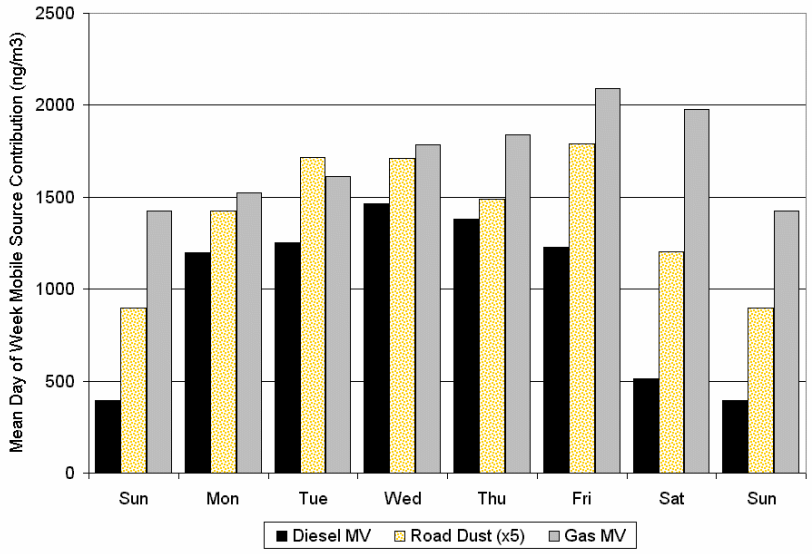


Figure 5. Mean day of week pattern in the $PM_{2.5}$ mass contribution (ng m^{-3}) from the three motor vehicle related sources determined by UNMIX for Toronto. Note: Sunday is shown twice.

In addition to the main sources described above, both PMF and UNMIX apportioned the remaining ~5% of the $PM_{2.5}$ among three primary sources. These were potentially linked to industrial emissions, including metal smelters, and primary coal and/or oil burning emissions. Some characteristics of these sources were similar between the two methods. For example, a high selenium (Se) to sulphate ratio was the main characteristic of the primary coal source and arsenic (As) was the main 'marker' for one of the sources labeled as being related to smelters. The main common feature between the two sets of results, however, was the minor mass contribution these sources were found to have on the observed $PM_{2.5}$ in Toronto.

In general, the independent application of different receptor models provided more confidence in the results. However, it is important to note that in both approaches, as with most other receptor models, deducing the number of sources is not straightforward. For example, Lee et al. (2003) reported that an eleven-source solution for Toronto also produced reasonable results. Similar model comparisons for Phoenix, AZ, $PM_{2.5}$ also demonstrated this fact (Willis, 2000). Thus, even when reasonable agreement is obtained, as reported above for Toronto, the results need to be interpreted in light of uncertainties in the measurement data, variations in environmental conditions and in actual emission characteristics and in light of the inherent subjectivity involved in determining the number of sources and uncertainty associated with the receptor method. PMF considers uncertainties in the measurements and error bars in

the source contributions are provided in Lee et al. (2003). However, accurate uncertainty estimates encompassing all of the issues are very difficult to quantify and will vary among different analyses and datasets. Nonetheless, some idea of uncertainty is important for decision-making. Applying multiple techniques, as presented here, provides a range of results, which can be viewed as an indication of overall uncertainty. Including analyses of different $PM_{2.5}$ datasets in these comparisons would also be valuable. For Toronto, two additional years of measurements are currently being used for this purpose. Ultimately, the receptor method results also need to be reconciled with known characteristics of the airshed of interest (e.g., general knowledge or inventory-based information of the types of local and regional sources expected to contribute to $PM_{2.5}$) and they need to be carefully examined to determine if the temporal (e.g., seasonal and day of week patterns) and meteorological (e.g., variation with wind direction) characteristics are realistic.

The results from the receptor models are not inconsistent with our earlier estimate of a 30-45% contribution from local $PM_{2.5}$ sources. Simply assuming that the motor vehicle component (~20%) and about 50% of the secondary nitrate, which amounts to 10% of the $PM_{2.5}$ in the warm season, are locally emitted/produced leads to consistency, although at the lower end of the range. In reality, the other sources found by the receptor models were also likely to have had a regional and local component. All proportions would likely have changed from sample to sample, as well. However, larger portions of the motor vehicle and secondary nitrate sources were clearly from local sources compared to the secondary coal and the secondary organic acid sources.

CONCLUSIONS

Application of receptor-based methods can provide information useful for managing the risk to human health attributed to poor air quality. In urban areas experiencing complex source-receptor relationships, due to a significant impact by regional-scale transport and/or due to a high population density in the surrounding area, these methods are essential for gaining a better understanding of the $PM_{2.5}$ issue.

The type, quantity and spatial-temporal resolution of the $PM_{2.5}$ data that are available governs the amount of information that can be obtained from receptor-based analyses. For assessing regional vs. local contributions, operating at least one site to measure the regional background $PM_{2.5}$ along with the urban network is critical. For Toronto, Ontario, several measurement sites within the city and in the surrounding area were compared under specific wind flow patterns to estimate that 30-45% of the $PM_{2.5}$ is locally generated. During periods of low regional background concentrations, which usually occur with northerly wind flow, the percent local contribution is largest. This percent can surpass 50% at urban sites significantly influenced by traffic emissions.

Detailed chemical characterization of $PM_{2.5}$ is essential for application of receptor models. The more complete this characterization and the greater the amount of data available the more confident source apportionment results can be expected to be. Given the inherent uncertainty in measurement data, in actual source profile information and the subjective nature of the interpretation of receptor model results, applying more than one model can help increase confidence in results. Use of multiple models can be even more effective if they are run by independent groups of experienced investigators. This procedure was followed for Toronto $PM_{2.5}$ source apportionment and the separate receptor model runs were found to be in good agreement. This apportionment indicated that motor vehicle related emissions (i.e., exhaust and road dust), most likely of local origin, were responsible for about 20% of the $PM_{2.5}$. Gasoline engine vehicles were found to be a greater overall contributor compared to diesel vehicles. Secondary $PM_{2.5}$ from coal-fired power plant emissions was a significant contributor and also played a role in enhancing production of secondary organic carbon mass on fine particles. Secondary fine particle nitrate was found to be the single-most important source, particularly in the cooler months. Based upon the current Ontario emissions inventory, 55-60% of the observed fine particle nitrate in Toronto is estimated to be related to motor vehicle NO_x emissions.

The information provided through the receptor-based analyses presented in this paper can be used to target potential strategies for reducing ambient $PM_{2.5}$ in Toronto. However, it is important to note that these results do not provide information on which of the identified $PM_{2.5}$ sources produce the most toxic (acute or chronic) particles. Clearly, air quality improvement strategies should, if possible, attempt to consider the relative toxicity of the mix of emissions originating from the various sources, thereby leading to ambient $PM_{2.5}$ reduction strategies providing the greatest

benefit to public health. Additional research is needed to gain a better understanding of which sources and or $PM_{2.5}$ chemical constituents or precursors are more strongly linked to adverse human and environmental health impacts.

REFERENCES

Brook, J.R., Vega, E., and Watson, J.G. 2003. *Receptor Methods*, Chapter 7 – North American Research Strategy on Tropospheric Ozone (NARSTO) Particulate Matter Assessment. Released March 2003 www.cgenv.com/Narsto/PMAssessment.html.

Brook, J.R., Lillyman, C.D., Mamedov, A., Shepherd, M. 2002. Regional transport and urban contributions to fine particle ($PM_{2.5}$) concentrations in southeastern Canada. *J. Air Waste Manage. Assoc.* 52:174-185.

Brook, J.R., Dann, T.F., and Bonvalot, Y., 1999. Observations and interpretations from the Canadian fine particle monitoring program. *J. Air Waste Manage. Assoc.* 49:35-44.

Chen, L.W.A., Doddridge, B.G., Dickerson, R.R., Chow, J.C., and Henry, R.C. 2002. Origins of fine aerosol mass in the Baltimore–Washington corridor: implications from observation, factor analysis, and ensemble air parcel back trajectories. *Atmos. Environ.* 36:4541-4554.

Environment Canada 2003. National Pollutant Release Inventory (NPRI), www.ec.gc.ca/pdb/npri/npri_home_e.cfm.

Gordon, G.E. 1988. Receptor Models. *Environ. Sci. Technol.* 22:1132-1142.

Henry, R.C. 2002. Multivariate receptor models - current practice and future trends. *Chemom. Intell. Lab. Sys.* 60:43-48.

Henry, R.C. 1997. History and fundamentals of multivariate air quality receptor models. *Chemom. Intell. Lab. Sys.* 37:37-42.

Hopke, P.K. 1985. *Receptor Modeling in Environmental Chemistry*. New York: John Wiley & Sons, Inc.

Kim, B.-M., and Henry, R.C. 2000. Extension of self-modeling curve resolution to mixtures of more than three components. Part 3, *Chemom. Intell. Lab. Syst*. 52:145-154.

Lee, P.K., Brook, J.R., Dabek-Zlotorzynska, E., and Mabury, S. 2003. Identification of the major sources contributing to $PM_{2.5}$ observed in Toronto. *Environ. Sci. Technol.* 37:4831-4840.

Lee, J.H., Hopke, P.K., Poirot, R.L., Lioy, P.J., and Oxley, J.C. 2002. Identification of sources contributing to mid-Atlantic regional aerosol. *J. Air Waste Manage. Assoc* 52:1186-1205.

MSC (Meteorological Service of Canada) 2001. Precursor Contributions to Ambient Fine Particulate Matter in Canada: A Report by the Meteorological Service of Canada (MSC). Ministry of Public Works and Government Services, 2001. En56-167/20001E, ISBN: 0-662-30650-3, May 2001.

NARSTO 2003. North American Research Strategy on Tropospheric Ozone (NARSTO) Particulate Matter Assessment. Released March 2003 www.cgenv.com/Narsto/PMAssessment.html.

Paatero, P. 1997. Least squares formulation of robust, non-negative factor analysis. *Chemom. Intell. Lab. Sys.* 37**:**23-35.

Poirot, R.L., Wishinski, P.R., Hopke, P.K., and Polissar, A.V. 2001. Comparitive application of multiple receptor methods to identify aerosol sources in northern Vermont. *Environ. Sci. Technol.* 35:4622-4636.

Polissar, A.V., Hopke, P.K., and Poirot, R.L. 2001. Atmospheric aerosol over Vermont: chemical composition and sources. *Environ. Sci. Technol.* 35:4604-4621.

Watson, J.G. 1984. Overview of receptor model principles. J. Air Pollut Control Assoc. 34:619-623.

Watson, J.G., and Chow, J.C. 2002. Particulate pattern recognition. In *Introduction to Environmental Forensics*, eds. B.L. Murphy, and R. Morrison, pp. 429-460. New York: Academic Press.

Willis R.D. 2000. Workshop on UNMIX and PMF as applied to $PM_{2.5}$. Final Report for USEPA under Contract No. 68-D5-0049. Office of Research and Development, National Exposure Research Laboratory, Human Exposure and Atmospheric Sciences Division, June 2000, EPA/600/A-00/048.

MODELLING THE INTRA-URBAN VARIABILITY OF AMBIENT TRAFFIC POLLUTION IN TORONTO, CANADA

M. Jerrett,[1] M.A. Arain, P. Kanaroglou, B. Beckerman, D. Crouse,,[2] N.L. Gilbert,[3] J.R. Brook,[4] N. Finkelstein,[2] M.M. Finkelstein[5]

ABSTRACT

The objective of the paper is to model determinants of intra-urban variation in ambient concentrations of nitrogen dioxide (NO_2) in Toronto, Canada, with a land use regression (LUR) model. Although researchers have conducted similar studies in Europe, this work represents the first attempt in a North American setting to characterize variation in traffic pollution through the LUR method. NO_2 samples were collected over two weeks using duplicate two-sided Ogawa passive diffusion samplers at 95 locations across Toronto. Independent variables employed in subsequent regression models as predictors of NO_2 were derived by the Arc 8 geographic information system (GIS). Some 85 indicators of land use, traffic, population density, and physical geography were tested. The final regression model yielded a coefficient of determination (R^2) of 0.69. For the traffic variables, density of 24-hour traffic counts and road measures display positive associations. For the land use variables, industrial land use and counts of dwellings within 2000 m of the monitoring location were positively associated with NO_2. Locations up to 1500 m downwind of major expressways had elevated NO_2 levels. The results suggest that a good predictive surface can be derived for North American cities with the LUR method. The predictive maps from the LUR appear to capture small-area variation in NO_2 concentrations. These small-area variations in traffic pollution are probably important to the exposure experience of the population and may detect health effects that would have gone unnoticed with other exposure estimates.

[1] University of Southern California, Los Angeles, CA, USA.
[2] McMaster University, Hamilton, ON, Canada.
[3] Health Canada Ottawa, ON, Canada.
[4] Environment Canada, Downsview, ON, Canada.
[5] University of Toronto, Toronto, ON, Canada.

INTRODUCTION

Policymakers and scientists have shown growing interest in the health effects of chronic exposure to ambient air pollution. Traffic-related air pollution is of particular interest from a regulatory perspective because the demand for transportation will probably outpace improvements in vehicle technologies over the next decade (Delucchi, 2000). European studies reporting large health effects for persons living close to major roads have also heightened concern about traffic pollution (Hoek et al., 2002). In spite of the interest in traffic pollution sparked by these coalescing concerns, uncertainties in exposure assessment methodologies continue to raise questions about the reliability and accuracy of risk estimates from chronic air pollution studies. This scientific uncertainty impedes efforts by policymakers to implement effective traffic pollution control programs.

In this context, it was sought to model determinants of intra-urban variation in ambient concentrations of nitrogen dioxide (NO_2) in Toronto, Canada, with a land use regression (LUR) model. NO_2, an important inorganic gas, serves as a good indicator of intra-urban traffic pollution (Niewenhuijsen, 2000). Although researchers have conducted similar studies in Europe, this work represents the first attempt in a North American setting to characterize variation in traffic pollution through the LUR method.

Background

LUR employs the pollutant of interest as the dependent variable and proximate land use, traffic, and physical environmental variables as independent predictors. Thus the methodology seeks to predict pollution concentrations at a given site based on surrounding land use and traffic characteristics. Specifically, this method uses measured pollution concentrations (y) at locations (s) as the response variable and land use types (x) within circular areas around s (called buffers) as predictors of the measured concentrations (see Figure 1). The incorporation of land use variables into the interpolation algorithm detects small-area localized variations in air pollution more effectively than standard methods of interpolation such as kriging (Briggs et al., 1997; Briggs et al., 2000; Lebret et al., 2000).

To date, LUR studies of criteria air pollutants have been conducted exclusively in Europe. Two studies (Briggs et al., 1997; Lebret et al., 2000) were part of the Small Area Variation in Air pollution Health (SAVIAH) Project that examined traffic-related air pollution in four European cities (Amsterdam, Huddersfield, Prague, Poznan). An updated model described by Briggs et al. (2000) investigates traffic-related air pollution in four United Kingdom (UK) urban areas (Huddersfield, Hammersmith and Ealing, Northampton, and Sheffield). The independent variables used for the prediction of mean NO_2 were road traffic volume, land-use type, and elevation. These variables produced good predictions with coefficient of determination (R^2) values ranging from 0.79 - 0.87.

More recently, Brauer et al. (2003) compared traffic-related $PM_{2.5}$ air pollution models in multiple European cities using the LUR technique. In each of the study cities, investigators fit two types of models: one available through a geographic information system (GIS) environment and another that included additional variables not available in the GIS. The results obtained for the Netherlands,[6] Munich, and Stockholm in the GIS environment showed R^2 values of 0.81, 0.67 and 0.66 for particle filter absorbance, respectively. The alternate model or "best" model included variables such as high traffic locations and street canyons. This model produced results with better R^2 values respectively of 0.90, 0.83, and 0.76.

Although no North American studies are directly analogous to the European studies, one American study has used the LUR approach to model variability in carbon dioxide (Wentz et al., 2002). This study was more concerned with understanding the land use and vegetative characteristics associated with this greenhouse gas rather than assessing exposures directly related to health effects. The results were more modest than in the European studies discussed above, with R^2 values ranging from 0.54-0.74. These results raise questions about whether the LUR method will perform adequately in North American cities, particularly for traffic pollutants such as NO_2 that display significant variation over scales as small as 50 metres (Hewitt, 1991).

[6] Multiples sites across many cities in the Netherlands, too numerous to mention, were sampled for this study.

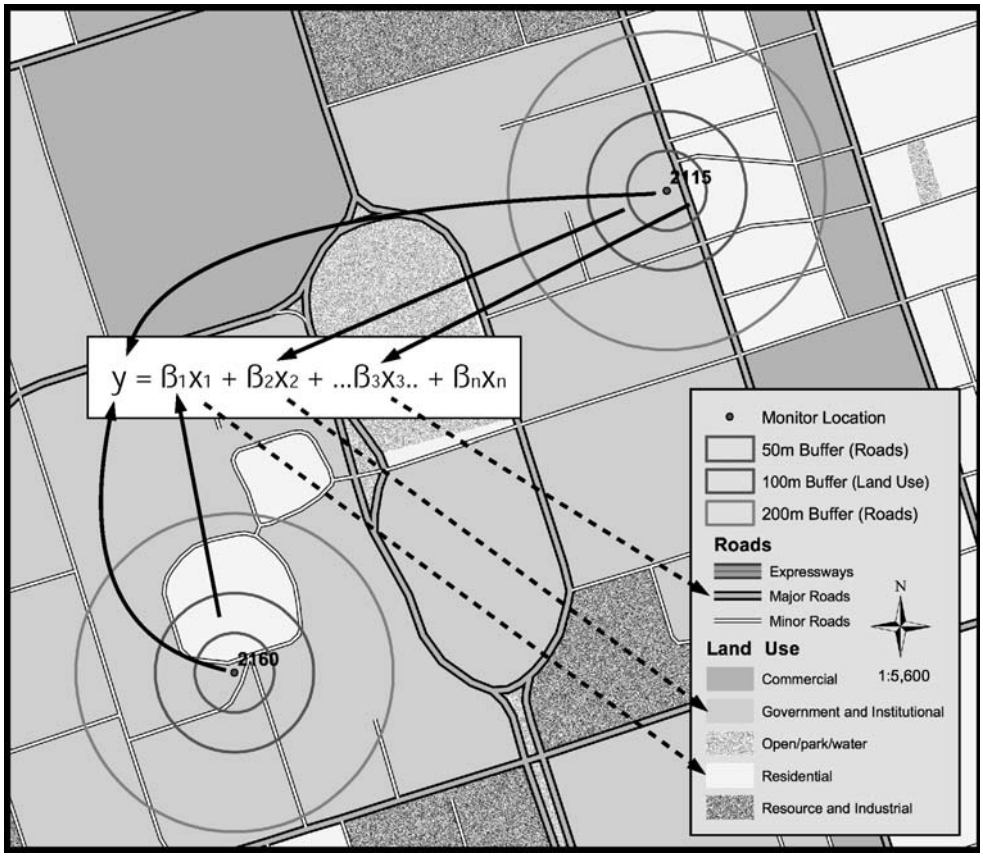


Figure 1: Elements of a land use regression model showing monitoring locations for NO_2 as the response variable and land use characteristics within buffers as the predictor or independent variables.

In addition to these empirical results, other differences in land use and traffic distinguish newer North American cities from those in Europe that evolved earlier into major conurbations. Fowler (1992) outlines these land use and transportation characteristics as deconcentrated, decentralized, large scale, homogenous and segregated. Compared to European centres, North American cities overall have lower population densities (deconcentrated), and a greater proportion of the population lives in suburban areas outside the city centre (decentralized). Moreover, cities in North America tend to have land use developments that occupy large tracts of land for single land use types, making their land use homogenous and large-scale (e.g., large residential subdivisions of many thousand houses, huge commercial shopping centres, and large industrial zones). And, individual land use types are segregated from each other creating a need for automotive travel between commercial and residential areas. All of these factors contribute to higher levels of automobile use (Newman and Kenworthy, 1989), energy consumption (Hough, 1995), and subsequent higher levels of pollution emissions. Each of these contrasts raises questions about the applicability or adaptability of the LUR methods developed in Europe to North America.

METHODS

This section outlines the study setting, the data sources used in the analyses, and the methods used to estimate NO_2 concentrations. NO_2 measurements are technically "mixing ratios," not concentrations, as these values do not incorporate a mass weight. For ease of reading and by convention, the wording concentration has been used in the remainder of this article.

Study Area

Toronto is the provincial capital of Ontario and Canada's largest city (estimated population: 2.6 million people, Statistics Canada, 2001; approximate area: 633 km^2). It is located on the north shore of Lake Ontario (situated at 43° 39' N, 79° 23' W) with a climate classified in the "Humid East" region of temperate North America (Getis and Getis, 1995). Similar to other large cities in North America, many expressways traverse the Toronto landscape, including some of the busiest in North America (e.g., Highway 401 has peak flows of about 400,000 vehicles per day).

Data - Measurements and Preparation of the Dependent Variable (NO_2)

NO_2 concentrations were measured for a two week period from September 9 to 25, 2002 at 100 locations across Toronto. NO_2 was selected to proxy for traffic related air pollution because it is relatively inexpensive to measure and has been used widely as a metric of exposure to traffic emissions (Briggs et al., 1997). To capture small-area variation in the NO_2 concentrations, 100 sampling locations across the city were used. NO_2 may display significant differences on scales as small as 50 m (Hewitt, 1991), and hence a dense monitoring network is needed to measure these small-area variations. This number is in line with the study by Lebret et al. (2000), that used between 70 and 80 NO_2 monitors for measurements in Amsterdam, Netherlands (land area of 25km^2), and Huddersfield, UK (land area of 300km^2), considering Toronto has a larger land area of 633km^2. Sampling locations were selected using a population-weighted location-allocation model based on potential NO_2 variability and the density of children aged 0-6, as this exposure assessment represents the first stage of a childhood asthma study (Kanaroglou et al., 2003). The outcome of using a location-allocation model is a sampling network that better captures the inherent variability in city-wide exposures.

Ogawa™ passive samplers were used to measure concentrations of NO_2. Two-sided samplers were deployed in pairs (yielding four observations per site) at a height of 2.5 meters because this was the first time that Ogawa NO_2 monitors were used for this type of ambient monitoring. The deployment of samplers took less than 72 hours. All samplers were removed 14 days after their installation. The nitrite content on collection pads was determined by ion chromatography (Gilbert et al., 2003). For each location, the arithmetic mean, standard deviation, and coefficient of variation of NO_2 results were calculated.

Only five of the 100 samplers deployed were removed due to vandalism or invalid measurements, leaving 95 observations for analysis. Figure 2 illustrates the locations of these 95 monitors against the backdrop of different land uses. Additionally, one more sample was removed from the analysis. It was a significantly high outlier, which further investigation revealed the presence of an active construction site in proximity to the monitor over the sampling period.

Collected NO_2 data were then checked thoroughly for consistency and reliability prior to analysis. There were two criteria used for this initial assessment. First, each site had to have at least two valid measurements. Sites with only one valid measurement were excluded from the analysis, as there was no means to verify the concentration observed with duplicate observations. Second, the coefficient of variation among observations at a given site (COV = *standard deviation/mean*) had to be less than 0.25 to count as valid. If the COV was greater than 0.25, then the observation that created the greatest amount of variability was removed and the first criterion was applied again. For cases with COV of 0.25 or greater, this process was performed until there were only two measurements remaining at each location. If COV remained greater than 0.25, then the entire sampling location was discarded.

As an initial exploratory procedure to determine overall trends and local autocorrelation in the NO_2 data, ArcGIS 8 software (ESRI Corp, Redlands, CA) was used to implement the geostatistical interpolation method known as 'kriging' (using the spherical model). After examining these general patterns in the data the LUR model was developed.

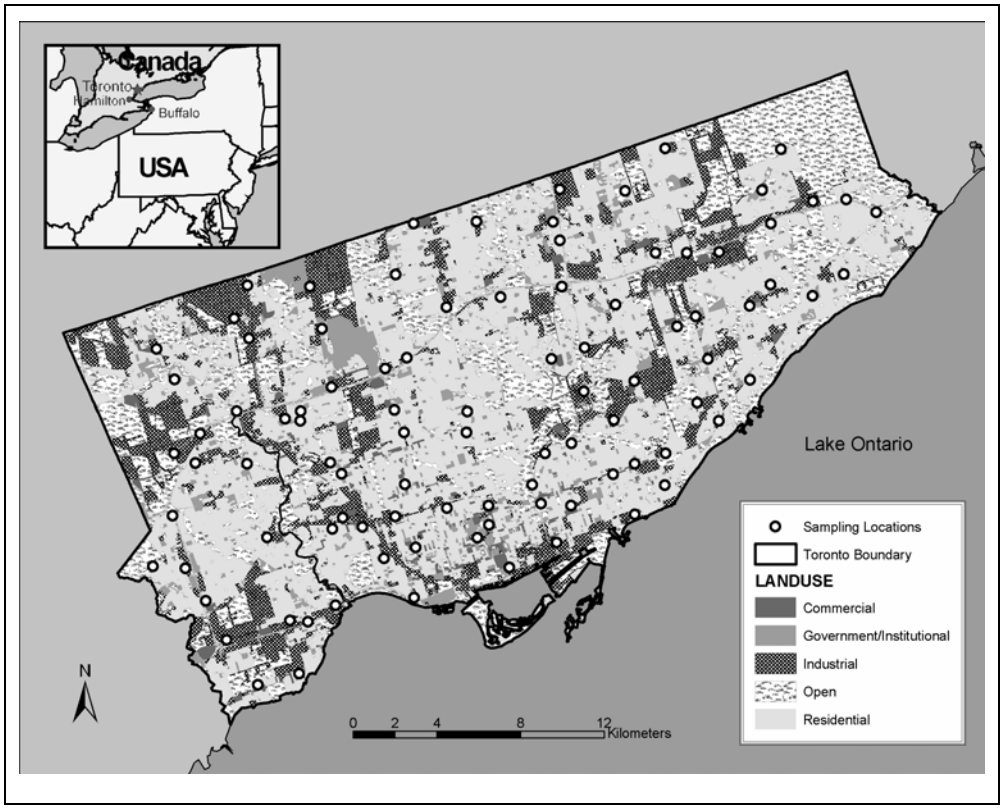


Figure 2: Toronto sampler locations against a backdrop of land use classification.

Data for Independent Variables

In total, 85 independent variables, or variations of different variables, were created with these data using ArcGIS 8 (see Appendix 1 for the complete list of variables tested in the analysis). The variables were grouped into five broad categories: (1) land use (area of different land uses within buffers of various radii around each sampling location); (2) road and traffic (lengths of different road types and traffic flow counts within buffers of different radii); (3) population (population density, density of dwelling units, average dwelling values); (4) physical geography (geographic location in terms of X and Y coordinates, and elevation); and (5) meteorology (wind direction in relation to major emission sources).

Toronto land use and road network data were acquired from a commercial source (i.e., DMTI Spatial Inc., Markham, ON, Canada). Average daily expressway or highway traffic counts were obtained from Environment Canada, and the City of Toronto Information Services provided similar data for major roads throughout the city. Population data were compiled from the Statistics Canada 1996 Census of Population (these were the most recent data available in GIS format). A digital elevation model developed by the Ontario Ministry of Natural Resources was compiled through ArcGIS 8.2 at a 5-metre resolution. Meteorological data were obtained from a network of 15 surface observation stations operated by various government agencies and private organizations in the Toronto region, including Meteorological Service of Canada, Ontario Ministry of Environment, and Ontario Power Corporation. The configuration of the network defined a modelling domain of some 8000 km^2. This ensured an interpolated (and not an interpolated-extrapolated composite) wind field prediction by maintaining a variable-width (22-35 km) buffer of data points around City of Toronto's footprint, thereby reducing the likelihood of erroneous prediction.

In total, 17-day (September 9-25) average zonal (E-W) and meridional (N-S) orthogonal components of wind – *u* and *v*, respectively – for the 17:01-18:00 segment of the day were calculated for each station. This hour not only represents the daily afternoon peak traffic flow for the 17 days period that the samplers were deployed, but it is also representative of the overall prevailing wind patterns. The orthogonal components were then used, following Goodin et al. (1980), as urban wind field datasets to assess the pollution-wind relationship.

An interpolation method found to be well suited for modelling vector fields – the Radial Basis Functions (RBF) Multiquadric (MQ) method – was used for wind field construction. MQ interpolation has been used in meteorology and in related disciplines for more than 30 years (e.g., Shaw and Lynn, 1972; Lynn, 1975; Sirayanone, 1988; Nuss and Titley, 1994; Hubbe et al., 1997) The *u* and *v* components of wind were interpolated as separate scalar entities, which then allowed for wind direction vector calculations to be performed within the ArcGIS framework, with the ultimate goal being the determination of the upwind-downwind relationship between the high traffic expressways and the wind direction. To determine this relationship, the interaction between two vector sets is obtained by applying the dot product operation:

$$\mathbf{a} \cdot \mathbf{b} = a_1 b_1 + a_2 b_2 = |\mathbf{a}||\mathbf{b}| \cos\theta \quad \text{(eq.1)}$$

Where **a** and **b** are the direction vectors with components a_1 and a_2 and b_1 and b_2, representing the direction to the nearest expressway and wind direction, with θ the angle that lies between two of them. This second part of the equality in equation 1 is an identity that provides two pieces of information: (1) the angle that accounts for the degree of relationship between the wind vector and expressway, and (2) the sign of $\cos\theta$ shows whether an expressway lies up or down wind of any specific point of interest in study area for that particular wind field. A negative value translates into a grid cell located downwind of an expressway. The opposite is true for positive values – grid cells are located upwind of an expressway.

RESULTS

The 94 NO_2 samples used in the analysis had an arithmetic mean of 32.2 ppb, with values ranging from 17.4 to 61.1 ppb (SD = 9.2). There were 19 samples that had a value greater than one standard deviation from the mean, i.e. greater than 41.4 ppb. Of these, 15 were located in proximity to expressways with the remaining 4 located within heavily trafficked corridors. Very good agreement was found between the paired Ogawa samples.

Model Selection

The NO_2 measurements were transformed with the natural logarithm because of a strong right-skew. Each of the 85 independent variables were tested through an individual bivariate regression model with SPSS 11.5. This identified variables that were highly correlated with the NO_2 observations. Appendix 1 shows the R^2 and t-score results from each bivariate analysis. Overall, the traffic and road length variables displayed the strongest association with NO_2 concentrations. Specifically, the density of roads within a 300-metre radius buffer of each sampler location (Rd_density) produced the best bivariate model, with a t-score of 7.03 and an R^2 of 0.35.

In the following step, the Rd_density variables were paired with other significant variables in a series of trivariate regression models. This series of analyses identified the pairing with population density (as calculated through a kernel estimate with population-weighted enumeration area (EA) centroids, and a 2000-m search radius (EA2000)), as the best two-variable combination for predicting NO_2 exposures. The process of manual forward stepwise regression was performed until five independent variables were included in the model. Although useful for exploration, the results were hampered by high levels of collinearity between the variables.

Other variable combinations were also tested with various "best subsets" regression analyses in Minitab 12.22. Best-subsets regressions generate regression models with the "best" R^2 values, mean square error predictions, and the Mallow Cp statistic, which is an indicator of how well the model fits the data without introducing bias (Hamilton, 1992). Due to limitations on the number of variables allowed by the software in each selection, the best subsets were limited to the most significant variables discerned through the bivariate and manual stepwise screening. Wind direction variables were added to the parsimonious model selected from the above procedures because the wind models and data took considerable time to construct.

After determining the appropriate model, standard regression diagnostics were applied to assess problems such as outliers, heteroskedasticity, and spatial autocorrelation. A series of cross-validation tests were also completed to assess the predictive capacity and stability of the final set of models used in the analysis.

Table 1 shows the model for predicting intra-urban variation in NO_2. This model produced an R^2 of 0.69. Each individual variable has a significant t-score and acceptable multicollinearity, as demonstrated by the Variance Inflation Factors (VIF). All of the coefficients have the expected signs. For the traffic variables, density measure of 24-hour traffic counts (TRAF500) and road measures (RD2_50 and RD1_200) display positive associations. For the land use variables, industrial land use (IND750) within 750 m, and counts of dwellings within 2000 m (DC2000) of the monitoring location were positively associated with NO_2 concentrations. Locations up to 1500 m downwind of major expressways (D_WIND15) had elevated NO_2 concentrations. A trend was observed in the data with higher NO_2 concentration values in the west to lower values in the east (X).

Table 1: Summary of the regression results for the logarithmic NO_2 Model.

				Number of obs =	94
Source	**SS**	**df**	**MS**	F(7,87) =	27.4
Regression	5.09	7	0.727	Prob > F =	0
Residual	2.29	86	0.027	R-square =	0.69
Total	7.38	93		Adj. R-square =	0.67
				Root MSE =	0.163

Variable*	**Coefficient**	**Std. Error**	**t**	**Prob > t**	**VIF**
LN(NO_2)					
(Constant)	8.06E+00	1.177	6.85	0.00	
RD1_200	1.84E-01	0.020	9.04	0.00	1.20
RD2_50	5.56E-01	0.300	1.85	0.07	1.11
IND750	1.63E-03	0.001	3.04	0.00	1.20
DC2000	8.28E-05	0.000	4.66	0.00	1.38
X	-8.01E-06	0.000	-4.31	0.00	1.06
D_WIND1500	1.32E-01	0.040	3.30	0.00	1.20
TRAF500	1.11E-03	0.001	1.96	0.05	1.32

*RD1_200 – measure of expressway within 200m; RD2_50 – measure of major roads within 50m; IND750 – measure of industrial land use within 750m; DC2000 – density of dwellings within 2000m (Kernel estimate); X – UTM NAD83 x-coordinate; D_WIND15 – Boolean identifier whether downwind and within 1500m of nearest expressway at PM-peak traffic; TRAF500 – Density measure of 24 hour traffic counts within 500m.

The scatter plot presented in Figure 3 demonstrates that this model produces reliable predictions with no significant outliers or heteroskedasticity. Examination of Cook's distance and leverage statistics confirmed the absence of significant outliers. Additionally, Moran's *I* tests suggest that spatial autocorrelation is insignificant in this model when using a first-order adjacency matrix.

Cross Validation of Regression Results

Several cross-validation analyses were also undertaken to confirm the predictive capacity and stability of the results. First, the regression model was run with a random selection of only 65 of the records. This action was repeated several times and produced comparable results to those achieved from the full data set in each instance. Each model with only 65 cases produced results that were remarkably similar to the model with all 94 cases.

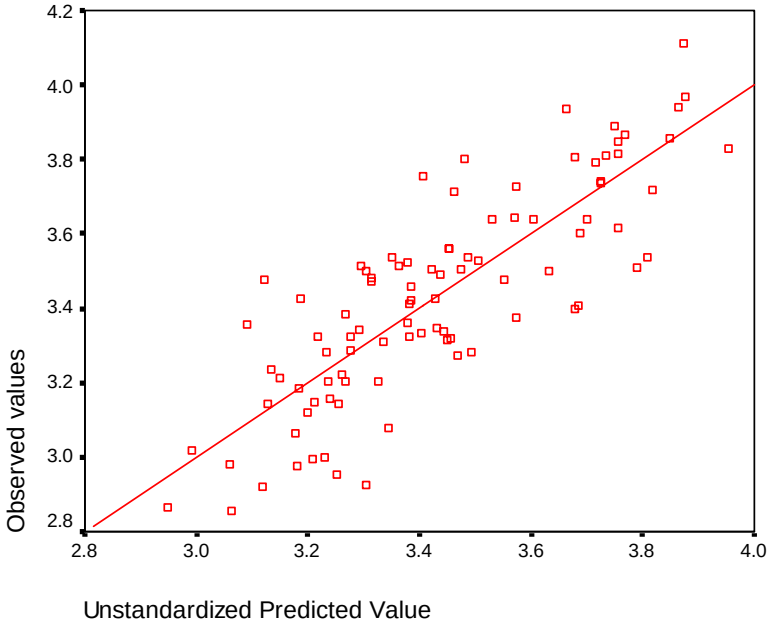


Figure 3: Logarithmic-observed mean NO_2 on predicted value.

Second, the coefficients were used from the model using 65 cases to predict NO_2 concentrations at the 29 excluded sampling locations. The model slightly over predicted for these locations, but the average difference was small in both absolute and relative terms. In each of the cross-validations, the average difference was not greater than 2 parts per billion, translating into an average relative difference of less than 4.0%. The results of this comparison are presented in Appendix 2.

Third, an attempt to compare the modeled results with the data collected by government operated continuous monitoring stations for five co-located sites for the 17-day sampling period was made. Data for only 3 of the 5 co-located sampling locations were available. Using government monitored levels as a reference datum for gauging measurement error by the passive Ogawa samplers, it was possible to conduct groups of impartial validations. First, a validation using longer term averages was also implemented. Five-year (1997-2001) average and the September, 2001, average NO_2 concentrations were available through Environment Canada's National Air Pollution Surveillance Network (NAPS) reports. Second, a temporally matched validation (the two week period in September 2002) using Ontario Ministry of Environment (MOE) data was conducted. The modeled values were also compared to the Ogawa measurements made under this study in September 2002. These results indicated that the differences in monitoring technology and temporal variation produce relatively moderate errors when comparing measured values to modeled values (Table 2). As more temporally matched data are used, the relative error is similar between the land use regression model and the government data decreases (Table 3). Yet, in the ideal case, more government monitoring sites would be necessary to conduct a thorough statistical analysis of the error.

Table 2: Comparison of the land use regression model (LUR) to National Air Pollution Surveillance network (NAPS) and respective Ogawa sampled values.

Station Location	LUR predicted NO_2 ppb (Sept/02)	NAPS Mean NO_2 ppb (1997-2001)	NAPS Mean NO_2 ppb (Sept/01)	Ogawa Mean NO_2 (Sept/02)	% Difference LUR-NAPS (1997-2001)	% Difference LUR-NAPS (Sept/01)	% Difference LUR-Ogawa (Sept/02)
60410	26.9	24.0	21.6	28.2	***12.1***	***24.5***	***-4.6***
60413	25.3	24.3	21.8	20.1	***4.1***	***16.1***	***25.9***
60403	36.9	28.9	26.4	38.0	***27.7***	***39.8***	***-2.9***

Table 3: Comparison of the land use regression model and Ogawa sampled values to temporally matched government monitoring data.

Station Location	LUR predicted NO_2 ppb (Sept/02)	Ogawa Mean NO_2 (Sept/02)	MOE Mean NO_2 ppb (Sept/02)	% Difference LUR-MOE (Sept/02)	% Difference Ogawa-MOE (Sept/02)
60410	26.9	28.2	23.4	***13.0***	***17.0***
60413	25.3	20.1	19.9	***21.3***	***1.0***
60403	36.8	38.0	28.5	***22.6***	***25.0***

Mapping the Model

Kriging analysis was initially used to explore the overall trends present in the sampling data. Figure 4 shows a kriging surface generated with a spherical model of NO_2 across the city using the 95 data points. The downtown area of the city appears to have the highest levels of NO_2. This was also the area of the one monitoring location that was located adjacent to an active construction site. Using this technique it was feasible to visualize possible outliers in the data. Further, the area to the east of the downtown core, along Lake Ontario, appears to have relatively low measures of NO_2.

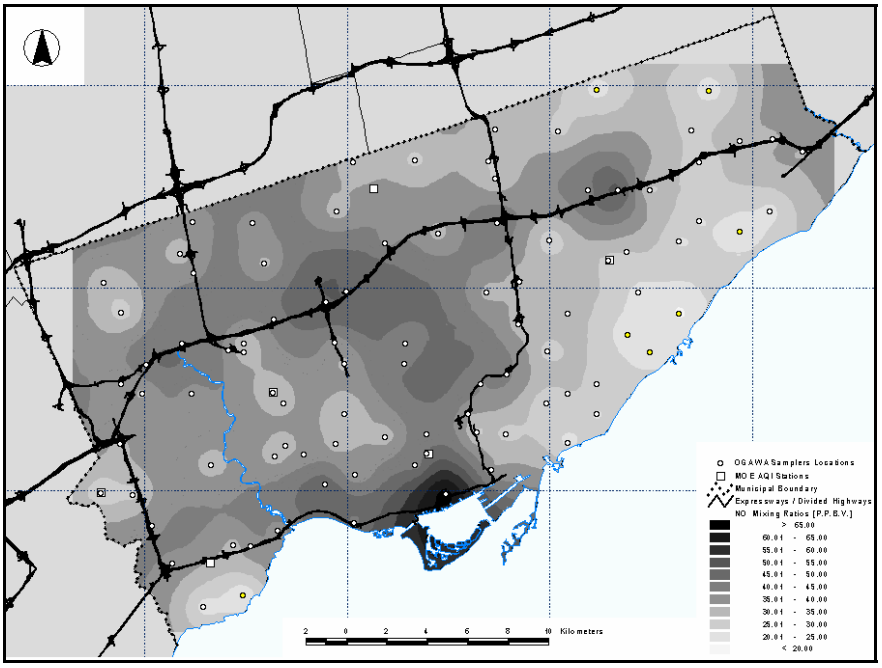


Figure 4: Kriged surface generated with a spherical model of NO_2 across the city using the 95 data points.

After deriving the parsimonious operational model, the coefficients were used to map a predicted pollution surface for exposure assignment in future health studies. Individual raster surfaces were created for each of the seven independent variables included in the model (listed in Table 1), which were then summed together with the Raster Calculator in ArcGIS 8 to generate the overall predicted surface.

The land use regression model created a predictive pollution surface with the expected characteristics (see Figure 5). Specifically, areas in proximity to expressways and in the downtown core appeared to have higher levels of NO_2, while areas with less development in the northeast of the city exhibit lower levels. Although the overall patterns remain similar, this map shows more detailed spatial variation than the kriging map.

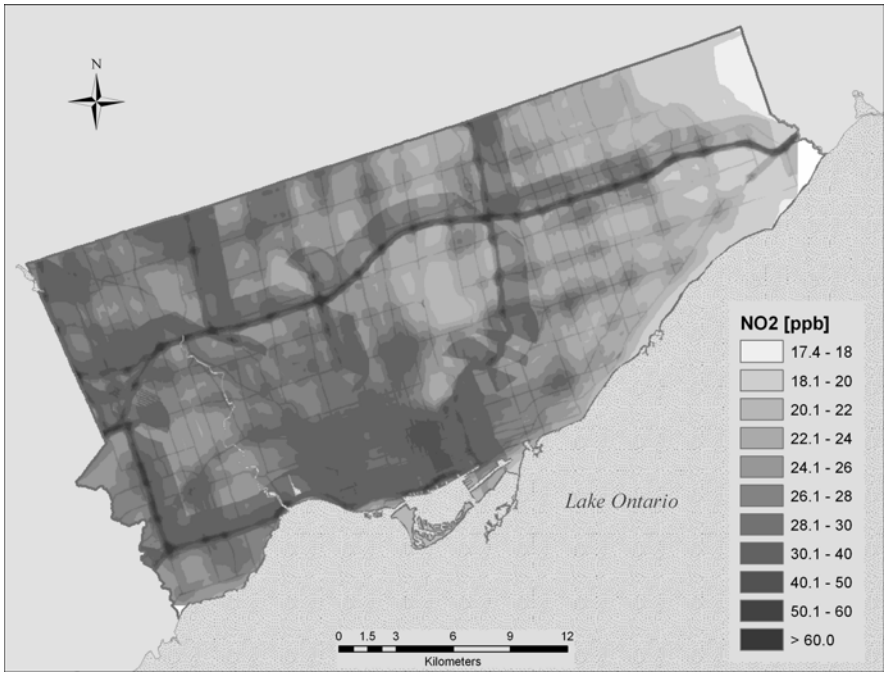


Figure 5: Operational land use regression predicted surface for Toronto.

DISCUSSION AND CONCLUSION

In this paper the determinants of ambient NO_2 throughout the City of Toronto, Canada, have been modeled. The results suggest that a stable predictive surface can be derived for this North American city using the LUR method. The difference in predictability when compared to some European models possibly arises from the variations in land use between European and North American cities. Further cross validation in other North American locations will lend insight on why these differences exist. As mentioned earlier, postwar sprawl in the North American city has five important characteristics that may contribute to the difference between the European and North American results presented here: (1) deconcentrated development (i.e., lower population density than in earlier periods); (2) decentralized, meaning more new development occurs in suburban rather than central areas; (3) homogenous with very little mix in land use types; (4) large scale meaning extensive subdivisions, industrial parks, and commercial centers; and (5) segregated land use types that create a need for travel between residential and commercial (Fowler, 1992; Ewing et al., 2002). Taken together, these characteristics increase the demand for travel by automobiles, and they probably create an exposure surface with more spatial variability than in European urban areas, which seem to have higher but less variable concentrations overall. Predicting a more complex exposure surface may have contributed to the lower R^2 values observed in this study.

In addition, some of the European studies exposed the monitors for longer periods or for multiple seasons. The longer temporal run of monitoring data may have stabilized the monitor readings, contributing to the proportion of explained variance. This paper is reporting the first two-week sampling period in a larger study. After subsequent seasonal sampling is conducted, the data will be pooled together in line with European studies with the expectation that similar predictive capacity will be achieved, as a result of a sample more representative of the overall urban pollution variability.

The sensitivity of the models to variable specification also requires attention in future research. Population density variables, for example, could produce a wide range of results, depending on the scale of the data inputs, the size of the buffer, and the method for operationalizing the variables. Again further empirical work in North America will be required to assess the adequacy of different variable specifications.

This model has been the first LUR to incorporate influence of wind direction on predicted pollution concentration, but much more must be done to include impact of other meteorological variables such as wind speed, temperature, humidity, and atmospheric stability. The way forward for improving the land use regression techniques appears to be the development of some hybrid model that combines the positive features of this method, particularly the local-scale land use information, with more sophisticated emission transport models such as MM5 coupled with emission models such as Mobile 5.

Good agreement was found between the 4 samples at each location (i.e., two double-sided Ogawa monitors). For future seasonal monitoring, only one Ogawa sampler will be used at each location, with 25% of locations containing duplicate Ogawa samplers. These duplicates will assure a means to assess data quality. Although the cross validation produced reasonable results for the same period of deployment with the same monitors, larger differences were found when comparing our predictions to temporally matched and five-year average concentrations from MOE and Environment Canada NAPS sites. The difference between the model predictions and the government monitors is difficult to assess with the limited number of available government stations to conduct cross-validations. Larger variations between Ogawa monitors and the government monitors may have occurred due to the small number of locations available from MOE and Environment Canada for comparison, the difference in monitoring technology, or the temporal variability in emission and meteorological variables. Regardless of the specific reason for this discontinuity in the results, this finding suggests longer monitoring periods covering all seasons may be necessary to capture the intra-urban variability in traffic pollution.

In future research, additional meteorological and point sources emission variables will be incorporated into the model. Additional monitoring will be conducted during different seasons with co-located fine particle monitoring stations to assess the composition and predictability of these pollutants. In a separate study, these NO_2 exposure models have linked to a large cohort of patients from respiratory clinics across Toronto to assess associations at the intra-urban scale to advance previous research using less sophisticated exposure metrics (see Finkelstein et al., 2003 for related studies). Along with other collaborators, similar monitoring networks have also been implemented in other Canadian cities (i.e., Hamilton, Montreal and Vancouver). Pooled estimates from these cities may be used to derive exposure assessments for linkage with the National Population Health Survey. These estimates will in turn allow for assessment of between and within city variation in air pollution exposures and health effects for the Canadian population.

In developing the national-level models, a trade-off inherent in LUR method will become more pronounced: the more the model is refined to specific conditions in one locale, the less transferable and operational it becomes. For example, inclusion of wind direction would require sophisticated meteorological models for each new area. The same could be said of including potential important microenvironmental variables such as street canyons. It would be virtually impossible to document each one of these canyons for extrapolating across many locations within a city, yet alone between them. One solution may arise in the form of remotely sensed data that could be used to assess characteristics such as street canyons, and this would allow for incorporation of these data into the computing environment.

The LUR maps showing predicted surfaces appear to capture small-area variation in NO_2 concentrations more effectively than geostatistical alternatives such as kriging. These small-area variations are probably more important to the exposure experience of subjects in a given health study and, as a result, may detect health effects that would have gone unnoticed with government monitoring data or even kriging estimates. For this reason, the LUR appears worthy of further study in a North American context.

Acknowledgements

Funded by Health Canada and the Canadian Institutes of Health Research. We thank Chris Giovis, Michael Heffernan, Natalia Restrepo, Kelly Williams, and Shuhua Yi, McMaster University, for assistance with deploying the monitors and preparing data. In addition, we thank Dr. Christopher Morgan and his colleagues at the City of Toronto for approval to deploy the monitors on City property and for the traffic data. The Ministry of Transportation for Ontario gave timely approval for entrance onto their highways. We also acknowledge helpful comments from Dr. Mark Goldberg, McGill University, on the cross-validation methods. Additionally we acknowledge funding from EPA grant RD83186101, NIEHS grants 5P01 ES11627, 5P01 ES09581, and the Southern California Environmental Health Sciences Center funded by NIEHS grant 5P30 ES07048. Any remaining errors or omissions are the responsibility of the authors.

REFERENCES

Brauer, M., Hoek, G., van Vliet, P., Meliefste, K., Fischer, P., Gehring, U., Heinrich, J., Cyrys, J., Bellander, T., Lewne, M., and Brunekreef, B. 2003. Estimating long-term average particulate air pollution concentrations: application of traffic indicators and geographic information systems. *Epidemiology* 14:228-239.

Briggs, D., Collins, S., Elliott, P., Fischer, P., Kingham, S., and Lebret, E. 1997. Mapping urban air pollution GIS: a regression-based approach. *Int. J. Geogr. Inf. Syst.* 11:699-718.

Briggs, D.J., de Hoogh, C., Gulliver, J., Wills, J., Elliott, P., Kingham, S., and Smallbone, K. 2000. A regression-based method for mapping traffic-related air pollution: application and testing in four contrasting urban environments. *Sci. Total Environ.* 253:151-167.

Delucchi, MA. 2000. Environmental externalities of motor-vehicle use in the US. *J. Trans. Econ. Pol.* 34:135-168.

Environmental Systems Research Institute (ESRI) 1999-2002. Deterministic methods for spatial interpolation. *ArcGIS Desktop Help*, ESRI, Redlands, CA.

Ewing, R., Pendall, R., and Chen, D. 2002. Measuring Sprawl and Its Impacts. *Smart Growth America*. www.smartgrowthamerica.org/sprawlindex/MeasuringSprawlTechnical.pdf. Accessed: October 6, 2004.

Finkelstein, M., Jerrett, M., DeLuca, P., Finkelstein, N., Verma, D.K., Chapman K., and Sears, M.R. 2003 A cohort study of income, air pollution and mortality. *Can. Med. Assoc. J.* 169:397-402.

Fowler, E.P. 1992. *Building Cities That Work*. Montreal and Kingston: McGill-Queen's University Press.

Getis, A., and Getis J. 1995. *The United States and Canada: The Land and the People*. Dubuque, IA, USA: Wm. C. Brown Publishers.

Gilbert, N.L., Woodhouse, S., Stieb, D.M., and Brook, J.R. 2003. Ambient nitrogen dioxide and distance from a major highway. *Sci. Total Environ.* 312:43-46.

Goodin, W.R., McRae, G.J., and Seinfeld, J.H. 1980. An objective analysis technique for constructing three-dimensional urban-scale wind fields. *J. Appl. Meteor.* 19:98-108.

Hamilton, L. 1992. *Regression with Graphics*. Belmont, CA: Duxbury Press.

Hewitt, C.N. 1991. Spatial variations in nitrogen dioxide concentrations in an urban area. *Atmos. Environ.* 25B:429-434.

Hoek, G., Brunekreef, B., Goldbohm, S., Fischer, P., and van den Brandt, P.A. 2002. Association between mortality and indicators of traffic-related air pollution in the Netherlands: a cohort study. *Lancet* 360:1203-1209.

Hough, M. 1995. *Cities and Natural Process*. London: Routledge.

Hubbe, J.M., Doran, J.C., Liljegren, J.C., and Shaw, W.J. 1997. Observations of spatial variation of boundary layer structure over the Southern Great Plains cloud and radiation testbed. *J. Appl. Meteorol.* 66:1221-1231.

Kanaroglou, P., Jerrett, M., Morrison, J., Beckerman, B., Arain, A., Gilbert, N., and Brook, J. 2003. Establishing an air pollution monitoring network for intra-urban population exposure assessment: a location-allocation approach. *In 12th Symposium (June 2003) Proceedings of the Transport and Air Pollution Conference,* R. Joumard ed., 1:27-34.

Lebret, E., Briggs, D., Van Reeuwijk, H., Fischer, P., Smallbone, K., and Harssema, H. 2000. Small area variations in ambient NO_2 concentrations in four European areas. *Atmos. Environ.* 34:177-185.

Lynn, P.P. 1975. Rainfall interpolation using multiquadratic surfaces. *Comp. Appl. Natur. Social Sci.* 2:321-334.

Myers, D.E. 1994. Spatial interpolation: an overview. *Geoderma.* 62:17-28.

Nieuwenhuijsen, M.J. 2000. Personal exposure monitoring in environmental epidemiology. In *Spatial Epidemiology: Methods and Applications*. P. Elliott, J. Wakefield, N. Best, and D. Briggs, eds. Oxford: Oxford University Press.

Newman, P., and Kenworthy, J. 1989. *Cities and automobile dependence: A sourcebook*. Brookfield, VT: Gower Technical.

Nuss, W.A., and Titley, D.W. 1994. Use of multiquadric interpolation for meteorological objective analysis. *Mon. Wea. Rev.* 122:1611-1631.

Shaw, E.M., and Lynn, P.P. 1972. Areal rainfall evaluation using two surface-fitting techniques. *Bull. Int. Assoc. Sci. Hydrol.* 17:419-433.

Sirayanone, S. 1988. Comparative Studies of Kriging, Multiquadric-Biharmonic, and Other Methods for Solving Mineral Resource Problem. Unpubl. Doctoral Dissertation, Iowa State Univ., Ames, IA, p. 355.

Statistics Canada. 2001. 2001 Census of Canada. Available online: http://www12.statcan.ca/english/census01/release/index.cfm.

Wentz, E., Gober, P., Balling Jr., R.C., and Day, T.A. 2002. Spatial Patterns and Determinants of Winter Atmospheric Carbon Dioxide Concentrations in an Urban Environment. *Annals Assoc. Amer. Geo.* 92:15–28.

APPENDIX 1:
BIVARIATE REGRESSION ANALYSIS ON 83 VARIABLES

	Variable	Units	Description	R square	t score
1	Dist_Exp	Km	Distance to nearest RD1	.277	-5.966
2	Basket	Binary	Identifier of samplers located at major 'basket weave' intersections	.045	2.101
3	TruckVol500	Count	Sum of 24hour truck counts on selected RD1s within 500m	.247	4.955
4	HwyFlow1000	Count	Sum of 24hour traffic counts on RD1s within 1000m	.242	5.446
5	RdFlow300_24	Count	Sum of 24hour traffic counts on roads within 300m	.061	2.450
6	RdFlow500_24	Count	Sum of 24hour traffic counts on roads within 500m	.157	4.169
7	AmFlow500	Count	Sum of AM peak traffic counts on roads within 500m	.143	3.935
8	PmFlow500	Count	Sum of PM peak traffic counts on roads within 500m	.137	3.846
9	AmDiv24	%	Am peak counts divided by 24 hour totals (within 500m)	.001	-.335
10	Rd_Density	Ha	Total area of RD1, RD2 and RD3 within 300m	.347	7.032
11	RD1_50	Km	Length of road within 50m	.233	5.315
12	RD2_50	Km		.063	2.494
13	RD3_50	Km		.073	-2.706
14	RD1_50200	Km	Length of road between 50 & 200m (annulus buffer)	.311	6.479
15	RD2_50200	Km		.045	2.086
16	RD3_50200	Km		.115	-3.482
17	RD1_200	Km	Length of road within 200m	.314	6.524
18	RD2_200	Km		.055	2.326
19	RD3_200	Km		.120	-3.569
20	RD1_300	Km	Length of road within 300m	.301	6.325
21	RD2_300	Km		.082	2.885
22	RD3_300	Km		.106	-3.315
23	RD1_300500	Km	Length of road between 300 & 500m (annulus buffer)	.154	4.110
24	RD2_300500	Km		.106	3.326
25	RD3_300500	Km		.047	-2.150
26	RD1_500	Km	Length of road within 500m	.245	5.489
27	RD2_500	Km		.128	3.702
28	RD3_500	Km		.074	-2.727
29	RD1_750	Km	Length of road within 750m	.245	5.487
30	ELEV	m	Elevation at sampling site	.008	-.852
31	X	UTM	Geographic location (east/west)	.114	-3.451
32	Y	UTM	Geographic location (north/south)	.047	-2.146
33	EADens	Count	Enumeration Area population density (polygon thematic map)	.003	-.525
34	EA750	Count	Enumeration Area population density kernel estimate, 750m	.014	1.135
35	EA1000	Count		.022	1.444
36	EA1250	Count		.030	1.707
37	EA1500	Count		.038	1.927
38	EA2000	Count		.045	2.095
39	EAs2000	Count	Simple density estimate	.043	2.053
40	EA2500	Count		.045	2.101
41	CTDens	Count	Census Tract population density (polygon thematic map)	.001	.358
42	CT750	Count	Census Tract population density kernel estimate, 750m	.016	-1.219
43	CT1000	Count		.012	-1.048
44	CT1250	Count		.006	-.773
45	CT1500	Count		.003	-.490
46	CT2000	Count		-.010	.160
47	CTs2000	Count	Simple density estimate	.011	1.014
48	CT2500	Count		-.007	.602
49	PC2500	Count	Postal Code population density kernel estimate, 2500m	.006	.726
50	PC5000	Count		.028	1.638
51	DC1000	Count	Enumeration Area density of dwellings kernel estimate, 1000m	.013	1.104
52	DC2000	Count		.030	1.683
53	DC2500	Count		.030	1.706
54	DC5000	Count		.050	2.221
55	DwVal	Count	Enumeration Area average dwelling value (polygon thematic map)	.003	-.491
56	Dw1000	Count	Enumeration Area average dwelling value, kernel estimate, 1000m	.013	1.098
57	Dw1500	Count		.024	1.523
58	Dw2000	Count		.030	1.682
59	Dw2500	Count		.030	1.697

	Variable	Units	Description	R square	t score
60	Open300	Ha	Area of land use within 300m	.043	2.033
61	Res300	Ha		.058	-2.395
62	Comm300	Ha		.018	-1.288
63	Indust300	Ha		.028	1.628
64	Gov/Inst300	Ha		.000	-.029
65	Open400	Ha	Area of land use within 400m	.029	1.672
66	Res400	Ha		.065	-2.546
67	Comm400	Ha		.012	-1.057
68	Indust400	Ha		.031	1.721
69	Gov/Inst400	Ha		.001	.260
70	Open500	Ha	Area of land use within 500m	.019	1.340
71	Res500	Ha		.069	-2.625
72	Comm500	Ha		.003	-.533
73	Indust500	Ha		.031	1.725
72	Comm500	Ha		.003	-.533
73	Indust500	Ha		.031	1.725
74	Gov/Inst500	Ha		.002	.447
75	Open750	Ha	Area of land use within 750m	.003	.486
76	Res750	Ha		.071	-2.663
77	Com750	Ha		.017	1.275
78	Indust750	Ha		.033	1.781
79	Gov/Inst750	Ha		.009	.927
80	D_WIND	BOOL	Down wind or not	.030	1.689
81	D_WIND5	BOOL	Down wind or not within 500m	.193	4.709
82	D_WIND10	BOOL	Down wind or not within 1000m	.193	4.720
83	D_WIND15	BOOL	Down wind or not within 1500m	.193	4.720
84	TRAF300	Count/km^2	Density estimate of 24 hour traffic count within 300m	.053	2.287
85	TRAF500	Count/km^2	Density estimate of 24 hour traffic count within 500m	.062	2.473

APPENDIX 2:
CROSS-VALIDATION SHOWING ABSOLUTE AND RELATIVE DIFFERENCE BETWEEN OBSERVED AND MODELED VALUES FOR 30 RANDOMLY SELECTED CASES

Cross Validation 1				Cross Validation 2				Cross Validation 3			
Final Model	Cross Predicted 1	Difference	% Difference	Final Model	Cross Predicted 2	Difference	% Difference	Final Model	Cross Predicted 3	Difference	% Difference
35.6	34.6	-1.0	-2.8	31.5	29.7	-1.8	-6.2	35.6	36.8	1.2	3.3
41.1	42.1	1.0	2.5	25.3	25.0	-0.3	-1.4	41.1	42.1	1.1	2.5
41.8	38.1	-3.7	-9.7	48.1	47.3	-0.8	-1.6	41.8	41.7	-0.1	-0.3
30.1	31.0	0.9	2.7	29.3	29.8	0.5	1.5	30.1	31.4	1.3	4.1
32.7	36.4	3.7	10.1	31.7	30.1	-1.5	-5.1	32.7	35.1	2.4	6.8
39.9	45.6	5.6	12.3	22.7	23.1	0.4	1.8	39.9	41.7	1.7	4.1
25.5	25.0	-0.6	-2.3	42.7	44.6	1.8	4.1	25.5	25.5	0.0	0.0
29.5	29.5	0.1	0.2	19.9	20.2	0.3	1.6	29.5	29.4	-0.1	-0.4
31.1	31.5	0.5	1.4	28.5	28.2	-0.3	-1.1	31.1	29.9	-1.2	-4.0
45.0	59.0	14.0	23.7	39.6	39.1	-0.6	-1.4	45.0	53.8	8.8	16.3
41.4	48.3	6.9	14.3	21.4	22.1	0.7	3.3	41.4	45.0	3.6	8.0
25.9	25.6	-0.3	-1.1	34.1	34.0	-0.1	-0.2	25.9	25.4	-0.5	-1.8
27.2	27.9	0.6	2.3	28.3	29.0	0.6	2.2	27.2	27.1	-0.1	-0.4
45.5	52.3	6.8	13.0	24.5	24.7	0.1	0.5	45.5	50.4	4.9	9.7
46.9	52.9	6.0	11.4	30.6	31.1	0.5	1.5	46.9	51.5	4.6	9.0
25.9	26.9	1.0	3.7	27.8	27.8	-0.1	-0.2	27.8	28.0	0.2	0.6
26.2	25.6	-0.7	-2.6	32.3	30.7	-1.6	-5.2	32.3	32.4	0.1	0.2
52.0	53.6	1.6	3.0	24.2	23.9	-0.4	-1.6	24.2	24.2	0.0	-0.1
47.7	49.6	2.0	4.0	25.0	24.7	-0.2	-0.9	25.0	25.3	0.3	1.4
26.2	25.6	-0.6	-2.3	34.8	33.7	-1.1	-3.3	34.8	36.2	1.3	3.6
42.8	43.8	1.0	2.3	29.4	29.8	0.4	1.4	29.4	29.6	0.2	0.6
33.3	33.6	0.3	0.8	44.3	46.3	2.1	4.4	44.3	46.5	2.2	4.7
39.9	39.7	-0.2	-0.5	21.3	22.0	0.7	3.4	21.3	21.5	0.2	1.2
24.1	23.9	-0.2	-0.8	39.0	38.1	-0.9	-2.2	39.0	41.7	2.7	6.4
25.4	27.9	2.5	8.8	24.0	24.3	0.3	1.4	24.0	25.6	1.6	6.4
31.9	32.0	0.1	0.3	40.4	41.7	1.3	3.0	40.4	42.9	2.5	5.8
35.6	35.5	-0.1	-0.4	22.6	22.9	0.3	1.3	22.6	22.7	0.1	0.4
26.9	29.4	2.5	8.5	27.5	27.4	-0.1	-0.2	27.5	27.5	0.1	0.3
31.3	36.0	4.7	13.0	32.1	32.6	0.5	1.4	32.1	32.7	0.6	1.9
	Average =	**1.9**	**4.0**		**Average =**	**0.0**	**0.1**		**Average=**	**1.4**	**3.1**

TWO YEAR FINE AND ULTRAFINE PARTICLES MEASUREMENTS IN ROME, ITALY

Achille Marconi, Giorgio Cattani, Mariacarmela Cusano, Marcello Ferdinandi, Marco Inglessis, Giuseppe Viviano, Gaetano Settimo,[1] Francesco Forastiere[2]

ABSTRACT

Long-term aerosol measurements have been conducted at two sites in Rome, Italy, April 2001 through March 2003, in a traffic-oriented site, and an urban background site, close to the city center. The main objective was to establish validated and consistent data sets of particle number concentrations (PNC) in Rome to be used for epidemiological analyses of cardiovascular health effects. Particle number concentrations were measured by a condensation particle counter (CPC 3022A, TSI). Other pollutants, (PM_{10}, $PM_{2.5}$, CO, NO_2, NO, NOx, O_3) were simultaneously measured at the traffic-oriented site. During the study period, the mean (standard deviation) 24-hr PNC were $4.69x10^4$ ($1.99x10^4$) cm^{-3} and $2.46x10^4$ ($1.10x10^4$) cm^{-3} respectively at the traffic-oriented site and at the urban background site. Mean (standard deviation) 24-hr mass concentration of $PM_{2.5}$ was 23.1 (11.9) $\mu g\ m^{-3}$, while for PM_{10} it was 41.3 (17.9) $\mu g\ m^{-3}$. Higher values for all the pollutants, except ozone, were recorded during the winter period in comparison with the summer period, and a higher variability of the results was also observed during cold months. The comparison between the daily PNC measured at the two sites showed a good correlation (r = 0.74). CO (r = 0.77), NO (r = 0.82) and NO_X (r = 0.83) were all highly correlated with PNC (simultaneous obs. n. 576). Diurnal and seasonal pattern of PNC can be attributed to the combined effect of motor vehicle emissions and meteorological conditions.

[1] Italian National Institute of Health, Rome, Italy.
[2] Azienda Sanitaria Locale Roma E, Rome Italy

INTRODUCTION

Epidemiological research during the last decade has indicated that exposure to air pollution at the levels presently measured in European urban environments is associated with an increase in mortality and with a variety of health conditions, including emergency room visits and hospital admissions for respiratory and cardiovascular diseases. Particulate matter (PM) appears to be the air pollutant most consistently associated with adverse health outcomes (Dockery and Pope, 1994; Schwartz et al., 1996; Milligan et al., 1998; Ostro and Chestnut, 1998; Dockery, 2001; Pope et al., 2002; WHO, 2002). With respect to dimension, urban particles are broken down into three groups: ultrafine particles, accumulation mode particles (which together form the fine particle mode) and coarse mode particles. Ultrafine particles contribute very little to the overall mass, but are very high in number, which in episodic events can reach several hundred thousand/cm^3 in urban air (Oberdörster, 2001).

Regarding the sources, ultrafine particles in polluted urban environments consist of three main types:

- primary particles originating from road traffic: these particles are directly discharged during combustion processes and are believed to contribute to the majority of the total number of particles in city centers.
- secondary particles (such as ammonium sulfate and nitrate) originating in the atmosphere from oxidation of precursor pollutants (SO_2, NO_x) followed by neutralisation with gaseous ammonia as well as organic aerosol originating from photochemical induced oxidation of some volatile organics compounds (NMVOCs).
- inflow particles: these particles are known to result from long-range transport as well as from regional energy production and industrial activities (CAFE, 2004).

Motor vehicle emissions usually constitute the most significant source of ultrafine particles in an urban environment. The number of ultrafine particles (0.01 to 0.1 μm, normally expressed as Particle Number Concentration - number per cubic centimetre of atmospheric air) is hypothesised to be of particular concern (Seaton et al., 1995; Oberdörster and Utell, 2002).

The main objective of this study was to establish validated and consistent data sets of particle number concentrations in Rome. These data sets were collected and used in the framework of the research project funded by the European Union: “Health effects of air pollution on susceptible sub-population – traditional air pollutants, ultrafine particles and myocardial infraction: database and health assessment, HEAPSS”. The main objective of this project is to quantify the risk of hospitalisation and death due to air pollution, in particular airborne ultrafine particles, in individuals with coronary heart disease in five European cities. This paper reports the data available after the first two years of continuous monitoring in Rome. The results regarding environmental monitoring (Aalto et al., 2004) and health effects (Lanki et al., 2004) in the five cities have been reported elsewhere.

EXPERIMENTAL METHODS

The data reported in this study are relative to the period from 4/2001 to 3/2003. Rome is a large metropolitan area, with about 3.000.000 inhabitants. In the metropolitan area very high traffic concentrations are reported, in many roads, especially from 7 to 9 a.m. and from 6 to 8 p.m. The ratio of diesel to gasoline cars circulating in the town is about 1:4. The residential heating season usually lasts five months; fuels are mostly natural gas and heating oil. In the last few years SO_2 and lead concentrations decreased significantly in ambient air due to the decreasing content of sulphur compounds and lead tetralchile in fuels.

Sampling sites

In our study we have used data from two separate measurement locations, in the urban area of Rome, Italy (Figure 1).

Primary site

The primary site is located 2 km east of the city center on the front yard of the Italian National Institute of Health (INIH). Inlets for particle measurements were at about 8 m from the curb and about 20 m from the street, Viale Regina Elena, and approximately 3 m from the ground. A flow of 25000 cars/day was estimated, constant throughout the year except for August. Traffic intensity was presumed to be roughly intermediate between the center and suburban areas. The area is not subject to industrial emissions. This site could be considered as a traffic-oriented site.

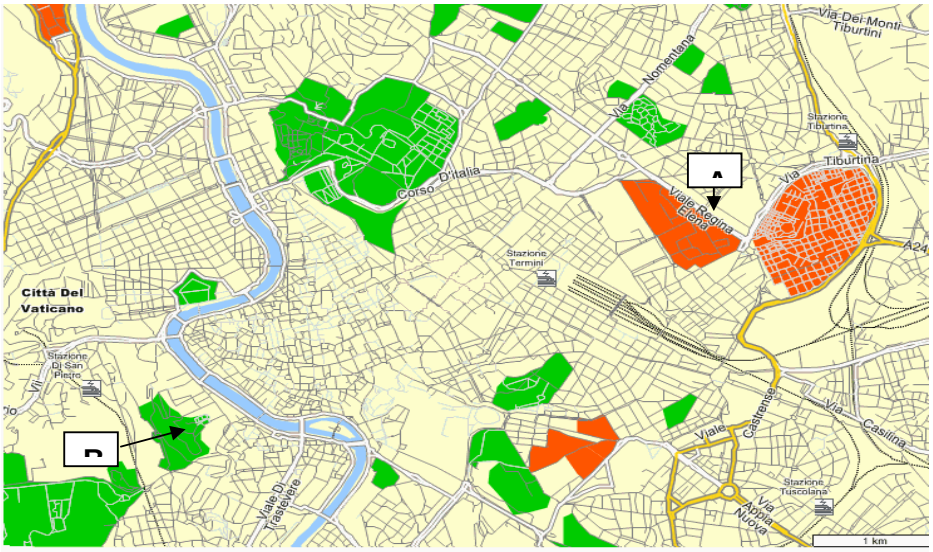


Figure 1. Measurement locations. A: INIH - primary site, traffic-oriented; B: Botanical garden - secondary site, urban background.

Secondary site

The secondary site is located in a park (botanical garden) situated in the area surrounding the center of town. From the sampling site, at the east side there is a little hill (Gianicolo) characterized by a large variety of plants and trees. At the south side there is the famous district of Trastevere (about 400 meter from the sampling point), which is a limited traffic area (only for residents). The closest traffic street is located about 400 meters from the sampling point. So we can consider this an urban background site.

For the purpose of this study, the following air samplers were used:

Two low volume samplers, model SKYPOST PM (TCR Tecora - Italy); one operating at 1 m^3/h, equipped with an omnidirectional aerosol inlet designated as reference for PM_{10} by the United States Environmental Protection Agency (US-EPA) and a WINS impactor to separate the particles into the "fine" size fraction ($PM_{2.5}$). The second operating at 2.3 m^3/h, equipped with an omnidirectional aerosol inlet designated as reference for the thoracic fraction (PM_{10}) according to CEN standard EN 12341 (1998).

PM was collected daily on 47-mm diameter glass fiber filter (Pall Corporation, USA). The use of this type of filters was considered a satisfactory compromise between advantages (reduced cost, low hygroscopicity, lower friability with respect to quartz filters) and disadvantages (the potential occurrence of artifacts, e.g. sulphates). Glass fiber filters have been recommended in the recent draft of the European reference method for $PM_{2.5}$ (CEN, 2004).

Each sixth day, polymethylpentane-ringed, 2.0-µm pore size, 47-mm diameter Teflon filters (Gelman, USA) were used for chemical characterization of collected particles (results not included in this study).

Although the PM samples were collected daily, the number of data (shown in Table 1 and 2) is lower than the data of other pollutants due to instrumental mechanical and electronic problems. Standard filter handling and weighing procedures were adopted in order to minimize errors.

Equilibration and weighing were carried out under controlled conditions. The filters were equilibrated pre- and post-sampling in an air-conditioned room under controlled temperature conditions (20°C ± 1°C) and relative humidity (50% ± 5 %). Conditioning time within the weighing environment was at least 24 hours. Weighing was performed inside the same room, using a Sartorius model M5P 000V001 electrobalance (readability: 0.001 mg).

Table 1. INIH site (traffic-oriented), April 2001 – March 2003: mean of the 24-h average concentrations and descriptive statistics of particle number concentrations, $PM_{2.5}$, PM_{10}, CO, NO, NO_x, NO_2 and O_3.

	Valid N	Mean	S.D.	Min	25th Perc	Median	75th Perc	Max
$PM_{2.5}$ ($\mu g\ m^{-3}$)	387	24.0	12.2	4.4	16.2	21.3	29.0	87.9
PM_{10} ($\mu g\ m^{-3}$)	389	42.0	18.7	6.7	30.2	38.6	48.5	124.9
$PM_{2.5}/PM_{10}$	314	0.58	0.13	0.21	0.49	0.58	0.67	0.99
CO ($mg\ m^{-3}$)	659	1.4	0.8	0.3	0.8	1.2	1.7	6.3
NO_X ($\mu g\ m^{-3}$)	661	86.8	54.7	14.4	49.6	70.7	108.5	344.9
NO ($\mu g\ m^{-3}$)	661	41.9	44.9	1.6	12.7	25.7	52.3	288.9
NO_2 ($\mu g\ m^{-3}$)	661	44.9	14.5	12.1	34.4	44.5	53.9	89.2
O_3 ($\mu g\ m^{-3}$)	688	32.8	18.1	2.9	17.3	32.9	46.8	86.2
Particles number (cm^{-3})	630	4.56E+04	2.47E+04	3.50E+03	2.76E+04	4.10E+04	5.80E+04	1.40E+05

Table 2. INIH (Primary site, traffic-oriented): averages, standard deviations and Coefficient of Variation for all the pollutants over different period of the 2 years of monitoring - 4/2001-3/2003 - (spring-summer period: April through September; autumn-winter period: October through March).

Period		$PM_{2.5}$ ($\mu g\ m^{-3}$)	PM_{10} ($\mu g\ m^{-3}$)	$PM_{2,5}/PM_{10}$	CO ($mg\ m^{-3}$)	NO_X ($\mu g\ m^{-3}$)	NO ($\mu g\ m^{-3}$)	NO_2 ($\mu g\ m^{-3}$)	O_3 ($\mu g\ m^{-3}$)	Particles (cm^{-3})
	N	**84**	**117**	**78**	**147**	**127**	**127**	**127**	**173**	**153**
4/01-9/01	mean	17.7	36.5	0.51	0.96	56.5	17.8	38.7	46.6	3.65E+04
spring-summer	S.D.	7.1	10.7	0.14	0.42	23.9	13.8	12.4	13.2	1.47E+04
	CV%	39.8	29.3	27.4	43.4	42.3	77.8	32.0	28.3	40.3
	N	**54**	**68**	**54**	**182**	**182**	**182**	**182**	**150**	**181**
10/01-3/02	mean	32.2	50.3	0.61	2.02	122.3	72.2	50.1	17.8	6.15E+04
autumn - winter	S.D.	18.5	24.2	0.12	0.93	60.0	53.7	11.1	10.3	2.69E+04
	CV%	57.5	48.2	19.9	46.3	49.1	74.4	22.2	58.0	43.7
	N	**143**	**89**	**81**	**183**	**178**	**178**	**178**	**183**	**158**
4/02-9/02	mean	20.9	37.5	0.57	0.92	52.8	16.7	36.2	44.9	3.04E+04
spring-summer	S.D.	6.8	12.0	0.09	0.26	18.1	9.0	11.0	13.0	1.38E+04
	CV%	32.4	32.1	16.0	28.4	34.3	53.9	30.4	29.0	45.6
	N	**106**	**115**	**101**	**147**	**174**	**174**	**174**	**182**	**138**
10/02-3/03	mean	29.1	46.1	0.63	1.56	108.0	55.1	52.9	19.7	5.40E+04
autumn - winter	S.D.	13.1	22.7	0.14	0.66	57.9	48.2	15.4	11.7	2.60E+04
	CV%	45.1	49.2	22.3	42.3	53.6	87.5	29.1	59.5	48.1
	N	**138**	**185**	**132**	**329**	**309**	**309**	**309**	**323**	**334**
4/01-3/02	mean	23.4	41.5	0.55	1.53	94.4	49.0	45.4	33.3	4.93E+04
first year	S.D.	14.6	18.2	0.14	0.91	57.9	49.6	12.9	18.7	2.51E+04
	CV%	62.3	43.8	25.6	59.3	61.4	101.2	28.4	56.1	50.9
	N	**249**	**204**	**182**	**330**	**352**	**352**	**352**	**365**	**296**
4/02 - 3/03	mean	24.4	42.4	0.60	1.20	80.1	35.7	44.4	32.4	4.14E+04
second year	S.D.	10.7	19.3	0.13	0.58	50.8	39.4	15.8	17.7	2.35E+04
	CV%	44.1	45.5	20.8	47.8	63.4	110.6	35.4	54.6	56.9

S.D: standard deviation; N: number of valid data; mean: mean of the 24-h daily averages over the period.

Each filter was weighed twice both before and after the sampling session (once after conditioning, and then again after 24 – 48 h).

The limit of detection, (LOD) was calculated on the basis of the estimate of the collected mass imprecision following the ISO standard (ISO 15767, 2003). The LOD was 25 μg (1.0 $\mu g/m^3$ for the $PM_{2.5}$ sampler operating at 1 m^3/h and 0,5 $\mu g/m^3$ for the PM_{10} sampler operating at 2.3 m^3/h).

As a quality control check, three field blank filters were weighed during each weighing session. If the mean calculated mass difference between pre- and post-sampling exceeded 50 μg, then the sample was discharged.

Particles number concentration (PNC) was measured by two TSI model 3022A condensation particle counters (CPC). This type of counter can monitor particles larger than 0.02 μm in diameter, but still have a 50 % counting efficiency at 0.007 μm (Agarwald and Sem, 1980; Sem, 2002). Sampling lines were stainless steel tubing 2.5 m long and with an inner diameter of 4 mm. Inlet reached one meter out from the outside wall of the container in which the instruments were placed.

The HEAPSS standard operating procedures (SOP), containing the measurement protocol of total particle number concentration in ambient air, was strictly followed (Aalto et al., 2004).

One CPC was factory calibrated and serviced after six months of continuous work. Factory service and calibration assured proper operating conditions and the correct readings of the instruments. During the campaign both instruments were compared against a reference instrument once a year and a new calibration for the instrument operating at the INIH site was made by the University of Helsinki, according to the quality assurance procedure of the HEAPPS project.

The CPC measurement program recorded the data at least once a minute. Concentrations of traditional air pollutants (hourly averages of CO, O_3, NO, NO_x, NO_2) from fixed monitors, installed in a monitoring station in operation for many years at the same site (INIH), were measured according to standard procedures already employed in several European studies of air pollution.

At the secondary site only PNC were measured starting from February 2002. The instrument went out of service in July 2002. After factory maintenance and re-calibration and several comparison trials of the two CPCs performed at the primary site, it was operating again in January 2003.

In the correlation analysis of the data obtained at the INIH site, the conventional Pearson correlation coefficient was used.

RESULTS

Descriptive statistics calculated from the whole set of 24-h average concentrations of PNC, $PM_{2.5}$, PM_{10}, CO, NO, NO_x, NO_2 and O_3, measured at the INIH site, are presented in Table 1. Averages, standard deviations and the Coefficient of Variation for all pollutants over different periods of the two years of monitoring - 2001-2003 - (spring-summer period: April through September; autumn-winter period: October through March) are presented in Table 2.

Mean values through the year were lower than the target value recommended by the EU Directive (1999/30/EC) for all the pollutants except for PM_{10}. No target limits values are still proposed by EU for $PM_{2.5}$ and PNC.

A slight pattern toward decreasing concentrations was found by comparing the data of the first year (4/01 - 3/02) and the second year (4/02 - 3/03) for all the pollutants except for PM_{10} and $PM_{2.5}$. This pattern was similar for CO and particle number concentration (-21.3 % and -16.1% respectively).

Higher daily mean values for all the pollutants, except ozone, were recorded during winter period. Focusing on PNC, the peak events were found during winter. The cleanest month was August, during which the main urban activities are substantially reduced (in Italy many people are on holiday during this period and most of the commercial/industrial activities are closed or strongly reduced). January was the most polluted month: the mean values throughout this month were 3 – 4 times higher than the average values during August.

Variation in the parameters measured can be examined mathematically by comparing the coefficients of variation (CV; standard deviation divided by the mean). In general, the variability was higher during winter. Even if $PM_{2.5}$, PM_{10} and PNC showed a significant variability (Table 2), the highest variability was associated

with the gaseous compounds (like NO), which reflects the uneven local emissions from the primary traffic sources at the site.

The $PM_{2.5}/PM_{10}$ ratios were higher during the winter season (0.61 and 0.63 for the first and the second year respectively), than during summer (0.51 and 0.57).

As shown in Table 3, CO (r = 0.77) NO (r = 0.82) and NO_X (r = 0.83) were all highly correlated with the particle number concentrations (simultaneous obs. n. 576). Daily $PM_{2.5}$ and PM_{10} levels were found to be poorly correlated with the daily PNC.

Table 3. INIH site – traffic-oriented: relationship (number of observations) between the different variables, expressed by the Pearson correlation coefficient.

Variable	$PM_{2.5}$	PM_{10}	CO	NO_X	NO	NO_2	O_3	Particles
$PM_{2.5}$	-							
PM_{10}	0.8511 (314)	-						
CO	0.6452 (343)	0.5729 (337)	-					
NO_X	0.7052 (361)	0.6269 (349)	0.8954 (612)	-				
NO	0.6710 (361)	0.6037 (349)	0.8970 (612)	0.9767 (661)	-			
NO_2	0.5550 (361)	0.4598 (349)	0.5852 (612)	0.7456 (661)	0.5851 (661)	-		
O_3	-0.4044 (382)	-0.3080 (380)	-0.7039 (626)	-0.7082 (629)	-0.6756 (629)	-0.5851 (629)	-	
Particles	0.5534 (327)	0.4976 (332)	0.7688 (577)	0.8336 (576)	0.8168 (576)	0.6234 (576)	-0.6341 (597)	-

Figure 2 shows a time series plot of the daily average of PNC. These data exhibit large temporal variability with occasional spikes that exceeds the mean by a factor 2 or more. The comparison between the two sites (traffic oriented, INIH, and Urban Background, Botanical Garden) is only possible for the PNC. The comparison of the results (Table 4) is limited to the 172 simultaneous 24-averages values available. Urban background values were about 50% lower with respect to the traffic-oriented site. Comparison between the PNC measured at the two sites showed a good correlation (r = 0.74 – Figure 3). These findings do not seem particularly affected by the different period of the year.

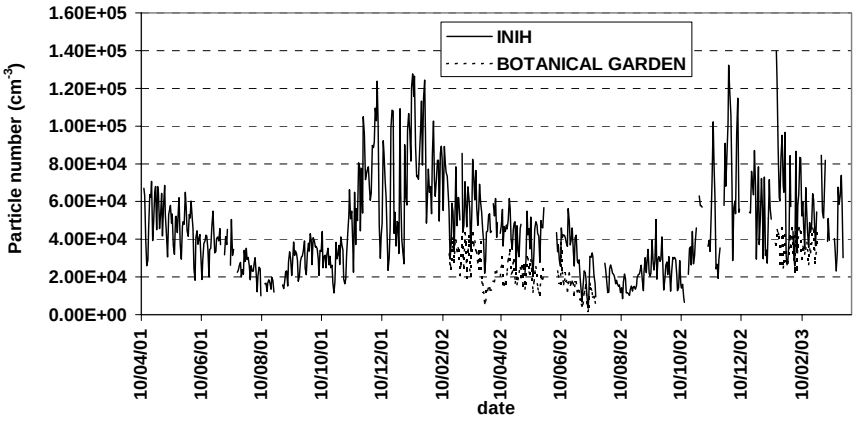


Figure 2. Time series of daily average total number concentration of particles measured at two sites (INIH – traffic-oriented and botanical garden - urban background).

Table 4. Mean of the 24-h average concentrations and descriptive statistics of particle number concentrations (cm^{-3}): comparison between the two sites. INIH (traffic-oriented) and Botanical garden (urban background) number of contemporary observations: 172.

	overall data		autumn - winter		spring - summer	
	INIH	**botanical garden**	**INIH**	**botanical garden**	**INIH**	**botanical garden**
Valid n	172	172	73	73	99	99
mean	4.69E+04	2.46E+04	5.89E+04	3.43E+04	3.80E+04	1.75E+04
ds	1.99E+04	1.10E+04	2.12E+04	7.17E+03	1.31E+04	7.14E+03
min	3.50E+03	1.73E+03	2.58E+04	1.93E+04	3.50E+03	1.73E+03
25^{th} perc	3.38E+04	1.64E+04	4.62E+04	2.77E+04	2.90E+04	1.29E+04
median	4.59E+04	2.36E+04	5.73E+04	3.49E+04	4.11E+04	1.68E+04
75^{th} perc	5.63E+04	3.39E+04	6.77E+04	3.92E+04	4.73E+04	2.23E+04
max	1.40E+05	4.83E+04	1.40E+05	4.83E+04	6.17E+04	3.83E+04

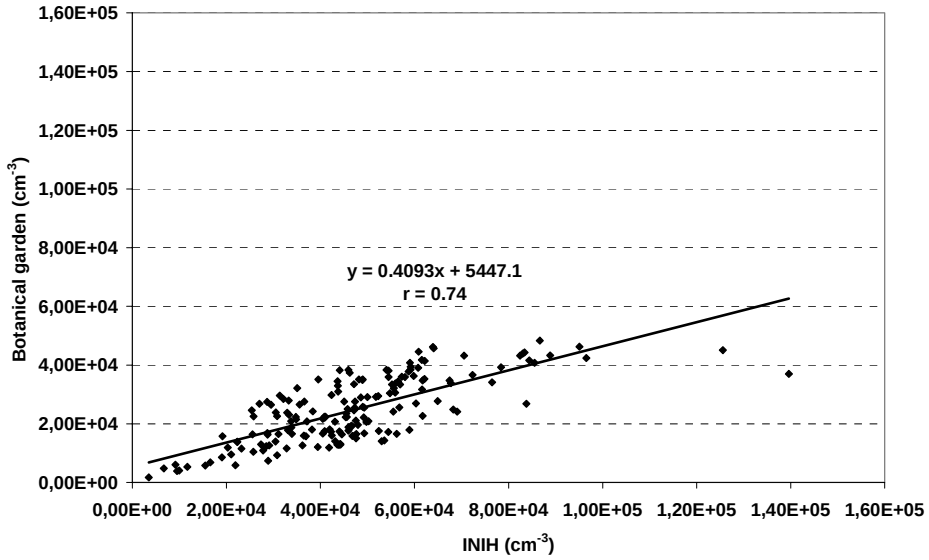


Figure 3. Correlation between daily average total concentration of particles measured at two sites (INIH – traffic- oriented and botanical garden - urban background). Number of pairs: 172. February 2002 through March 2003.

Figure 4 shows the diurnal variation of PNCs, from both sites. In both sites PNCs were very well correlated with the daily traffic flow, with highest mean values during rush hours between eight and nine a.m. and at around seven or eight p.m. Peak concentrations at the two sites followed the same pattern, but resulted higher at the traffic oriented site.

DISCUSSION

From the results reported in Tables 1 and 2 it can be argued that the general seasonal pattern observed could be due to the weaker atmospheric convective processes in winter. This pattern for PNC, as well as for other traffic-related air pollutants, was found more pronounced during the cool season than in summer also in other studies (Wiedensohler et al. 2002; Jeong et al., 2004; Gomiscek et al., 2004). During winter the morning and evening PNC peaks at rush hours can be considered the result of the motor vehicle emissions combined with a lower mixing layer height and lower ambient temperature, which favours nucleation mechanisms, at least for particles with dimensions up to some tenths of nanometers (McMurry et al., 2002). The morning peak still present in summer months, but with lower absolute values of PNC than in winter, might be related predominantly to

particles directly emitted by traffic and to the more favourable conditions of atmospheric dispersion. The high correlation found between gaseous compound as CO, NO and NO_x and PNC agrees essentially with other studies carried out in three European cities (Ruuskanen et al., 2001) and in Göteborg (Sweden) (Janhäll et al., 2004), in which it was shown that nitric oxide can be considered a better tracer of traffic related ultrafine particles, than traffic intensity itself. Moreover, in a recent study, significant correlation at street level was observed between CO, NO_x, and ultrafine particles in a street canyon, close to the central Copenhagen, indicating that the traffic is the major source of ultrafine particles in the air (Wahlin et al., 2001).

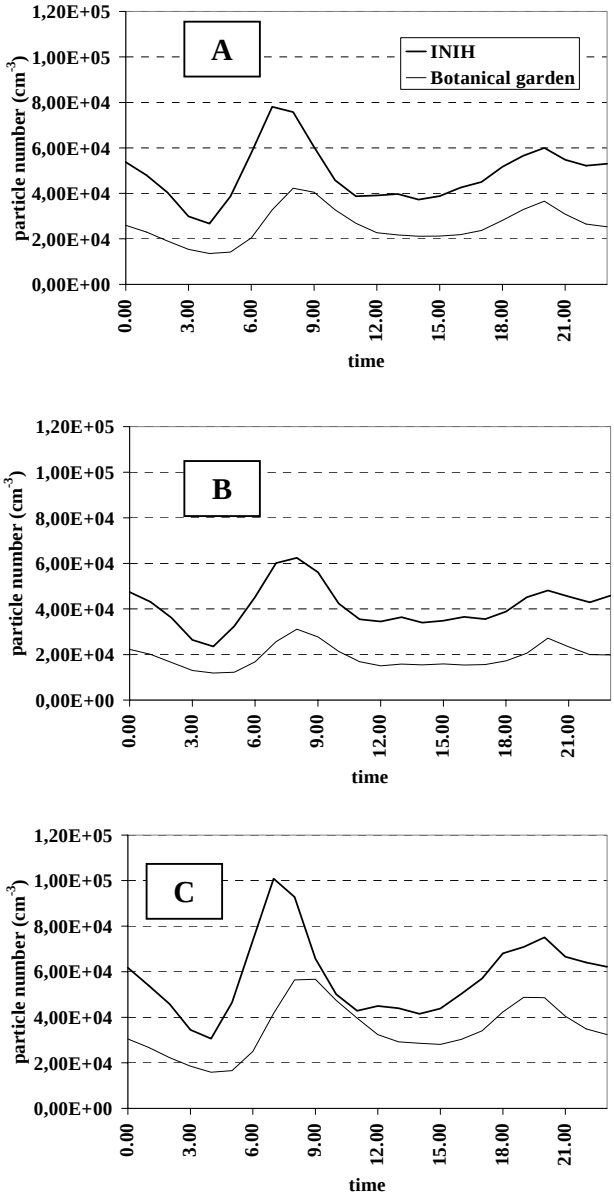


Figure 4. Diurnal variation of total number concentration of particles measured at two sites (INIH – traffic-oriented and botanical garden - urban background); A: all data available; B: spring-summer period; C: autumn-winter period.

These results underline the importance of the primary particles originating from road traffic. These particles are directly emitted during combustion processes and constitute a significant contribution to the total number of

particles in city centers (Ruuskanen et al., 2001; Jeong et al., 2004). This fact is highlighted by the strong correlation between PNC and the traffic flow during the day and the higher PNC at the traffic site. The significant correlation between PNC and CO and NO_x concentrations may suggest that nucleation occurs as the exhaust mixes with the cool ambient air (Shi and Harrison, 1999). In these circumstances the formation of ultrafine particles could be attributed more to the direct emission than to the photochemical gas-to-particle conversion (Jeong et al., 2004).

The diurnal behaviour of the PNC with the two peaks at rush hours appeared to be caused mainly by the traffic variability during the day and by the height of the vertical mixing. The lower evening peak might be related to the increase of the height of the boundary layer later in the day or to the diurnal pattern of the relative amounts of petrol and diesel vehicles (Aalto et al., 2004).

The behaviour of the PNC at the urban background site follows a similar pattern, with the only difference of the lower absolute values of the concentrations.

The PNC difference between the two sites is probably due to the distance from emission sources, but the influence of the traffic is still significant, even at distances of some hundreds of meters from streets. The reason for the average decrease of about 50% at the urban background site, could be explained by the findings of the measurements carried out in Los Angeles (Zhu et al., 2002) and Copenhagen (Ketzel and Berkowitcz, 2004). In the first study the relative concentrations of CO, black carbon BC, and particle number tracked each other well as distance from the freeway increased. PNC (6-220 nm) decreased exponentially with downwind distance from the freeway. It was suggested that both atmospheric dispersion and coagulation contribute to the rapid decrease in particle number concentration and change in particle size distribution as the distance from the freeway increased. In the second study more emphasis is given to the dilution process with little alteration of the size distribution.

The values of the $PM_{2.5}/PM_{10}$ ratios obtained in this study confirm the results shown in several previous studies (Harrison et al., 1997; D'Innocenzio et al., 1998; Janssen et al., 1999; Marconi et al., 2000; Gomiscek et al., 2004). The higher value of this ratio in winter might reflect the increasing occurrence of particles from both nucleation and accumulation mechanisms combined with poor atmospheric mixing conditions. The decrease of the $PM_{2.5}/PM_{10}$ ratio during summer could be due to the contribution of the events of long range transport from the Sahara desert (more frequent in spring-summer), and to the resuspension of particles belonging to the coarse fraction, which is more pronounced during dry weather and better mixing conditions, normally occurring during summer months.

The poor correlation between PNC (where the particles in the ultrafine range, less than 0,1 μm, are predominant) and mass concentration of the respirable fraction of particulate matter, $PM_{2.5}$ (where a dominant contribution to the mass is due to the larger particle fractions) is consistent with the findings of a number of recent studies (Laakso et al., 2003; Stanier et al., 2004; Jeong et al., 2004).

CONCLUSION

PNC, $PM_{2.5}$, PM_{10}, CO, NO_x, NO_2 and O_3 were continuously monitored in a traffic-related site and, simultaneously, PNC were measured at an urban background site in Rome. A clear seasonal and daily pattern was found, with higher 24-mean values during winter and peak concentrations during rush hours. The occurrence of the morning events tended to follow the typical pattern of mixing depths in winter, the season with the lowest mixing depth. These results underline the importance of the primary particles originating from road traffic. Although the absolute values found in the urban background site were lower (about 50%) than the values found in the traffic related site, they show a good correlation, suggesting dispersion of these particles also at relatively long distances from the primary sources (hundreds of meters). The magnitude of these findings needs more attention, because the selection of the sampling site could influence the classification of the pollution level (Aalto et al., 2004) and its use in epidemiological studies. On the other hand, a statistically significant association between PNC concentration and hospital admissions for myocardial infarction has been observed in Rome (Lanki et al, 2004). The modest correlation between PNC and currently measured mass-based aerosol indicators $PM_{2.5}$ and PM_{10} suggests that ultrafine number concentration information from these indicators could not be inferred, so independent measurement systems are required if the relationships between health outcomes and ultrafine number concentrations are to be assessed.

Acknowledgements

We gratefully acknowledge Pasi Aalto, Department of Physical Sciences of the University of Helsinki, Finland, for the valuable support and assistance in the implementation of standard operating procedures, for performing instruments calibration, for the supervision, and for solving the technical problems encountered during the use of the Condensation Particle Counters.

REFERENCES

Aalto, P., Hämeri, K., Paatero, P., Kulmala, M., Bellander, T., Berglind, N., Bouso, L., Castaño-Vinyals, G., Cattani, G., Cyrys, J., Von Klot, S., Lanki, T., Marconi, A., Nyberg, F., Pekkanen, J., Peters, A., Sjöval, B., Sunyer, J., Zetzsche, K., and Forastiere, F. 2004. Aerosol number concentration measurements in five European cities using TSI-3022 condensation particle counter over three year period during HEAPSS (Health Effects of Air Pollution on Susceptible Subpopulations). *Air Waste Manage. Assoc. J.* (in press).

Agarwald, J.K., and Sem, G.J. 1980. Continuous flow, single-particle-counting condensation nucleus counter. *J. Aerosol Sci.* 11:343-357.

CAFE Working Group on Particulate Matter 2004. *Second Position Paper on Particulate Matter, Final draft*, 6th April 2004.

Comité Européen de Normalisation (CEN). 1998. Air quality - Determination of the PM_{10} fraction of suspended particulate matter – Reference method and field test procedure to demonstrate reference equivalence of measurement methods. European Standard 12341.

Comité Européen de Normalisation (CEN) 2004. Ambient air quality – Reference gravimetric measurement method for the determination of the $PM_{2.5}$ mass fraction of suspended particulate matter, Draft prEN 14907, March, 2004.

D'Innocenzio, F., Di Filippo, P., Lepore, L., and Marconi, A. 1998. PM_{10} and $PM_{2.5}$ concentrations in urban air and size fraction distribution of toxic metals. *Ann. Chimica* 88:281-289.

Dockery, D.W. 2001. Epidemiologic evidence of cardiovascular effects of particulate air pollution. *Environ. Health Perspect.* 109:483-486.

Dockery, D.W., and Pope, C.A. 1994. Acute respiratory effects of particulate air pollution. *Annu. Rev. Publ. Health* 15:107-32.

EC 1999. Council directive 1999/30/EC of April 1999 relating to limit values for sulphur dioxide, nitrogen dioxide and oxide of nitrogen, particulate matter and lead in ambient air. *Official Journal of the European Commission (26.6.1999)*. L 163/41-60.

Gomiscek B., Hauck, H., Stopper, S., and Preining, O. 2004. Spatial and temporal variations of PM_1, $PM_{2.5}$, PM_{10} and particle number concentration during the AUPHEP-project. *Atmos. Environ*. 38:3917-3934.

Harrison, R.M., Deacon, A.R., and Jones, M.R. 1997. Sources and processes affecting concentrations of PM_{10} and $PM_{2.5}$ particulate matter in Birmingham (U.K.). *Atmos. Environ*. 31: 4103-4117.

ISO 15767: 2003. Workplace Atmospheres – Controlling and characterizing errors in weighing collected aerosols.

Janssen, L.H., Buringh, E., Van Der Meulen, A., and Van Den Hout, K.D. 1999. A method to estimate the distribution of various fractions of PM_{10} in ambient air in the Netherlands. *Atmos. Environ.* 33:3325-3334.

Janhäll, S., Jonsson, A.M., Molnàr, P., Svensson, E.A., Hallquist, M. 2004. Size resolved traffic emission factors of submicrometer particles. *Atmos. Environ.* 38:4331-4340.

Jeong, C.H., Hopke, P.H., Chalupa, D., Utell, M. 2004. Characteristics of nucleation and growth events of ultrafine particles measured in Rochester, NY. *Environ. Sci. Technol.* 38:1933-1940.

Ketzel, M., and Berkowicz, R. 2004. Modelling the fate of ultrafine particles from exhaust pipe to rural background: an analysis of time scales for dilution, coagulation and deposition. *Atmos. Environ.* 38:2639-2652.

Laakso, L., Hussein, T., Aarnio, P-, Komppula, M., Hiltunen, V., Viisanen, Y., and Kulmala, M. 2003. Diurnal and annual characteristics of particle mass and number concentrations in urban, rural and Arctic environments in Finland. *Atmos. Environ.* 37:2629-2641.

Lanki, T., Pekkanen, J., Aalto, P., Elosua, R., Berglind, N., D'Ippoliti, D., von Klot, S., Kulmala, M., Nyberg, F., Peters, A., Picciotto, S., Salomaa, V., Sunyer, J., Tittanen, P., and Forastiere, F. 2004. Effects of modelled particle number concentrations and other air pollutants on hospitalisation for first acute myocardial infarction in five European cities. The HEAPSS study. *Epidemiology* (submitted).

Marconi, A., Menichini, E., Ziemacki, G., Cattani, G., and Stacchini, G. 2000. Misure di materiale particellare PM_{10} e $PM_{2.5}$ nell'atmosfera di Roma. *Ann. Ist. Super. Sanità*. 36:285-289.

McMurry, P.H., and Woo, K.S. 2002. Size distributions of 3-100-nm urban Atlanta aerosols: measurement and observations. *J. Aerosol Med.* 15:169-78.

Milligan, P.J., Brabin, B.J., Kelly, Y.J., Pearson, M.G., Mahoney, G., Dunne, E., Heaf, D., and Reid, J. 1998. Association of spatial distribution of childhood respiratory morbidity with environmental dust pollution. *J. Toxicol. Environ. Health A.* 55:169-84.

Oberdörster, G. 2001. Pulmonary effects of inhaled ultrafine particles. *Int. Arch. Occup. Environ. Health* 74:1-8.

Oberdörster, G., and Utell, M. 2002. Ultrafine particles in the urban air: to the respiratory tract-and beyond? *Environ. Health Perspect.* 110:A440-A441.

Ostro, B., and Chestnut, L. 1998. Assessing the health benefits of reducing particulate matter air pollution in the United States. *Environ. Res.* 76:94-106.

Pope, C.A., Burnett R.T., Thun, M.J., Calle, E.E., Krewski, D., Ito, K., and Thurston, G.D. 2002. Lung cancer, cardiopulmonary mortality, and long-term exposure to fine particulate air pollution. *J. Am. Med. Assoc.* 287:1132-1132.

Ruuskanen, J., Tuch, Th., Ten Brink, H., Peters, A., Khlystov, A., Mirme, A., Kos, G.P.A., Brunekreef, B., Wichmann, H.E., Buzorius, G., Vallius, M., Kreyling, W., and Pekkanen, J. 2001. Concentrations of ultrafine, fine and $PM_{2.5}$ particles in three European cities. *Atmos. Environ.* 35:3729-3738.

Schwartz, J., Dockery, D.W., and Neas, L.M. 1996. Is daily mortality associated specifically with fine particles? *J. Air Waste Manage. Assoc.* 46:927-939.

Seaton, A., MacNee, W., Donaldson, K., and Godden, D. 1995. Particulate air pollution and acute health effects. *Lancet* 345:176-8.

Sem, G.J. 2002. Design and performance characteristics of three continuous-flow condensation particle counters: a summary. *Atmos. Research* 62:267–294.

Shi, P., and Harrison, R.M. 1999. Investigation of ultrafine particle formation during diesel exhaust dilution. *Environ. Sci. Technol.* 33:3730-3736.

Stanier, C.O., Khlystov, A.Y., and Pandis, S.N. 2004. Ambient aerosol size distributions and number concentrations measured during the Pittsburgh Air quality Study (PAQS). *Atmos. Environ.* 38:3275-3284.

Wahlin, P., Finn, P., and Van Dingenen, R. 2001. Experimental studies of ultrafine particles in streets and the relationship to traffic. *Atmos. Environ.* 35:S63-S69.

Wiedensohler, A., Wehner, B., and Birmili, W. 2002. Aerosol number concentrations and size distributions at mountain-rural, urban-influenced rural, and urban-background sites in Germany. *J. Aerosol Med.* 15:237-43.

World Health Organisation (WHO) 2002. *Guidelines for concentration and exposure-response measurement of fine and ultra fine particulate matter for use in epidemiological studies*. WHO-European Commission, Schwela, D., Morawska, L., Kotzias D. European Commission-Joint Research Center and World Health Organisation, Geneva, EUR 20238 EN.

Zhu, Y., Hinds, W.C., Kim, S., and Sioutas, C. 2002. Concentration and size distribution of ultrafine particles near a major highway. *J. Air Waste Manage. Assoc.* 52:1032-42.

IMPACT OF IRON AND STEEL INDUSTRY AND WASTE INCINERATORS ON HUMAN EXPOSURE TO DIOXINS, PCBs AND HEAVY METALS: RESULTS OF A CROSS-SECTIONAL STUDY IN BELGIUM

Sébastien Fierens, Hélène Mairesse, Jean-François Heilier,[1] Jean-François Focant, Gauthier Eppe, Edwin De Pauw,[2] Alfred Bernard[1]

ABSTRACT

We evaluated the impact of two iron and steel plants and two municipal solid waste incinerators (MSWI) in Wallonia (Belgium) on the exposure of residents to dioxins, PCBs, and heavy metals. A total of 142 volunteers living around these facilities were recruited and compared with 63 referents from a rural area with no industrial source of pollution. Information about smoking habits, dietary habits, anthropometric characteristics, residential history and health status was obtained from a self-administered questionnaire. The volunteers provided blood under fasting conditions in order to evaluate the body burden of dioxins (17 PCDD/Fs congeners) and PCBs. Samples of blood and urine were also taken for the determination of cadmium, mercury and lead. After adjustment for covariates, concentrations of cadmium, mercury and lead in urine or blood were not increased in subjects living in the vicinity of MSWIs or sinter plants by comparison with referents. Residents around the sinter plants and the MSWI located in the industrial area had concentrations of dioxins and PCBs in serum similar to that of referents. By contrast, subjects living in the vicinity of the MSWI in the rural area showed significantly higher serum levels of dioxins (geometric mean, 38 vs 24 pg TEQ/g fat, $p<0.0001$) and coplanar PCBs (geometric mean, 10.8 vs 7.0 pg TEQ/g fat, $p<0.05$). Whereas age-adjusted dioxin levels in referents did not vary with local animal fat consumption, concentrations of dioxins in subjects living around the incinerators correlated positively with their intake of local animal fat, with almost a doubling in subjects with the highest fat intake. These results indicate that dioxins and coplanar PCBs emitted by MSWIs can indeed accumulate in the body of residents who regularly consume animal products of local origin.

[1] Université catholique de Louvain, Belgium.
[2] University of Liège, Liège, Belgium.

INTRODUCTION

Emissions from industries contribute to global air pollution by emitting a large range of pollutants. Municipal solid waste incinerators (MSWIs) and iron and steel plants represent major sources of atmospheric pollution by dioxins (polychlorinated dibenzo-*p*-dioxins/dibenzofurans) (Everaert and Baeyens, 2002; Gilpin et al., 2003; Zook and Rappe, 1994). Since the mid-nineties, stringent emission standards have been imposed to MSWIs in order to achieve dioxin emission levels below 0.1 ng TEQ/Nm³. With the progressive abatement of emissions from MSWIs, the iron and steel industry is now becoming the major contributor to the global atmospheric pollution by dioxins (Anderson and Fisher, 2002; Buekens et al., 2001). The human impact of these pollutants is still of concern because of their ability to bio-accumulate in the food chain and then in human fatty tissues (Startin and Rose, 2003). Incinerators are potential sources of other pollutants, especially heavy metals which can also bio-accumulate in the food chain (Lisk, 1988; Schuhmacher et al., 2002).

In order to assess the human exposure impact of these pollutants, we carried out an epidemiological study to compare the concentrations of dioxins, polychlorinated biphenyls, cadmium, mercury and lead in blood and urine of subjects living in the vicinity of two sinter plants and two MSWIs, with levels found in referent subjects recruited in a rural area with no industrial source of pollution.

MATERIALS AND METHODS

After approval by the University Ethics Committee, a total of 142 volunteers living around either a MSWI or an iron and steel facility were recruited. Fifty-one subjects aged 21 to 80 years were living within a distance of 2 km from the MSWI of Thumaide (MSWI 1), in a rural area. This MSWI was constructed in 1980 and has a capacity of 12.4 tons/hour. Thirty-three subjects aged 33 to 65 years were recruited within a distance of 2 km from the MSWI of Pont-de-Loup (MSWI 2), in an industrial area. This second MSWI was constructed in 1978 and its capacity is 15.5 tons/hour. Fifty-eight subjects aged 25 to 67 years were living within a distance of 4 km from two iron and steel plants of the Cockerill-Sambre company (Usinor group), one plant located in the suburbs of Liège (n=12) and the other in the suburbs of Charleroi (n=46). These subjects were compared with 63 referents recruited in three villages (Bertrix, Daverdisse and Nassogne) situated in rural areas in the Ardenne (South of Belgium), with no known local source of pollution. After having given their informed consent, the volunteers provided approximately 200 ml of fasting blood and a urine sample. Information about smoking habits, dietary habits, anthropometric characteristics, residential history and health status was obtained via a self-administered questionnaire. Total fat intake was calculated from the questionnaire, on the basis of consumption of poultry, bovine and swine products. The local animal fat intake was calculated from the consumption of poultry and bovine products only. For each type of food, the amount produced locally was estimated by asking in the questionnaire the proportion produced locally by the participants themselves (like poultry products) or purchased in the farms located in the vicinity of the place of residence. Swine products were not considered for local fat intake because of the higher dependence of pigs to manufactured feeds, and their lower ability to accumulate dioxins (Bernard et al., 2002). Fish consumption information was obtained by the questionnaire. We quantified serum concentrations of the seventeen 2,3,7,8- substituted polychlorinated dibenzodioxin / dibenzofuran congeners (PCDD/Fs or dioxins), coplanar PCBs (IUPAC n° 77, 81, 126 and 169) and 12 PCB markers (IUPAC n° 3, 8, 28, 52, 101, 118, 138, 153, 180, 194, 206 and 209) by gas chromatography/high-resolution mass spectrometry (GC-HRMS) (Focant et al., 2001; Focant and De Pauw, 2002). The dioxin and coplanar PCB concentrations were expressed in toxic equivalent (WHO-TEQ) (Van den Berg et al., 1998). The concentrations of cadmium and mercury in urine and the blood lead concentrations were determined by atomic absorption spectrometry (Cardenas et al., 1993).

All variables except age and body mass index (BMI) were normalized by log transformation. Differences between groups were assessed by analysis of variance (ANOVA) followed by Dunnett's multiple-comparison post-hoc tests. Chi-square tests were used to compare homogeneity of prevalence between exposed groups and referents. Determinants of dioxin, PCB and heavy metal concentrations were identified by stepwise multiple linear regression analysis. All analyses were carried out with SAS version 8.0, Enterprise Guide version 2.0 (SAS Institute Inc., Cary, NC).

RESULTS

The characteristics of the studied population are presented in Table 1. The four groups were comparable with respect to the sex ratios (51 to 61% of women) and age, with the exception of the group around MSWI 2 who

were slightly younger than the other groups. There were no statistically significant differences in the smoking habits between the groups. The total animal fat consumption was rather similar between the four groups but as expected, the groups around MSWI 2 and the sinters plants, localized in industrial areas, consumed significantly less animal fat from local productions.

Table 1. Characteristics of the studied population.

	Referents	**MSWI 1**	**MSWI 2**	**Sinter plants**
N	63	51	33	58
Number of women	34 (54%)	26 (51%)	20 (61%)	32 (55%)
Number of men	29 (46%)	25 (49%)	13 (39%)	26 (45%)
Age (years)[a]	52.9 (7.8)	53.3 (12.5)	46.1 (8.4)*	52.0 (10.3)
BMI (kg/m^2)[b]	25.3 [24.4-26.1]	27.4 [26.1-28.7]*	27.2 [25.7-28.8]*	25.9 [24.9-27.0]
Tobacco consumption				
Number of smokers	10 (16%)	7 (14%)	5 (15%)	12 (21%)
Consumption (pack-years)	9.0 [3.9-21.1]	23.8 [9.8-57.9]	15.3 [10.5-22.2]	20.4 [10.2-40.7]
Number of past-smokers	14 (22%)	18 (35%)	4 (12%)	14 (24%)
Consumption (pack-years)	13.5 [7.8-23.6]	24.2 [14.2-41.2]	24.8 [11.9-51.8]	13.9 [7.9-24.2]
Alcohol consumption				
Number of consumers	48 (76%)	45 (88%)	23 (70%)	42 (76%)
Consumption (drinks/week)	6.1 [4.7-7.9]	9.6 [7.2-12.9]*	4.4 [3.2-6.1]	7.3 [5.7-9.3]
Animal fat consumption (terrestrial food chain)				
Total animal fat (g/week)	285 [256-316]	272 [244-304]	254 [225-288]	292 [245-347]
Local animal fat				
Number of consumers	46 (73%)	50 (98%)*	22 (66%)	22 (38%)*
Local animal fat (g/week)	99 [77-127]	109 [90-133]	66 [43-101]*	47 [33-66]*
Fish consumption				
Number of consumers	49 (79%)	49 (96%)*	28 (85%)	52 (91%)
Consumption (g/week)[c]	254 [212-302]	174 [157-192]*	184 [137-248]	283 [225-355]

Data are geometric mean [95% Confidence Interval], except otherwise stated. [a]Arithmetic mean (SD). [b]Harmonic mean [95 % CI]. BMI = Body mass index. [c]Fresh weight. *$p<0.05$ for ANOVA with Dunnett's post-hoc tests or Chi^2 tests in case of prevalence.

The mean concentrations of pollutants in blood or urine in the different groups are compared in Table 2. Residents around the sinter plants showed concentrations of dioxins, PCBs and lead in blood that were similar to that of referents. The concentrations of cadmium and mercury in urine of these residents were also not increased. There was no increase in the concentrations of these pollutants in the group living around the MSWI in the industrial area (MSWI 2). The only group to show statistically significant increases in studied biomarkers was the group of subjects living around the MSWI in the rural area (MSWI 1). These subjects showed a significant increase of the serum levels of dioxins and coplanar PCBs compared to referents as well as with the rest of the population. Their concentrations of heavy metals in the urine or blood were however not significantly increased, although this group had the highest average urinary excretion of cadmium, even after adjustment for smoking status.

All these observations were confirmed by stepwise multiple linear regression analyses testing age, gender, BMI, animal fat consumption, fish consumption, alcohol consumption, tobacco smoking and place of residence as independent variables. The accumulation of dioxin was significantly influenced by age, residence around MSWI 1 (partial $r^2=0.12$; $p<0.0001$), animal fat consumption and BMI while the body burden of coplanar PCBs was determined by residence around MSWI 1 (partial $r^2=0.20$; $p<0.0001$), age, BMI and fish consumption. After adjustment for covariates, the increases of dioxins and coplanar PCBs in the serum of residents around MSWI 1 averaged 54% and 49%, respectively. As illustrated in Figure 1, the adjusted dioxin concentrations were significantly correlated with the intake of local animal fat in residents around the two MSWIs, with almost a doubling in subjects with the highest fat intake. Such a relationship was not found in referents, reflecting the absence of contamination of local food chain in the control area.

Table 2. Concentrations of pollutants in blood or urine.

	Referents n=63	MSWI 1 n=51	MSWI 2 n=33	Sinter plants n=58	Total n=205
Dioxins and polychlorinated biphenyls					
PCDD/Fs (pg TEQ/g lipids)	23.9 [21.4-26.6]	37.9* [32.8-43.8]	24.1 [20.2-28.9]	23.8 [20.8-27.1]	26.8 [24.9-28.8]
Coplanar PCBs (pg TEQ/g lipids)	7.0 [6.1-8.0]	10.8* [9.2-12.8]	6.4 [5.1-7.9]	6.3 [5.3-7.6]	7.5 [6.8-8.2]
PCDD/Fs+cPCBs (pg TEQ/g lipids)	31.3 [28.2-34.8]	49.0* [42.4-56.7]	30.6 [25.6-36.7]	30.7 [26.8-35.2]	34.7 [32.3-37.8]
Σ12 PCB markers (ng/g lipids)	416 [385-450]	465 [406-530]	375 [320-440]	402 [358-451]	416 [393-441]
Heavy metals					
Cd urine (µg/g creatinine)	0.49 [0.41-0.59]	0.62 [0.53-0.74]	0.43 [0.33-0.58]	0.49 [0.40-0.61]	0.51 [0.46-0.57]
Hg urine (µg/g creatinine)	1.95 [1.75-2.16]	1.80 [1.53-2.11]	2.11 [1.75-2.55]	1.79 [1.55-2.06]	1.89 [1.76-2.02]
Pb blood (µg/L)	45.8 [39.4-53.2]	43.3 [36.1-51.9]	39.4 [33.3-46.6]	42.2 [36.1-49.3]	43.1 [39.7-46.7]

Data are geometric mean [95% Confidence Interval]. *p<0.05 (ANOVA with Dunnett's post hoc test).

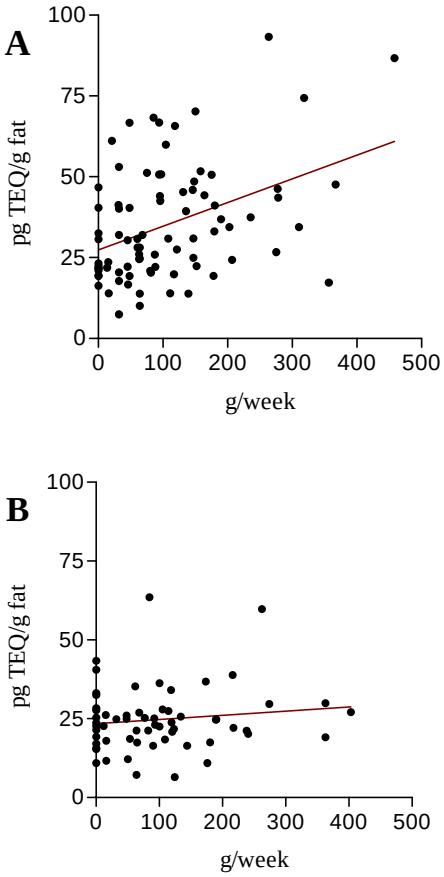


Figure 1. Correlation between dioxin serum levels and consumption of locally produced animal fat. A. Residents around the two MSWIs (n=84; Spearman r=0.37; p=0.0006). B. Referents (n=63; Spearman r=0.06; p=0.64). Dioxin values are adjusted for age, BMI and total fat consumption.

Stepwise regression analyses confirmed also the absence of significant impact of the MSWIs or of the sinter plants on the exposure of residents to heavy metals. The determinants of urinary cadmium were age, tobacco smoking and gender (higher in women). Blood lead correlated with age, alcohol consumption and gender (higher in men) whereas urinary mercury correlated only with BMI and gender (higher in women).

DISCUSSION AND CONCLUSION

The present study shows that dioxins and coplanar PCBs emitted by MSWIs can accumulate in the body of people living around these facilities. The accumulation requires however a regular consumption of local animal products contaminated by relatively high emissions of dioxins. The two studied MSWIs indeed had been in activity since the early 80s and emitted, during many years, large amounts of dioxins (>50 ng TEQ/ TEQ/Nm³). This resulted in high levels of dioxins in the food chain and in particular in cow's milk in nearby farms, some samples having concentrations up to 38.9 pg TEQ/g fat. For these reasons, it is unlikely that the increased dioxin body burden found in the present study occurs around MSWIs complying with emission standards currently in force in most countries. Although dioxin emissions of the MSWI in the rural area resulted in a significant contamination of residents, with almost a doubling of the body burden in subjects with the highest local fat intake, the observed values were comparable to values prevailing in the 80s in Belgium and most industrialized countries (Schecter et al., 1994; Wittsiepe et al., 2000). The values remained also lower than current levels found in populations regularly eating seafood (Kiviranta et al., 2000).

By contrast with MSWIs, emissions from the sinter plants were not associated with an increased dioxin body burden of residents. One possible explanation is that the dioxin emissions from the two studied plants were not sufficiently high to contaminate the local food chain. The lower local animal fat consumption of residents around these facilities has probably also contributed to decrease the risks of exposure, like also the fact that dioxins emissions from this industry are dominated by furans (PCDFs), known to be less persistent in the environment and farm animals than dioxin congeners (PCDDs) (Bernard et al., 2002).

In conclusion, although iron and steel industry and waste incinerators contribute to global air pollution, it appears that they significantly increase the exposure of residents to persistent pollutants like dioxins only when high emission levels are coupled with a regular consumption of locally produced food.

Acknowledgements

We thank volunteers who provided us with samples. This work was supported by the Ministry of Environment of the Walloon Region. S. Fierens is a research fellow of the Brussels-Capital Region. A. Bernard is research director of the National Fund for Scientific Research, Belgium.

REFERENCES

Anderson, D.R., and Fisher, R. 2002. Sources of dioxins in the United Kingdom: the steel industry and other sources. *Chemosphere* 46:371-381.

Bernard, A., Broeckaert, F., De Poorter, G., De Cock, A., Hermans, C., Saegerman, C., and Houins, G. 2002. The Belgian PCB/dioxin incident: analysis of the food chain contamination and health risk evaluation. *Environ. Res.* 88:1-18.

Buekens, A., Stieglitz, L., Hell, K., Huang, H., and Segers, P. 2001. Dioxins from thermal and metallurgical processes: recent studies for the iron and steel industry. *Chemosphere* 42:729-735.

Cardenas, A., Roels, H., Bernard, A.M., Barbon, R., Buchet, J.P., Lauwerys, R.R., Rosello, J., Hotter, G., Mutti, A., and Franchini, I. 1993. Markers of early renal changes induced by industrial pollutants. I. Application to workers exposed to mercury vapour. *Br. J. Ind. Med.* 50:17-27.

Everaert, K., and Baeyens, J. 2002. The formation and emission of dioxins in large scale thermal processes. *Chemosphere* 46:439-448.

Focant, J.F., and De Pauw, E. 2002. Fast automated extraction and clean-up of biological fluids for polychlorinated dibenzo-p-dioxins, dibenzofurans and coplanar polychlorinated biphenyls analysis. *J. Chromatogr. B-Anal. Technol. Biomed. Life Sci.* 776:199-212.

Focant, J.F., Eppe, G., Pirard, C., and De Pauw, E. 2001. Fast clean-up for polychlorinated dibenzo-p-dioxins, dibenzofurans and coplanar polychlorinated biphenyls analysis of high-fat-content biological samples. *J. Chromatogr. A.* 925:207-221.

Gilpin, R.K., Wagel, D.J, and Solch, J.G. 2003. Production, distribution, and fate of polychlorinated dibenzo-*p*-dioxins, dibenzofurans, and related organohalogens in the environment. In *Dioxins and Health*, eds. A Schecter and T.A. Gasiewicz, pp. 55-87. Hoboken: John Wiley & Sons, Inc.

Kiviranta, H., Vartiainen, T., Verta, M., Tuomisto, J.T., and Tuomisto, J. 2000. High fish-specific dioxin concentrations in Finland. *Lancet* 355:1883-1885.

Lisk, D.J. 1988. Environmental implications of incineration of municipal solid-waste and ash disposal. *Sci. Total Environ.* 74:39-66.

Schecter, A., Furst, P., Furst, C., Papke, O., Ball, M., Ryan, J J., Hoang, D.C., Le, C.D., Hoang, T.Q., and Cuong, H.Q. 1994. Chlorinated dioxins and dibenzofurans in human tissue from general populations: a selective review. *Environ. Health Perspect.* 102:159-171.

Schuhmacher, M., Domingo, J.L., Agramunt, M.C., Bocio, A., and Muller, L. 2002. Biological monitoring of metals and organic substances in hazardous waste incineration workers. *Int. Arch. Occup. Environ. Health* 75:500-506.

Startin, J.R., and Rose, M.D. 2003. Dioxins and dioxinlike PCBs in food. In *Dioxins and Health*, eds. A. Schecter and T.A. Gasiewicz, pp. 89-136. Hoboken: John Wiley & Sons, Inc.

Van den Berg, M., Birnbaum, L., Bosveld, A.T., Brunstrom, B., Cook, P., Feeley, M., Giesy, J.P., Hanberg, A., Hasegawa, R., Kennedy, S.W., Kubiak, T., Larsen, J.C., van Leeuwen, F.X., Liem, A.K., Nolt, C., Peterson, R.E., Poellinger, L., Safe, S., Schrenk, D., Tillitt, D., Tysklind, M., Younes, M., Waern, F., and Zacharewski, T. 1998. Toxic equivalency factors (TEFs) for PCBs, PCDDs, PCDFs for humans and wildlife. *Environ. Health Perspect.* 106:775-792.

Wittsiepe, J., Schrey, P., Ewers, U., Selenka, F., and Wilhelm, M. 2000. Decrease of PCDD/F levels in human blood from Germany over the past ten years (1989-1998). *Chemosphere* 40:1103-1109.

Zook, D., and Rappe, R. 1994. Environmental sources, distribution, and fate of polychlorinated dibenzodioxins, dibenzofurans and related organochlorines. In *Dioxins and Health*, ed. A. Schecter, pp. 79-113. New York: Plenum Press.

PART II - HEALTH EFFECTS

HEALTH EFFECTS ASSOCIATED WITH EXPOSURE TO AMBIENT AIR POLLUTION
Discussion Paper

Jonathan Samet.[1], Daniel Krewski[2]

ABSTRACT

The World Health Organization has identified ambient air pollution as a high public health priority, based on estimates of air pollution related death and disability-adjusted life years derived in its Global Burden of Disease initiative. The NERAM Colloquium Series on Health and Air Quality was initiated to strengthen the linkage between scientists, policy makers and other stakeholders by reviewing the current state of science, identifying policy relevant gaps and uncertainties in the scientific evidence and proposing a path forward for research and policy to improve air quality and public health. The objective of this paper is to review the current state of science addressing the impacts of air pollution on human health. The paper is one of four background papers prepared for the 2003 NERAM/AirNet Conference on Strategies for Clean Air and Health, the third meeting in the international Colloquium Series. The review is based on the framework and findings of the U.S. National Research Committee (NRC) on Research Priorities for Airborne Particulate Matter and addresses key questions underlying air quality risk management policy decisions.

[1] Department of Epidemiology, Johns Hopkins Bloomberg School of Public Health, Baltimore, MD, USA
[2] McLaughlin Centre for Population Health Risk Assessment, University of Ottawa, Ottawa, ON, Canada

INTRODUCTION

The NERAM Colloquium Series on Health and Air Quality (2001-2006) was initiated to provide an opportunity for scientists, regulators, air quality managers, public health organizations, and representatives of the business community to examine the current state of science underlying the development of air quality objectives, to identify gaps and uncertainties in the science, and to make recommendations for research and policy directions to support air quality management strategies.

The objective of the 2003 NERAM/AirNet Conference on Strategies for Clean Air and Health, the third meeting in the Colloquium Series, is to provide an integrated review of best available information in four main areas supporting policy development: i) health effects of air pollution ii) air quality modeling iii) policy tools and inputs and iv) policy options. Background papers on each of the four topics are designed to provide delegates from a variety of disciplines with a common information base for consideration during the conference deliberations. This paper provides an overview of the current state of science addressing the impacts of air pollution on human health. Its review is based around the structure afforded by a committee process initiated in 1998 by the U.S. National Research Council.

In January 1998, the U.S. National Research Committee (NRC) on Research Priorities for Airborne Particulate Matter was convened at the request of Congress following the promulgation of new National Ambient Air Quality Standards (NAAQS) for $PM_{2.5}$. Realizing the potentially vast implications of these standards and unaddressed uncertainties in the scientific evidence supporting the standards, Congress requested the committee on the premise that new research evidence would lead to greater confidence in the basis for the new particulate matter NAAQS and provide guidance on emission control strategies that would most protect public health. The Committee's first report, *Research Priorities for Airborne Particulate Matter: I. Immediate Priorities and a Long-Range Research Portfolio* (NRC, 1998) set out a conceptual framework for addressing the health effects of airborne particulate matter (Figure 1) and identified the ten most critical research needs linked to key policy-related scientific uncertainties.

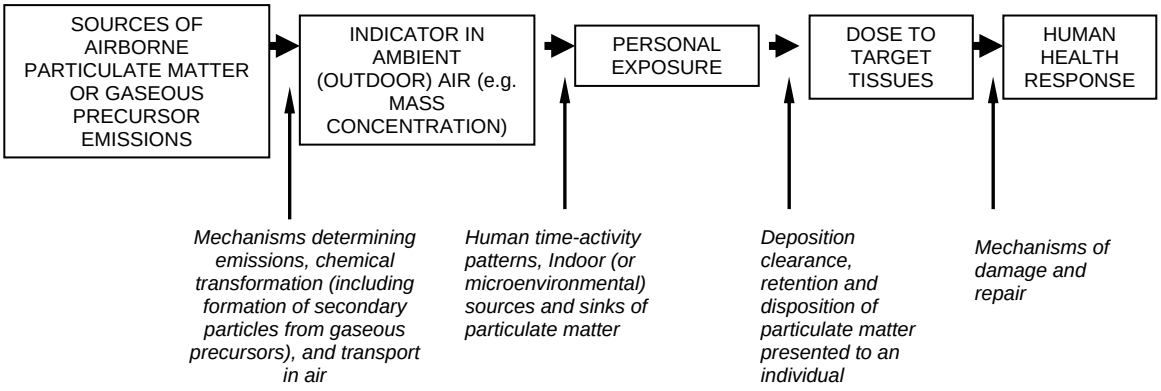


Figure 1: NRC 5 stage framework for integrating particulate matter research (NRC, 1998).

The 5 main components of the framework are as follows (NRC, 1998):

Sources—Outdoor sources of particulate matter (or gaseous precursors) that can adversely affect public health include motor vehicles; fossil-fueled electric power plants; industrial facilities; agricultural practices, consumer products; other human sources; and natural processes, such as forest fires or wind erosion. This element of the framework includes factors determining the release characteristics, dispersal, and transport of emissions that lead to atmospheric contamination.

Ambient Indicators—The mass concentration or other measures of indicators such as PM_{10} or $PM_{2.5}$ in ambient air. Data for the indicators are collected at fixed outdoor monitoring sites to determine regulatory attainment of the NAAQS, or in some health studies, to represent particulate-matter exposure in a given area.

Exposure—The concentration of particulate-matter indicator actually coming into contact with an individual over a specified period. Actual exposure to humans is determined by ambient air concentrations, contributions from indoor sources, and human time-activity patterns. The relevant point of contact is the breathing zone of the individual.

Dose—The amount and specific types of toxicants deposited in various parts of the respiratory tract and other sites within the body over a specified period. The dose of specific toxicants received by an individual is affected by factors such as retention and clearance of PM (or specific constituents of PM) from target tissues in the lung and respiratory tract. The dose delivered to specific tissues might result in injury and altered performance of repair mechanisms.

Response—Changes in specific human health parameters attributable to tissue doses resulting from inhaled particulate matter. These biological responses can be expressed in terms of molecular or cellular changes in the lung or other tissues, overall tissue damage, or ultimately, clinical signs of toxicity, such as illness or premature death.

Applying the Framework and criteria such as scientific value, value for decisionmaking, and feasibility and timing, the Committee identified the following 10 highest priority research topics (not in order of priority or timing).

Research Topic 1. Outdoor Measures Versus Actual Human Exposures

What are the quantitative relationships between concentrations of particulate matter and gaseous copollutants measured at stationary outdoor air-monitoring sites and the contributions of these concentrations to actual personal exposures, especially for subpopulations and individuals?

Research Topic 2. Exposures of Susceptible Subpopulations to Toxic Particulate-Matter Components

What are the exposures to biologically important constituents and specific characteristics of particulate matter that cause responses in potentially susceptible subpopulations and the general population?

Research Topic 3. Characterization of Emission Sources

What are the size distribution, chemical composition, and mass-emission rates of particulate matter emitted from the collection of primary-particle sources in the United States, and what are the emissions of reactive gases that lead to secondary particle formation through atmospheric chemical reactions?

Research Topic 4. Air-Quality Model Development and Testing

What are the linkages between emissions sources and ambient concentrations of the biologically important components of particulate matter?

Research Topic 5. Assessment of Hazardous Particulate Matter Components

What is the role of physicochemical characteristics of particulate matter in eliciting health effects?

Research Topic 6. Dosimetry: Deposition and Fate of Particles in the Respiratory Tract

What are the deposition patterns and fate of particles in the respiratory tract of individuals belonging to presumed susceptible subpopulations?

Research Topic 7. Combined Effects of Particulate Matter and Gaseous Co-Pollutants (Toxicological and Epidemiological Studies)

How can the effects of particulate matter be disentangled from the effects of other pollutants? How can the effects of long-term exposure to particulate matter and other pollutants be better understood?

Research Topic 8. Susceptible Subpopulations

What subpopulations are at increased risk of adverse health outcomes from particulate matter?

Research Topic 9. Mechanisms of Injury

What are the underlying mechanisms (local pulmonary and systemic) that can explain the epidemiological findings of mortality/morbidity associated with exposure to ambient particulate matter?

Research Topic 10. Analysis and Measurement

To what extent does the choice of statistical methods in the analysis of data from epidemiological studies influence estimates of health risks from exposures to particulate matter? Can existing methods be improved? What is the effect of measurement error and misclassification on estimates of the association between air pollution and health?

The NRC study serves as a comprehensive framework for reviewing the current state of scientific evidence underlying the linkage between exposure to air pollution and human health effects and for examining progress towards addressing critical research needs for air quality policy development. Two reports have subsequently been released (National Research Council, 1999; National Research Council, 2001). These reports provided some redirection of the initial research agenda, expanding the work to be done on emissions characterization and modeling, and also charting the initial progress. At the time of writing this paper the Committee was finalizing its fourth and final report, gauging progress on the research agenda.

This paper will: i) discuss the findings of North American and European studies according to the NRC priority research topics; ii) synthesize the findings according to key questions underlying air quality policy decisions; and iii) discuss the implications of the findings for the development of future air quality management strategies.

POPULATION HEALTH EFFECTS OF PARTICULATE AIR POLLUTION

The World Health Organization (2002) has identified ambient air pollution as a high priority in its Global Burden of Disease initiative and estimated that air pollution is responsible for 1.4% of all deaths and 0.8% of disability-adjusted life years globally. In this section, we examine the population health impacts of air pollution, including both mortality and morbidity. Characterizing the population health risks of ambient air pollution is critical to the development of risk management policies and strategies. Our discussion focuses on the following question.

What are the Adverse Health Effects of Particulate Air Pollution?

Epidemiological and toxicological studies have demonstrated that air pollution is associated with a range of adverse health outcomes, ranging from mortality to subclinical respiratory symptoms. Some investigations have provided estimates of the population health impact of ambient air pollution in terms of the outcomes given in the air pollution health effects pyramid shown in Figure 2. For example, Kunzli et al. (2000) found that air pollution accounted for more than 25,000 new cases of chronic bronchitis (adults), more than 290,000 episodes of bronchitis (children), more than 0.5 million asthma attacks, and more than 16 million person-days of restricted activities in Austria, France, and Switzerland.

Evidence of associations between mortality and particulate air pollution is derived primarily from time-series and cohort studies. Time-series studies are used to associate daily fluctuations in ambient air pollution levels with daily mortality rates, and have consistently demonstrated increased mortality on or shortly following days with increased ambient particulate levels in urban centres worldwide. The most convincing data of this type is derived from combined analyses of data from multiple urban areas, including 90 of the largest U.S. cities (Dominici et al., 2003a), as well as 25 European (Katsouyanni et al., 2003), and 8 Canadian (Burnett et al., 2000) cities. An ongoing analysis of data from the U.S., Europe, and Canada (Katsouyanni et al., 2002) will provide the most complete compilation and assessment of time-series data on particulate air pollution and mortality to date.

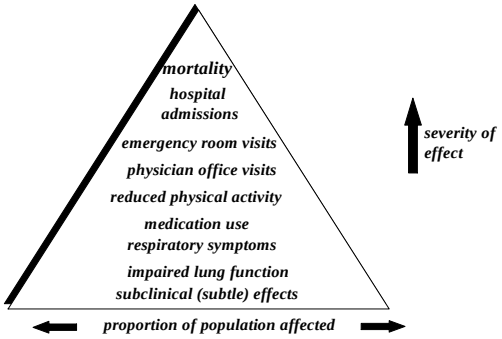


Figure 2. Air pollution health effects pyramid.

Because air pollution is a complex mixture of particulate and other co-pollutants, it is important to consider the relative contributions of all pollutants to pollution-related mortality. Burnett et al. (2000) found gaseous co-pollutants to be associated with increased mortality, with ozone, carbon dioxide, sulfur dioxide, and nitrogen dioxide all accounting for an appreciable proportion of excess morality associated with short-term exposure to ambient air pollutants. Stieb et al. (2002a) conducted a meta-analysis of published excess mortality rate ratios from time-series studies, and found significant increases in risk for both particulate and gaseous co-pollutants.

Urban air also contains a number of toxic substances such as benzene (Krewski et al., 2000) and manganese (Salehi et al., 2003) that require consideration. Burnett et al. (2000) have shown that trace elements including zinc, nickel and iron are associated with increased mortality in time-series studies based on data from 8 Canadian cities. Although air toxics are not the primary focus of this article, the National Research Council (2001) has developed methodologies for assessing the risk of short-term exposures to highly hazardous substances for periods ranging from 10 minutes to 8 hours, and has developed exposure guidelines for three levels of adverse effect: irritation, disabling effects, and lethality (National Research Council, 2000, 2002, 2003).

The analysis of time-series data of this type requires careful consideration of seasonal trends in the data, as well as adjustment for important covariates such as climate and gaseous co-pollutants (Burnett et al., 1995). Although Dominci et al. (2003) and Ramsay et al. (2003a, b) have recently shown that time-series analysis can lead to overestimation of risk as well as overstatement of the precision of risk estimates due to convergence problems in the generalized additive model, correction of this problem using more stringent convergence criteria confirms that mortality is associated with short-term fluctuations in particulate air pollution (HEI, 2003a). Case-crossover analyses can also be applied to time-series data (Lin et al., 2002, 2003a). Although these analyses avoid the need to consider secular trends in the data, case-crossover and time-series generally lead to similar conclusions (Fung et al., 2003).

Although time-series analyses provide important information about the effects of short term exposure on mortality, they are unable to identify effects with a lag period longer than several weeks (Dominici et al., 2000). The effects of long-term exposure to particulate air pollution can be identified through cohort mortality studies. Dockery et al. (1993) reported an increase in overall and cardiorespiratory mortality in six cities in the midwest and northeast U.S. in relation to $PM_{2.5}$ concentrations monitored from the mid 1970s through to 1989. Pope et al. (1995) reported similar findings by following a group of over 550,000 subjects established by the American Cancer Society in over 150 large U.S. cities from 1982 to 1989. Further follow-up of this cohort through to 1998 confirmed the original findings, and provided evidence of an increased risk of lung cancer mortality due to $PM_{2.5}$ (Pope et al., 2002). Additional analyses of this cohort, with follow-up through to 2000, are underway (Krewski et al., 2003; 2004). Abbey et al. (1999) are continuing the Adventist Smog and Health (ASHMOG) cohort study of 6,338 nonsmoking California Seventh-day

Adventists to further assess the relationship between mortality and long term exposure to particles and other air pollutants.

These cohort mortality studies have demonstrated consistent associations between long-term exposure to particulate matter and mortality, and have played a pivotal role in the establishment of air quality objectives for $PM_{2.5}$ in the U.S. (Greenbaum et al., 2001). Krewski et al. (2000, 2003) showed that long-term exposure to sulfur dioxide was also associated with mortality in their re-analysis of the ACS cohort with follow-up through to 1989, originally examined by Pope et al. (1995). Although SO_2 demonstrated more robust associations with mortality than did $PM_{2.5}$ in this comprehensive re-analysis, a plausible biological pathway by which SO_2 might lead directly to increased mortality has not been identified. In contrast, $PM_{2.5}$ is now thought to lead to increased cardiovascular mortality as a consequence of accelerated atherosclerosis (Pope et al., 2003), possibly mediated by the induction of endothelins and cytokines.

An important consideration in the evaluation of cohort studies linking spatial variation in ambient air pollution levels with mortality is the possibility of spatial autocorrelation, leading to a lack of independent observations within the source region (Jerrett et al., 2003a, b). Statistical methods for addressing spatial autocorrelation in studies of this type are beginning to emerge (Çakmak et al., 2003; Ma et al., 2003) and have been applied in analyses of the ACS cohort (Krewski et al., 2003). An important consideration in the appraisal of spatial data is the choice of the geographic unit of analysis, which can affect risk estimates (Willis et al., 2003).

The time-series and cohort studies conducted to date indicate that both short-term and long-term exposures to particulate matter can lead to increased mortality in the general population. The extent of associated life shortening, however, remains uncertain. The possibility that short-term exposures may primarily affect frail individuals with pre-existing co-morbidities has been investigated using time-based regression approaches, initially by Zeger et al. (1999) and subsequently by Dominici et al. (2003b), Zanobetti et al. (2000), Schwartz (2000), and Fung et al. (2004). The findings indicate that the so-called harvesting hypothesis cannot fully explain all of the excess mortality associated with short-term exposures to particulate air pollution.

Exposures to particulate air pollution have been shown to increase the number of hospital admissions for a range of cardio-respiratory diagnoses and events (Peters et al., 2000). Particles have been implicated as an explanation for an increase in cardio-respiratory admissions for 168 acute care hospitals in Ontario, Canada (Burnett et al., 1995), for cardiopulmonary admissions in Los Angeles (Linn et al., 2000), for acute respiratory admissions (Oftedal et al., 2003), and for admissions diagnosed as respiratory, cardiac, and both cerebral and peripheral vascular diseases (Burnett et al., 1999).

A number of studies have also investigated the contribution of particles and gaseous forms of air pollution to both acute and chronic morbidity-related health effects. Particulate air pollution has been linked to a decline in airway hyperresponsiveness among children in more polluted areas (Jang et al., 2003). Gent et al. (2003) found that acute episodes of asthma for children using maintenance medications are linked to pollutant levels far below EPA standards. In another study, a decline in lung function was positively associated with a six-month period of high ozone and particle concentrations (Calderon-Garciduenas et al., 2003). Air pollution has also been positively associated with decreased lung function in elementary school children (Koenig et al., 1993). Stieb et al. (2002b) found a 22% increase in the number of disability days associated with a change in particulate concentration equal to the interquartile range.

There is also compelling evidence that morbidity from particulate and gaseous air pollution is found more often among particularly susceptible populations. The evidence comes from time-series studies and also of observational studies of susceptible persons in panel studies and case-based designs. A discussion of plausible mechanisms that could be responsible for attenuating the effect of air pollution on cardiovascular morbidity has been discussed elsewhere (Utell et al., 2002). Age is also a strong predictor of susceptibility. Many studies have concluded that young children and the elderly are more at risk for respiratory and cardiopulmonary disease associated with exposure to air pollution (Burnett et al., 1997a, b; Anderson et al., 2003; Sunyer et al., 2003). Recent research suggests health impacts of air pollution in early stages of human development (Burnett et al., 2001). In one study, low ambient air

pollution concentrations were associated with adverse pregnancy outcomes including low birth weight, preterm birth, and intrauterine growth retardation (Liu et al., 2003). In another study, particles were found to be positively associated with first hospitalizations due to respiratory disease in early childhood (Yang et al., 2003). Level of socioeconomic status has been shown to modify the effect of ambient air pollution associated with asthma hospitalizations for males and females (Lin et al., 2003b). Recent work on the potential mechanisms of action of particulate air pollution point to pathways also influenced by diabetes; diabetics have double the risk of particulate-related cardiovascular admissions when compared to non-diabetics (Zanobetti & Schwartz, 2002).

The use of alternative research designs to the usual time-series approaches has yielded similar results and strengthens the case for causal interpretation of the air pollution and morbidity associations. For example, Lin et al. (2002) used both case-crossover and time-series analyses to assess the associations between particulate matter and asthma hospitalization among children 6-12 years of age. Both approaches yielded similar positive associations between coarse particles and hospitalization. A subsequent study by the same team used the case-crossover design to find a differential effect of gaseous air pollution on the same cohort. A significant effect of carbon monoxide and sulphur dioxide was found for boys and girls respectively, while the effect of nitrogen dioxide was positively associated with asthma hospitalization for both sexes (Lin et al., 2003b). Karakatsani et al. (2003) performed a case-control study nested in a defined cohort to investigate the association between long-term exposure to ambient air pollution and the development of chronic bronchitis, emphysema, and COPD.

These results indicate that both short-term and long-term exposures to particulate air pollution have been associated with increased mortality. Whereas the cohort studies conducted to date have revealed excess relative risks on the order of 15-30% for long-term exposure to 18 $\mu g/m^3$ $PM_{2.5}$, short-term exposures generally lead to excess relative risks of 5% or less. The apparently discrepant effects in the short- and long-term time frames remain unexplained. New methods for the joint analysis of time-series and cohort studies are being developed that will help to determine the relative contributions of short-term and long-term exposures to the burden of mortality associated with exposure to particulate air pollution.

There is now convincing and consistent evidence that particulate matter, and other forms of ambient air pollution, can exacerbate events leading to increased cardio-respiratory morbidity. As with the particle-mortality association, the most compelling results stem from research and analyses of multiple urban centres, and include the 16 cities across Canada representing 12.6 million people (Burnett et al., 1997a), 8 cities from across Europe (Atkinson et al., 2001; Le Tertre et al., 2002), as well as 8 urban counties (Schwartz, 1999) and 20 cities in the U.S. (Samet et al., 2000). Most air pollution-morbidity studies adopt time-series approaches and variation in findings occurs mainly in terms of location, characteristics of population under investigation, and by health outcome.

Major gains have also been made in understanding the intermediary mechanisms underlying cardiovascular and subtle pulmonary responses to ambient particulate air pollution. Controlled exposures studies indicate that laboratory-generated concentrated ambient particles (CAPs), carbonaceous ultrafine particles or diesel exhaust particles have provided support for effects on pulmonary or systemic markers of inflammation and leukocyte recruitment (Frampton, 2001). Elderly and young persons exposed to concentrated ambient particles in clinical studies have shown increases in fibrinogen levels (Devlin, 2003). Research has also found that exposure to ambient particle concentrations results in increased blood levels of endothelins which can affect vascular tone and endothelial function (Vincent et al., 2001; Brook et al., 2002). The integration of these findings with results from animal toxicology and in vitro studies allow for discussion of the biological mechanisms underlying health effects from ambient particle exposures and offer support for the epidemiologic findings previously described. More importantly, insight into the mechanisms underlying the actions of particles in the production of oxidative stress and cardiovascular effects provides a basis for understanding how particles cause adverse health effects.

EVIDENCE OF ADVERSE HEALTH EFFECTS: ADDRESSING NRC RESEARCH PRIORITIES

Current concern for airborne particulate matter and health dates to the early 1990s when a series of papers was published linking daily mortality counts to levels of particulate matter on the same or recent days. These findings were unexpected, as mortality was considered to be a crude and insensitive outcome for assessing the health effects

of air pollution, and levels of particulate matter had sharply declined from several decades earlier, when excess mortality had been clearly documented at times of high air pollution. Concern was heightened by the 1993 report from the Harvard Six Cities study (Dockery et al., 1993) which associated longer-term exposure to air pollution with mortality, implying that the findings of the daily time-series studies did not reflect only a brief life shortening (i.e., "harvesting" or "mortality displacement"). The findings of the Harvard study were replicated in a 1995 report from the American Cancer Society's nationwide study involving about one-half million Americans (Pope et al., 1995).

Initially, there was substantial skepticism around these reports and alternative explanations to a causal association of particulate matter with adverse health effects were raised including methodologic problems in the study designs and analytical methods and uncontrolled confounding by the effects of other factors, particularly weather. As particulate matter always exists as one component of a complex pollution mixture, the certainty with which effects could be attributed to particulate matter, versus other mixture components or to the mixture itself, was also questioned. Researchers were also not confident about the mechanisms that might underlie the effects observed at contemporary ambient levels of particulate matter.

Consequently, substantial research was initiated about a decade ago to reduce these critical uncertainties and to strengthen the evidence base for control of the health effects of particulate matter. As described earlier, a research agenda for this purpose was set out in early 1998 by the Committee on Research Priorities for Airborne Particulate Matter of the US National Research Council (1998). That Committee offered a systematic framework for identifying key uncertainties and for targeting research to reduce these uncertainties. In a multi-year research portfolio, the Committee set out ten topics that were to be addressed in a sequenced fashion. As this manuscript was prepared, the Committee's fourth report, coming approximately five years after its first was yet to be published. That report will provide a review of research over the five years from 1998, the time of the first report, to 2003, when the last report was prepared.

In this document, we offer a general review of progress in the decade since the early 1990s, drawing on our general knowledge of this area as well as several key synthesis documents (the US Environmental Protection Agency's draft Criteria document and Staff Paper; reviews prepared by WHO Euro, and the peer-reviewed literature in general). We note that there is now a greater consensus as to the role of airborne particles in causing adverse health effects; in fact, debate has shifted away from whether there is an effect of particulate matter to questions as to aspects of particles that determine toxicity and to the sources of those particles that are most injurious to health.

In several areas, substantial progress has been made over the last decade. Substantial research has been carried out on personal exposures to particulate matter and the contribution of particles from outdoor sources to total personal exposure, an area of investigation corresponding to Topic 1 of the National Research Council's Committee. Studies have been carried out in the United States and in Europe that have involved personal monitoring of various population groups of interest in order to characterize the contribution of outdoor particles to personal exposure, and particularly to variation in person exposure over time. This research was initiated, in part, to better understand the implications of using ambient monitors for population-level studies of particles and health, and also to considering exposure-based control strategies. The monitored populations have included children, healthy adults, and adults in groups within the population considered to be at risk for health effect of air pollution, including persons with chronic obstructive coronary heart disease and pulmonary heart disease. These studies have documented that particles in outdoor air penetrate indoors and make substantial contributions to day-to-day variation in total personal exposure, a critical finding with regard to interpreting the time-series studies. This work has not yet addressed personal exposures to particles in relationship to specific characteristics of the particles, based on size or chemistry; most have involved monitoring for $PM_{2.5.}$

Progress has been made in other areas, although without the relatively definitive overall gains that have been made for the topic of personal exposure. Work to advance understanding of mechanisms of toxicity continues to provide multiple leads, perhaps unified by the finding that particles incite inflammation and release of oxygen radicals with the potential for injury at sites of deposition. Particles also activate inflammatory cascades that may have local and systemic actions. Mechanisms by which inhaled particles could affect organs distant from the site of deposition are of particular interest, given the many studies now showing adverse effects of particles on the heart.

With regard to public health protection, identifying susceptible groups within the overall population is of particular interest. These groups may be susceptible on the basis of an underlying condition, particularly chronic heart and lung diseases, and vulnerable to exacerbation of these diseases. If these groups are to be protected, we need to characterize the exposure-response relationships for these adverse effects. Some information on dose-response relationships has been obtained by controlled exposures of human volunteers ("clinical studies") to concentrated ambient particles. The results have been variable across the centers where this work has been done, with some but not all studies showing evidence of adverse effects.

New methods have also been developed for research on particles, both for toxicologic and epidemiologic studies. The particle concentrator offers a useful tool for carrying out both animal and human studies. A principal limitation is that particles can be concentrated only in a single location; some groups are using portable concentrators so that studies can be carried out in locales of particular interest. Epidemiological approaches have also evolved. The single-city time-series study of a morbidity or mortality indicator has largely been replaced by multi-city studies, which are more informative because they gain information from both within-city temporal comparisons and across-city geographic comparisons. Examples of these approaches include the Air Pollution and Health: A European Approach (APHEA) studies and the National Morbidity Mortality and Air Pollution Study (NMMAPS) carried out in the United States. Additionally, the methodologic issues of "harvesting" and measurement error have been considered and their implications for interpreting the time-series studies addressed to some extent. Several analytic approaches have been described that provide some discrimination between harvesting and an effect of air pollution on the short to medium-term (one to two months). Several new cohort studies are in progress that are intended as efficient approaches to further characterizing the long-term effects of air pollution on mortality. For example, the group at Johns Hopkins is using the U.S. Medicare system to address short-and long-term effects of air pollution.

Over the last several years, new methodologic issues were raised in regard to the time-series approach. A problem was found in the application of the GAM function in S-plus to analyses of air pollution time-series data. The convergence criteria that were set by default were insufficiently strict for air pollution time-series data and incomplete convergence introduced an upward bias in the results of a number of critical studies. Reanalyses have now been carried out for some of the most significant of these studies, including APHEA and NMMAPS. Overall, the risk estimates tended to decline with reanalysis. Additional, more subtle methodologic issues were also described related to the adequacy of control for confounding and the sensitivity of findings to the particular statistical model chosen. While these issues have some generally minor quantitative implications, they have little consequence for interpretation of the general body of time-series studies of air pollution and health.

Some other critical topics on the research agenda for particulate matter and health remain largely unaddressed. The characteristics of particles that may figure in their toxicity have yet to be adequately characterized. Research in this area is complicated by the diversity of sources of particles and the resulting spectrum of characteristics. Additionally, different characteristics may be relevant to different health outcomes. The matrix defined by particle characteristics and health outcomes has yet to be satisfactorily explored. There has also been only limited research on combined effects of particles with other pollutants.

WHERE DO WE GO FROM HERE: POLICY DIRECTIONS

The preceding discussion of the current state-of-the-science on air pollution and health provides a basis for examining policy options for air pollution control. Specifically, we address the following policy related questions.

What are the qualitative and quantitative health risks at current ambient levels and how are they distributed in the population?

Epidemiologic studies have been able to identify a wide range of adverse health effects in the general population associated with current ambient air pollution levels, ranging from mortality to subtle subclinical effects (Figure 2). As noted earlier, long-term exposures to particulate air pollution are associated with larger relative risks of mortality than are short-term exposures, possibly a reflection of the cumulative effects of protracted exposure. These population based studies are notable in their consistency, and are supported by both toxicological and clinical

research (National Research Council, 2002, Health Effects Institute, 2003a; Cohen et al., 2003). Overall, the World Health Organization (2002) estimates that ambient air pollution may be responsible for some 1.4% of all deaths occurring annually. Cardiorespiratory health outcomes are of particular concern in terms of both morbidity and mortality, demonstrating the strongest associations with both particular and gaseous pollutants.

Evidence to date has demonstrated associations between ambient air pollution and adverse health outcomes across all segments of the population, including potentially susceptible subgroups such as children and the elderly (Wang et al., 2003). Although it has been postulated that air pollution may lead to increased mortality primarily affecting frail individuals with pre-existing co-morbidities, retrospective time-series studies have been unable to clearly identify specific co-morbidities that significantly modify pollution-mortality associations (Goldberg and Burnett, 2004). Empirical studies also indicate that pollution related mortality is not confined to individuals with pre-existing health conditions that may render them susceptible to air pollution exposure (Dominici et al., 2003b). Socio-economic status (SES) has been shown to modify the association between both long-term (Lin et al., 2003b) and short-term (Villeneuve et al., 2003) exposure to particulate air pollution, with risk increasing with decreasing SES. However, since SES is a marker for a complex set of lifestyle and other factors, the implications of this observed effect modification by SES are unclear at this time.

Which pollutants contribute most to population health effects?

Given a sufficiently strong evidence base, pollutant control strategies might be directed towards those sources contributing the greatest burden of risk to health. To establish the needed evidence base, researchers need to address the health effects of the overall air pollution mixture and then to decompose those effects into the source contributions, linking sources to pollutants to health effects. This is a difficult topic that would require an ambitious research agenda.

At present, regulators address a suite of major pollutants, including generally particulate matter, ozone, carbon monoxide, nitrogen dioxide and sulfur dioxide. This strategy has long been in place and partially reflects the view that each of these pollutants conveys risk to human health. However, there is a mix of primary pollutants (particles, carbon monoxide, sulfur dioxide, and nitrogen dioxide) and of secondary pollutants (particulate matter and ozone). Additionally, sulfur dioxide and nitrogen dioxide contribute to the formation of secondary particles and nitrogen dioxide is critical in the formation of ozone. Given these interrelationships and the not surprising correlations among concentrations of these pollutants, separating their effects may prove difficult. The emphasis on identifying toxicity-determining characteristics of particulate matter is even more challenging, as particulate matter itself is a mixture having multiple sources.

Can we identify the most important pollutants requiring mitigation?

Air pollution is a complex mixture of a number of individual pollutants, including particulates, gases, bioaerosols, and toxic substances. The concentrations of USEPA “criteria” pollutants such as particulate matter, ozone, sulfur dioxide, carbon monoxide, and nitrous oxides, tend to be highly correlated with each other in some locations, making it difficult to identify the effects of individual pollutants in observational studies. Estimates of the population health burden of short-term exposure to individual pollutants suggest that both particulate and gaseous co-pollutants contribute significantly to mortality in the general population (Burnett et al., 2000). Detailed analyses of the ACS cohort have shown mortality to be associated with both fine particulate matter and sulfur dioxide, with SO_2 being somewhat more robust against adjustment for co-pollutant exposure than $PM_{2.5}$ (Pope et al., 2002; Krewski et al., 2003; 2004). Since a plausible biological pathway exists for mortality associated with exposure to particulate matter, but not for sulfur dioxide, it is possible that SO_2 is a marker for exposure to another pollutant such as sulfate particles.

The weight of evidence suggests that particulate air pollution is an important component of the complex mixture to air pollutants to which we are exposed (Greenbaum, 2003). $PM_{2.5}$, which can penetrate deeper into the airways than coarser particles, has been the focus of intensive investigation, and has been consistently associated with morbidity

and mortality in the general population. Recent evidence has suggested a mechanistic pathway based on accelerated atherosclerosis for particulate induced cardiorespiratory conditions.

Current evidence also suggests a role for gaseous co-pollutants, particularly ozone, in the etiology of pollution-related morbidity and mortality. However, present evidence precludes a clear demarcation of the relative contribution of particulate and gaseous pollutants to the population health burden of ambient air pollution.

How effective would source control be in terms of reducing exposures to multiple pollutants?

The HEI Accountability working group (HEI, 2003) notes that while the link between a new regulation and resulting changes in emissions is the most straightforward component of the chain of accountability, its assessment has a number of possible complications.[3] Changes in ambient concentration resulting from specific emission reduction policies may be difficult to detect in routinely collected data on ambient air quality. A number of factors influence changes in ambient concentrations including the relative contribution of the regulated source to total emissions of the pollutant in question, concurrent changes in emissions of the same pollutant by other sources, and variation in meteorologic influences on atmospheric transport and chemical transformation. The link from ambient air quality to exposures and doses in the population are influenced by time-activity patterns, penetration of outdoor pollutants indoors and chemical transformations. In general, tools available to characterize the effects of regulatory action on pollutant emissions and air quality are better developed than tools for assessing the relation between ambient pollutant concentrations and exposures and doses received by the population.

Tracking the effect of changing emissions on air quality can be relatively straightforward for some primary pollutants, such as lead. The decline of atmospheric lead concentrations and blood lead levels following the phase out of leaded gasoline provides an example illustrating the close linkage between source control and reduced exposures. Tracking changes in secondary pollutants (e.g. ozone and fine particles) is more complex. For example, changes in vehicle emissions cannot be readily linked to ozone concentrations in urban areas, as concentrations are determined by multiple factors beyond emissions of ozone precursors. Models provide a means to quantify the complex chemical and meteorological interactions that relate precursor emissions to production of secondary pollutants and play a central role in air quality management for $PM_{2.5}$ and ozone.[4]

The HEI monograph notes that surprisingly little effort has been devoted to assessing how accurately models predict changes observed after implementation of specific emission policies. The expansion of speciation monitoring in the United States will provide improved tools for tracking ambient effects of changes in emissions and judging the validity of modeled projects and the effectiveness of corresponding control programs. To further assess the benefits of specific emission control policies, the monograph recommends that carefully designed small scale exposure studies of specific subpopulations be conducted to complement a large US personal exposure study, the National Health and Nutritional Examination Survey (NHANES), which recently assessed personal exposures to and blood levels of VOCs among a stratified random sample of 1000 US adults.

The accumulation of additional air pollution monitoring data will be of great value in tracking trends in ambient pollutant concentrations. Emission source controls on point and mobile sources have become increasingly stringent in the United States in recent decades (National Research Council, 2004). New models linking emissions data for point and mobile sources, to ambient concentrations, taking chemical and meteorological conditions into account, will provide a basis for an assessment of the benefits of sources control on exposures to multiple pollutants.

Should efforts be focused on the control of individual pollutants or should broader source control strategies focusing on air pollution mixtures be implemented?

The scientific basis for deciding among these (or other options) lies in answering the previous question with sufficient certainty. In part, focusing on particulate matter and ozone, whose concentrations are determined by a

[3] The material in this section is drawn from HEI (2003b) Chapter 3 “From Regulatory Action to Exposure and Dose”.
[4] See Reid et al. in this issue for further discussion of air quality models as a policy tool.

broad range of sources, may be an effective approach to controlling air pollution generally. Carbon monoxide has specific cardiac effects and emissions of carbon monoxide from motor vehicles have been drastically curtailed, illustrating the efficacy of targeted source control strategies when warranted by scientific evidence. For most countries and specific locales, we also lack the emissions inventory information that would be needed to target source control strategies and our tools for modeling the contributions of specific sources to concentrations and exposures are inadequate. At this juncture, we offer the opinion that the scientific evidence does not yet warrant a deviation from current approaches.

How have health risks declined with declining levels of air pollution?

Ambient air pollution levels have been generally declining over the last 20 years. Although this reduction in exposure to air pollution may be expected to lead to reduced health risk, it is difficult to directly demonstrate concomitant reductions in mortality and morbidity in the general population because of the modest reductions in ambient pollution levels and the relatively small attributable risk associated with air pollution. Recent time-series studies have consistently identified associations between air pollution and both mortality and morbidity, indicating that even current low levels of pollution are associated with adverse health outcomes. Cohort studies involving follow-up over extended periods afford an opportunity to assess changes in risk with changes in ambient air pollution levels. In the Harvard six cities study conducted by Dockery et al. (1993), prospectively monitored pollution levels exhibited a notable decline between the early 1970s and the late 1980s. Villeneuve et al. (2002) attempted to characterize the relationship between temporal changes in $PM_{2.5}$ levels and mortality in this study: it was found that the relative risk of pollution related mortality did not depend on when exposure occurred in relation to death, possibly because of the limited variation in the rate of decline in ambient pollution levels in these six cities. Follow up of the ACS cohort through to 1989 (Pope et al., 1995) and 1998 (Pope et al., 2002) suggested a decrease in the relative risk of mortality associated with $PM_{2.5}$. A true decline in the relative risk of mortality would, given a stable population at risk, imply a reduction in population attributable risk. However, it is unclear if this reduction in relative risk is due to increased exposure misclassification owing to increased population mobility in the extended follow-up.

What is the current evidence of benefits related to air pollution mitigation?

With increasing economic costs of control measures to achieve further improvements in air quality in North America and western Europe, policy makers and other stakeholders are seeking evidence of public health improvements resulting from past efforts to reduce air pollution.

The Health Effects Institute's recent monograph *"Assessing Health Impact of Air Quality Regulations Concepts and Methods for Accountability Research"* (HEI, 2003b)[5] states that carefully constructed evidence about the extent to which control measures have improved health is still lacking and proposes a framework and methods for accountability research. The challenges in assessing the health impact of air quality regulations are considerable. Diverse approaches are required to evaluate the impact of national, regional and local level interventions that take effect at various time frames. PM and photochemical oxidants are complex mixtures with multiple sources. Isolating the health effective component of various pollutant specific control measures is difficult. Large regulatory initiatives such as the US Clean Air Act may require long time frames before reductions in pollutant emissions translate into reduced incidence of mortality and morbidity from cardiovascular and respiratory diseases. The longer the time between implementation of a policy and manifestation of changes in disease incidence, the greater the possibility that other behavioral and environmental factors may influence risk.

Estimating benefits and costs

In developing clean air policies, governments have used risk estimates from epidemiologic studies to estimate avoided health impacts associated with various air quality scenarios (EPA 1997, 1999, Canada Wide Standards Development Committee for Particulate Matter and Ozone, 1999). These health impact estimates are expressed in monetary terms based on economic valuation literature and compared with various baseline scenarios and the costs of emission controls to achieve various ambient air quality levels. In its retrospective and prospective analysis of the

[5] The material in this section is drawn from the HEI Monograph. (HEI, 2003b).

Clean Air Act, EPA relied primarily on the long-term American Cancer Society cohort study by Pope and colleagues (1995) to estimate adult mortality associated with elevated PM. The retrospective analysis showed that aggregate monetized benefits ($22,171 billion 1990 dollars) are estimated to exceed costs ($523 billion 1990 dollars) by more than an order of magnitude for the first twenty years of the Clean Air Act (1970 to 1990). More than four-fifths of these estimated benefits are derived from avoided mortality at $4.6 million per life. The decline in PM concentrations during the first 20 years of the Clean Air Act is associated with the majority of mortality benefits. The phase out of lead in gasoline also contributed to reduced mortality during this period. In the prospective analysis, from 1990 to 2010, total monetized benefits also exceeded costs although the ratio of benefits to costs was lower than for 1970 to 1990. Subsequent disaggregated analyses suggest major differences in net benefit across titles of the Clean Air Act. Most analyses show that the new $PM_{2.5}$ NAAQS is likely to have a better cost-benefit ratio than the new ambient ozone standards.[6]

In the United Kingdom, the Committee on Medical Effects of Air Pollution (COMEAP) estimated the total life-years gained by the 2001 population of England and Wales if the annual average $PM_{2.5}$ concentrations were 1 $\mu g/m^3$ lower than that occurring in 2000. The estimates span a 20 fold range of potential gain in life expectancy from 0.2 to 4.1 million life-years (UK Department of Health, 2001).

Several quasi-experimental, observational studies have evaluated the health benefits associated with emission reductions. These studies include those conducted in Dublin, Ireland (Clancy et al., 2002) and Hong Kong (Hedley et al., 2002) in which new air quality regulations implemented in relatively short times had a major impact on emissions, and studies in which point-source industrial emissions decreased dramatically (Pope, 1989).

Focused air pollution regulations

The impact on mortality six years following a 1990 ban on the use of coal for domestic heating in Dublin was estimated. Following the ban, concentrations of black smoke and sulfur dioxide fell 35.6% and 11.3% respectively. Age standardized mortality from natural causes declined by 5.7%. The declines were more substantial for people less than 60 years of age (7.9%), for mortality due to respiratory causes (1.5%) and cardiovascular causes (10.3%) as opposed to other causes (1.7%) and in winter. The estimates were adjusted for weather, influenza epidemic and secular changes in countrywide mortality. The impact was qualitatively similar but quantitatively larger than what one would have predicted from existing European studies of air pollution and mortality (Katsouyanni et al., 1997).

In 1990, the sulfur content of fuel oil in Hong Kong for power generation and road transport was reduced to 0.5% or less by weight. This change was implemented over a single weekend. Ambient levels of both SO_2 and SO_4 were compared with indices of respiratory health in children living in two areas of Hong Kong that differed markedly in levels of ambient air pollution. After the reduction in fuel sulphur content, ambient levels of both SO_2 and SO_4 fell markedly in the more polluted area (80% and 38% respectively) but major changes were not observed in the less polluted areas. The improvement in air quality was associated with larger declines in respiratory symptoms among primary-school children in the more-polluted area than in the less-polluted area (Peters et al., 1996). The regulations were also associated with declines (2.1%, 2.0% and 3.9%) in the upward trend of annual average natural cause, cardiovascular and respiratory disease mortality, respectively, compared to the previous 5 years. Annual rates of mortality due to other causes were not markedly affected. Reductions in mortality were more pronounced in the most highly polluted areas. Hedley et al. (2002) estimated that the regulation resulted in gains in average life expectancy per year of exposure to the lower levels of air pollution: 20 days for women, 41 days for men.

Natural experiments

A widely cited example is the closing of the Provo, Utah steel mill for one year in 1996. Pope et al. (1996) studied mortality and morbidity associated with changes in particle concentrations and characteristics before, during and after the year in which the plant was not operating. The mill contributed approximately 50% of the Valley's respirable PM. PM concentrations during winters when the steel mill was open were nearly double those when it was closed. Elevated PM levels were associated with increase hospitalizations for pneumonia, asthma, bronchitis and pleurisy. Hospital admissions of children due to respiratory disease were two to three times higher during the years

[6] See background paper by Rabl et al. in this issue for further discussion of cost-benefit analysis of air quality policies

the mill was open. More recent toxicologic analyses of extracts collected from filters used to monitor PM levels in the Utah Valley show that the extracts caused increases in oxidant generation, airway inflammation, cytokine release and other indicators of lung injury. These effects were larger for particles collected when the steel mill was operating than when it was closed. The monograph notes that such studies however will never stand alone as accountability assessments as other factors could confound the observed effect. Also the observed reduction of PM concentration may have little relevance to the magnitude of concentration reductions that would result from intentional regulatory efforts to improve air quality.

What measures of risk should be used in the estimation of public health benefits of air pollution controls?

A critical step in estimating the benefits of proposed air pollution policies is the appropriate selection of adverse health outcomes and concentration response functions describing the quantitative association between ambient air pollution levels and the corresponding health effects. The findings of epidemiological studies provide the scientific basis for these decisions. Because of the importance of estimates of health benefits in decision-making and concerns about the methods used by EPA, the U.S. Senate directed EPA to request that the National Research Council (NRC) evaluate methods used to derive health benefits estimates and make recommendations on best practices for these assessments. The NRC Committee on Estimating the Health-Risk- Reduction Benefits of Proposed Air Pollution Regulations (NRC, 2002) undertook a review of EPA health benefits analyses conducted for rule making on PM and ozone standard setting, Tier 2 motor vehicle emissions standards and gasoline control requirements, heavy duty engine and vehicle standard and highway diesel fuel sulfur control requirements, and the 1990 Clean Air Act Amendments. The Committee considered the structure of the analyses and methodological issues regarding exposure assessment, the selection of health outcomes and the concentration response function, the analysis of uncertainty and the presentation of the methods and results.

The Committee identified the following issues regarding the selection and definition of mortality and morbidity outcomes:

- Clinically diagnosed illnesses, such as chronic bronchitis and asthma attacks are typically evaluated in benefits analyses. These outcomes cover a wide range of severity levels and time courses. For example, chronic bronchitis can range from a chronic cough to a severe chronic airway obstruction that requires long-term care. EPA needs to investigate and improve the methods used to reconcile differences between the severity of disease described in air pollution epidemiology and that commonly used to develop estimate of background disease prevalence and incidence.
- Mortality estimates tend to dominate the overall health benefits estimates when a dollar value is assigned to them. Data on morbidity is less comprehensive and needs to be improved, especially if the value assigned to mortality decreases and morbidity outcomes begin to play a more dominant role in the benefits analyses.
- A key assumption is that there is a causal association between types of air pollution and adverse health outcomes. The Committee identified two general issues pertaining to confounding that contribute to uncertainty in the interpretation of studies on air pollution health effects. First, air pollutants often have a common source and are subject to similar atmospheric dispersion processes. Concentrations of pollutants tend to be correlated, and their independent effects are often difficult to disentangle using multi pollutant models. PM is a heterogeneous mixture, and the component responsible for observed effects has not been determined. The second issue is the possibility that some unknown factor that has not been controlled for explains the association between ambient concentrations in air pollution and risk of health effects. EPA recommended that the evidence of causality be summarized to justify the inclusion or exclusion of health outcomes and to assess the uncertainty associated with the assumption of causality.

In its critique of selection of epidemiological studies selected by EPA for use in its benefits analyses the Committee offered the following assessment and recommendations:

- The selection of the American Cancer Society (ACS) cohort study (Pope et al., 1995) for the evaluation of PM-related premature mortality was reasonable as cohort studies provide a more complete assessment of health effects than time-series studies because they include long-term cumulative effects of air pollution. Cohort studies

also provide data to estimate the number of life-years lost in a population, not just the number of lives lost, thus allowing for several valuation methods to be used.

- EPA used U.S. studies to provide data to estimate health benefits. Data for many health outcomes in the U.S. studies are restricted to a specific age group. For example, the data for hospital admissions apply to persons 65 years or older, primarily because the data come from Medicare databases. EPA should use the findings of recent studies conducted outside the U.S. that provide information on health outcomes with broader age ranges and on outcomes not currently evaluated, such as levels of use of the primary care system. The Committee encourages EPA to estimate and report benefits by age, sex and other demographic factors when possible.
- The Committee recommended that EPA thoroughly review the selection of the best estimate for long-term effects of air pollution on mortality given the reanalyses of the ACS study and Harvard six cities study (Krewski et al., 2000), the recent extended analysis of the original ACS study (Pope et al., 2002), the availability of a third cohort study (Abbey et al., 1999; McDonnell et al., 2000), and the publication of the first European cohort study (Hoek et al., 2002). The Committee recommended that EPA consider derivation of a weighted-mean estimate from the cohort studies following review of the entire database. The inclusion or exclusion of studies and the weighting scheme should be justified and clearly explained.
- EPA used time-series studies to derive benefits estimates of selected morbidity outcomes for its primary analyses. The Committee concluded that consideration of these studies is appropriate to estimate acute effects from short term exposure, however a thorough investigation of problems associated with the use of S-plus statistical software to fit generalized additive models (GAM) to the data (HEI, 2003) is required to determine the effect on the coefficients used in the benefits estimates.
- To evaluate short term effects of air pollution, EPA should use concentration-response functions from studies that integrate over several days or weeks the exposure period and the time period of the event, rather than those that restrict these time periods to 1 or 2 days.

The Committee emphasized the large amount of uncertainty inherent in health benefit analyses that is rooted in incomplete scientific knowledge and concluded its report with the following recommendations to improve the transparency of assumptions and the estimation and communication of uncertainty.

- Health benefits analyses should reflect the plausibility and uncertainty of the concentration response function, such as imprecision of exposure and response measures, potential confounding factors, and extrapolation from the study population to the target population. EPA should strive to present the results of its health benefits analyses in ways that avoid conveying an unwarranted degree of certainty, including placing less emphasis on single numbers and more emphasis on ranges.
- EPA should document clearly the rationale for its selection of studies and concentration-response coefficients as these choices require judgment on the part of the analyst.
- EPA should evaluate threshold assumptions in a consistent and transparent framework using several alternative assumptions in the formal uncertainty analysis.

What scientific evidence is needed for development of location specific standards?

At present, air pollution control strategies are based largely on national ambient air quality objectives, taking into account both health implications of pollution exposure and the cost of pollution mitigation (Raizenne, 2003). As we work towards the long-term goal of reducing ambient air pollution, there may be opportunities for more cost-effective pollution mitigation actions in some areas as compared to others. The development of location specific air pollution standards and guidelines requires local air pollution monitoring data, as well as information on mitigation options and costs in those areas. This information may be used to design cost-effective location specific air pollution control programs that will contribution to national and international air pollution control objectives.

What scientific evidence is lacking to guide air pollution control strategies?

Progress in refining policy directions for air quality risk management will require additional scientific information in a number of areas (Maynard, 2003). Maynard et al. (2003) noted that research is needed on indicators of population health impacts, individual exposure and outcome studies, identification of sensitive subpopulations, effects of

individual pollutants on population health, comparative risk assessment, characterization of uncertainty in risk estimates, effectiveness of policy interventions, predictions of the benefits; targeted interventions, and effective stakeholder engagement in risk management decisions.

Here, we focus on the following four overarching issues related to the NRC's framework for integrating particulate matter research.

i) assessing the toxicity of components of ambient air pollution

Further research is needed to identify the toxic components of the complex mixture of ambient air pollutants to which we are exposed, particularly to clarify the role of particulate and gaseous co-pollutants. Observational studies are limited by joint exposure to multiple pollutants, although a national study with sufficient variation in the composition of the pollutant mix may afford an opportunity to disentangle the effects of multiple pollutant exposures. Toxicological studies can be used to systematically study the effects of individual pollutants, joint exposures, and interactions among pollutants using factorial experimental designs. Controlled clinical studies may also be used to investigate these same effects at low exposure levels, following research ethics board guidelines.

Specific studies are also needed to better define the characteristics of particulate air pollution that mediate particle toxicity. Particle size appears to be correlated with toxicity, with smaller particles that can penetrate deeper into the airways demonstrating stronger associations with adverse health outcomes. In addition to particle mass and number, the role of toxic substances and biologically active agents affixed to particles also require further investigation. A deeper understanding of the toxic constituents of particulate air pollution will provide a firmer basis for the design of air quality standards and guidelines for particulate air pollution.

ii) effects of long-term exposure

Virtually all of the current evidence on the effects of long-term exposure to particulate and gaseous co-pollutants derives from three large cohorts: the Harvard six cities cohort (Dockery et al., 1993), the American Cancer Society cohort (Pope et al., 1995, 2002), and the Seventh Day Adventist cohort (Abbey et al., 1999). While providing consistent evidence of effects of an association between particulate air pollution and mortality, these cohorts are subject to certain limitations relating to exposure misclassification due to the use of fixed site ambient monitors for exposure assessment (Mallick et al., 2002) and adequacy of adjustment for important covariates such as socioeconomic status and co-morbidity. Of these three studies, only the Harvard six cities study was originally designed to examine the health risks of long-term exposure to ambient air pollution. All three studies focus primarily on mortality rather than morbidity; none of these three studies focuses on children.

Although further follow-up of all three cohorts will provide additional information on the adverse health outcomes associated with long-term exposure to particulate and gaseous pollutants, a prospective cohort study with more detailed exposure information would afford an opportunity to better characterize the population health impacts of ambient air pollution (Brunekreef, 2003; White & Suh, 2003). With regular monitoring of study subjects, a prospective study of this type could provide detailed information on morbidity associated with long-term exposure to air pollution, as well as mortality. Such a study could be linked to the ongoing supersites air quality monitoring program, which provides the most detailed information on ambient air quality collected to date. Personal exposure monitoring for some subjects would also help to provide more refined measures of exposure (Chang et al., 2003). Linking such an initiative to the proposed longitudinal study of children's health and the environment would afford an opportunity to examine health effects in a potentially susceptible population subgroup (Wigle, 2003).

iii) mechanisms of cardiovascular and respiratory mortality

Cardiovascular and respiratory mortality represent health outcomes that have been consistently associated with exposure to ambient air pollution. An important consideration in assessing the weight of evidence on the health effects of air pollution is a plausible mechanistic pathway by which pollutants may exert their adverse effects.

In recent years, a cohesive biological picture of particulate toxicity has begun to emerge, with accelerated atherosclerosis representing a plausible pathway for both morbidity and mortality. Further research is needed to clarify these initial results, along with parallel work on mechanistic pathways that may apply to gaseous co-pollutants. Collectively, this information will help to identify the toxic components of the complex mixture of air pollutants to which we are exposed.

iv) one atmosphere approach

Regulators are faced with a number of challenges in setting air quality objectives for individual pollutants. Uncertainties in source apportionment make it difficult to predict the impact of specific emissions controls on air quality. With the high degree of collinearity among criteria air pollutants, consideration might be given to air pollution control strategies that focus on the atmosphere as a whole, rather than emphasizing one pollutant over another. Such a "one atmosphere" approach would emphasize reduction in air pollution in general, with the expectation of achieving significant reductions in the concentrations of individual pollutants in ambient air. The adoption of this approach would reduce the need for more detailed scientific information on the adverse health effects of individual pollutants, but would require a greater commitment to the control of air pollution in general.

Acknowledgement

This paper was prepared as a background paper for discussion at the third meeting in the NERAM Air Quality and Health Colloquium Series. The Colloquium is sponsored by Canadian Petroleum Products Institute, Health Canada, Ontario Ministry of Environment, US EPA, and Shell International Petroleum Company Limited. Jonathan Samet is Co-Director of the Risk Sciences and Public Policy Institute, and Professor and Chair of the Department of Epidemiology at the Johns Hopkins Bloomberg School of Public Health. Daniel Krewski is NSERC/SSHRC/McLaughlin Chair in Population Health Risk Assessment at the University of Ottawa. The authors are grateful to Daniel Rainham of the McLaughlin Centre for Population Health and Lorraine Craig of NERAM for their assistance in preparing this manuscript.

REFERENCES

Abbey, D.E., Nishino, N., McDonnel, W.F., Burchette, R.J., Knutsen, S.F., Beeson, W.L., and Yang, J.X. 1999. Long-term inhalable particles and other air pollutants related to mortality in nonsmokers. *Am. J. Respir. Crit. Care* 159:373-382.

Anderson, H.R., Atkinson, R.W., Bremner, S.A., and Marston, L. 2003. Particulate air pollution and hospital admissions for cardiorespiratory diseases: are the elderly at greater risk? *Eur. Respir. J.* 21:39S-46S.

Atkinson, R.W., Anderson, H.R., Sunyer, J., Ayres, J., Baccini, M., Vonk, J.M., Boumghar, A., Forastiere, F., Forsberg, B., Touloumi, G., Schwartz, J., and Katsouyanni, K. 2001. Acute effects of particulate air pollution on respiratory admissions - Results from APHEA 2 project. *Am. J. Respir. Crit. Care* 164:1860-1866.

Brook, R.D., Brook, J.R., Urch, B., Vincent, R., Rajagopalan, S., and Silverman, F. 2002. Inhalation of fine particulate air pollution and ozone causes acute arterial vasoconstriction in healthy adults. *Circulation* 105:1534-1536.

Brunekreef, B. 2003. Design of cohort studies for air pollution effects. *J. Toxicol. Env. Health A* 66:1723-1734.

Burnett, R.T., Dales, R., Krewski, D., Vincent, R., Dann, T., and Brook, J.R. 1995. Associations between ambient particulate sulfate and admissions to Ontario hospitals for cardiac and respiratory diseases. *Am. J. Epidemiol.* 142:15-22.

Burnett, R.T., Brook, J.R., Yung, W.T., Dales, R.E., and Krewski, D. 1997a. Association between ozone and hospitalization for respiratory diseases in 16 Canadian cities. *Environ. Res.* 72:24-31.

Burnett, R.T., Dales, R.E., Brook, J.R., Raizenne, M.E., and Krewski, D. 1997b. Association between ambient carbon monoxide levels and hospitalizations for congestive heart failure in the elderly in 10 Canadian cities. *Epidemiology* 8:162-167.

Burnett, R.T., Smith-Doiron, M., Stieb, D., Çakmak, S., and Brook, J.R. 1999. Effects of particulate and gaseous air pollution on cardiorespiratory hospitalizations. *Arch. Environ. Health* 54:130-139.

Burnett, R.T., Brook, J., Dann, T., Delocla, C., Philips, O., Çakmak, S., Vincent, R., Goldberg, M.S., and Krewski, D. 2000. Association between particulate and gas phase components of urban air pollution and daily mortality in eight Canadian cities. *Inhal. Toxicol.* 12:15-39.

Burnett, R.T., Smith-Doiron, M., Stieb, D., Raizenne, M.E., Brook, J.R., Dales, R.E., Leech, J.A., Çakmak, S., and Krewski, D. 2001. Association between ozone and hospitalization for acute respiratory diseases in children less than 2 years of age. *Am. J. Epidemiol.* 153:444-452.

Çakmak, S., Burnett, R.T., Jerrett, M., Goldberg, M.S., Pope, A., Ma, R., Gultekin, T., Thun, M.J., and Krewski, D. 2003. Spatial regression models for large-cohort studies linking community air pollution and health. *J. Toxicol. Env. Health A* 66:1811-1824.

Calderon-Garciduenas, L., Mora-Tiscareno, A., Fordham, L.A., Valencia-Salazar, G., Chung, C.J., Rodriguez-Alcaraz, A., Paredes, R., Variakojis, D., Villarreal-Calderon, A., Flores-Camacho, L., Antunez-Solis, A., Henriquez-Roldan, C., and Hazucha, M.J. 2003. Respiratory damage in children exposed to urban pollution. *Pediatr. Pulm.* 36:148-161.

Canada Wide Standards Development Committee for Particulate Matter and Ozone. 1999. Compendium of Benefits Information. 99-08-17.

Chang, L.-T., Koutrakis, P., Catalano, P.J., and Suh, H.H. 2003. Assessing the importance of different exposure metrics and time-activity data to predict 24-h personal $PM_{2.5}$ exposures. *J. Toxicol. Env. Health A* 66:1825-1846.

Clancy, L., Goodman, P., Sinclair, H., and Dockery, D.W. 2002. Effect of air-pollution control on death rates in Dublin, Ireland: An intervention study. *Lancet* 360:1210–1214.

Cohen, A.J., Krewski, D., Samet, J., and Willes, R. (Eds.). 2003. Health and Air Quality: Interpreting Science for Decision Makers. *J. Toxicol. Env. Health A* 66:1489-1903.

Devlin, R.B. Ghio, A.J., Kehrl, H., Sanders, G., and Cascio, W. 2003. Elderly humans exposed to concentrated air pollution particles have decreased heart rate variability. *Eur. Respir. J.* 21:76s-80s.

Dockery, D.W., Pope, C.A., Xu, X.P, Spengler, J.D., Ware, J.H., Fay, M.E., Ferris, B.G. and Speizer, F.E. 1993. An association between air pollution and mortality in six U.S. cities. *New Engl. J. Med.* 329:1753-1759.

Dominici, F., Zeger, S.L., and Samet, J.M. 2000. A measurement error model for time-series studies of air pollution and mortality. *Biostatistics* 1:157-175.

Dominici, F., McDermott, A., Daniels, M., Zeger, S.L., and Samet, J.M. 2003a. Revised Analyses of the National Morbidity, Mortality, and Air Pollution Study, Part II. Mortality Among Residents of 90 Cities. In Health Effects Institute. 2003. Revised Analyses of Time-Series Studies of Air Pollution and Health. Special Report. Health Effects Institute, Boston MA.

Dominici, F., McDermott, A., Zeger, S.L., and Samet, J.M. 2003b. Airborne particulate matter and mortality: time-scale effects in four US cities. *Am. J. Epidemiol.* 157:1055-1065.

Environmental Protection Agency (US). 1997. The Benefits and Costs of the Clean Air Act, 1970 to 1990. EPA-410-R-97-002. Office of Air and Radiation, Washington DC.

Environmental Protection Agency (US). 1999. The Benefits and Costs of the Clean Air Act 1990 to 2010: EPA Report to Congress. EPA-410-R-99-001. Office of Air and Radiation, Washington DC.

Frampton, M.W. 2001. Systematic and cardiovascular effects of airway injury and inflammation: Ultrafine particle exposure in humans. *Environ. Health Perspect.* 109:529-532.

Fung, K.Y., Krewski, D., Chen, Y., Burnett, R., and Cakmak, S. 2003. Comparison of time series and case-crossover analyses of air pollution and hospital admission data. *Int. J. Epidemiol.* 32:1064-1070.

Fung, K.Y., Krewski, D., Burnett, R. T., and Dominici, F. 2004. Testing the harvesting hypothesis by time domain regression analysis. *J. Toxicol. Env. Health A.* To appear.

Gent, J.F., Triche, E.W., Holford, T.R., Belanger, K., Bracken, M.B., Beckett, W.S., and Leaderer, B.P. 2003. Association of low-level ozone and fine particles with respiratory symptoms in children with asthma. *J. Am. Med. Assoc.* 290:1859-1867.

Goldberg, M.S., and Burnett, R.T. 2004. A new longitudinal design for identifying subgroups of the population who are susceptible to the short-term effects of ambient air pollution. *J. Toxicol. Env. Health A.* To appear

Greenbaum, D.S. 2003. A historical perspective on the regulation of particles. *J. Toxicol. Env. Health* 66:1493-1498.

Greenbaum, D.S., Bachmann, J.D., Krewski, D., Samet, J.M., White, R., and Wyzga, R. E. 2001. Particulate air pollution standards and morbidity and mortality: Case study. *Am. J. Epidemiol.* 154:78S-90S.

Health Effects Institute (HEI). 2003a. Revised Analyses of Time-Series Studies of Air Pollution and Health Revised Analyses of the National Morbidity, Mortality, and Air Pollution Study, Part II Revised Analyses of Selected Time-Series Studies preprintversion. http://www.healtheffects.org/Pubs/TimeSeries.pdf.

Health Effects Institute (HEI). 2003b. Assessing Health Impact of Air Quality Regulations: Concepts and Methods for Accountability Research. Communication 11. HEI Accountability Working Group. Health Effects Institute, Boston MA.

Hedley, A.J., Wong, C.M., Thach, T.Q., Ma, S., Lam, T.H., and Anderson, H.R. 2002. Cardiorespiratory and all-cause mortality after restrictions on sulphur content of fuel in Hong Kong: An intervention study. *Lancet* 360:1646–1652.

Hoek, G., Brunekreef, B., Goldbohm, S., Fischer, P., and van den Brandt, P.A. 2002. Association between mortality and indicators of traffic-related air pollution in the Netherlands: a cohort study. *Lancet.* 360:1203-1209.

Hoover, B.K., Foliart, D.E., White, W.H., Cohen, A.J., Calisti, L.J., Krewski, D., and Goldberg, M.S. 2003. Retrospective data quality audits of the Harvard six cities and American Cancer Society studies. *J. Toxicol. Env. Health A* 66:1553-1562.

Jang, A.S., Yeum, C.H., and Son, M.H. 2003. Epidemiologic evidence of a relationship between airway hyperresponsiveness and exposure to polluted air. *Allergy* 58:585-588.

Jerrett, M., Burnett, R.T., Goldberg, M.S., Sears, M., Krewski, D., Catalin, R., Kanaroglou, P., Giovis, C., and Finkelstein, N. 2003a. Spatial analysis for environmental health research: concepts, methods and examples. *J. Toxicol. Env. Health A* 66:1783-1810.

Jerrett, M., Burnett, R.T., Willis, A., Krewski, D., Goldberg, M.S., DeLuca, P., and Finkelstein, N. 2003b. Spatial analysis of the air pollution-mortality relationship in the context of ecologic confounders. *J. Toxicol. Env. Health A* 66:1735-1778.

Katsouyanni, K., Touloumi, G., Spix, C., Schwartz, J., Balducci, F., Medina, S., Rossi, G., Wojtyniak, B., Sunyer, J., Bacharova, L., Schouten, J.P., Ponka, A., and Anderson, H.R. 1997. Short-term effects of ambient sulphur dioxide and particulate matter on mortality in 12 European cities: Results from time series data from the APHEA project. *Brit. Med. J.* 314:1658–1663.

Katsouyanni, K., Samet, J., Cohen, A., Anderson, H.R., Atkinson, R., Burnett, R.T., Dominici, F., Krewski, D., Le Tertre A., Medina, S., Schwartz, J., Touloumi, G., Zanobetti, A, and Zeger, S. 2002. Air pollution and Health: a Combined European and North American Approach (APHENA). Poster presentation. 1st Annual AIRNET Conference, December 11-12, 2002. London, U.K.

Katsouyanni, K., Touloumi, G., Samoli, E., Petasakis, Y., Analitis, A., Le Tertre, A., Rossi, G., Zmirou, D., Ballester, F., Boumghar, A., Anderson, H., Wojtyniak, B., Paldy, A., Braunstein, R., Pekkanen, J., Schindler, C., and Schwartz, J. 2003. Sensitivity Analysis of Various Models of Short-Term Effects of Ambient Particles on Total Mortality in 29 Cities in APHEA2. In Health Effects Institute. 2003. Revised Analyses of Time-Series Studies of Air Pollution and Health. Special Report. Health Effects Institute, Boston MA.

Koenig, J.Q., Larson, T.V., Hanley, Q.S., Robelledo, V., Dumler, K., Checkoway, H., Wang, S.Z., Lin, D., and Pierson, W.E. 1993. Pulmonary function changes in children associated with fine particulate matter. *Environ. Res.* 63:26-39.

Krewski, D., Burnett, R.T., Goldberg, M.S., Hoover, K., Siemiatycki, J., Jerrett, M., Abrahamowicz, M., and White, W.H. 2000. Reanalysis of the Harvard Six Cities study and the American Cancer Society study of particulate air pollution and mortality. A special report of the Institute's Particle Epidemiology Reanalysis Project. Cambridge, MA: Health Effects Institute.

Krewski, D., Burnett, R.T., Goldberg, M.S., Hoover, K., Siemiatycki, J., Jerrett, M., Abrahamowicz, M., and White, W.H. 2003. Overview of the re-analysis of the Harvard six cities study and American Cancer Society study of particulate air pollution and mortality. *J. Toxicol. Env. Health A* 66:1507-1552.

Krewski, D., Burnett, R.T., Jerrett, M., Pope, A., Rainham, D.G., Calle, E.E., Thurston, G.D., and Thun, M. 2004. Mortality associated with long-term exposure to ambient air pollution: ongoing analyses based on the American Cancer Society cohort. *J. Toxicol. Env. Health A.* To Appear.

Kunzli, N., Kaiser, R., Medina, S., Studnicka, M., Chanel, O., Filliger, P., Herry, M., Horak, F., Puybonnieux-Texier, V., Quenel, P., Schneider, J., Seethaler, R., Vergnaud, J.C., and Sommer, H. 2000. Public-health impact of outdoor and traffic-related air pollution: a European assessment. *Lancet* 356:795-801.

Le Tertre, A., Medina, S., Samoli, E., Forsberg, B., Michelozzi, P., Boumghar, A., Vonk, J.M., Bellini, A., Atkinson, R., Ayres, J.G., Sunyer, J., Schwartz, J., and Katsouyanni, K. 2002. Short-term effects of particulate air pollution on cardiovascular diseases in eight European cities. *J. Epidemiol. Commun. Health* 56:773-779.

Lin, M., Chen, Y., Burnett, R.T., Villeneuve, P.J., and Krewski, D. 2002. The influence of ambient coarse particulate matter on asthma hospitalization in children: Case-crossover and time-series analyses. *Environ. Health Perspect.* 110:575-581.

Lin, M., Chen, Y., Burnett, R.T., Villeneuve, P.J., and Krewski, D. 2003a. Effect of short-term exposure to gaseous pollution on asthma hospitalization in children: a bi-directional case-crossover analysis. *J. Epidemiol. Commun. Health* 57:50-55.

Lin, M., Chen, Y., Villeneuve, P.J., Burnett, R.T., Lemyre, L., Hertzman, C., McGrail, K., and Krewski, D. 2003b. Gaseous air pollutants and asthma hospitalization of children with low household income in Vancouver, British Columbia, Canada. *Am. J. Epidemiol.* 159:294-303.

Linn, W.S., Szlachcic, Y., Gong, H., Kinney, P.L., and Berhane, K.T. 2000. Air pollution and daily hospital admissions in metropolitan Los Angeles. *Environ. Health Perspect.* 108:427-434.

Liu, S., Krewski, D., Shi, Y., Chen, Y., and Burnett, R.T. 2003. Association between gaseous air pollutants and adverse pregnancy outcomes in Vancouver, Canada. *Environ. Health Perspect.* 111:1773-1778.

Ma, R., Krewski, D., and Burnett, R.T. 2003. Random effects Cox models: A Poisson modelling approach. *Biometrika* 90:157-169.

Mallick, R., Fung, K., and Krewski, D. 2002. Adjusting for measurement error in the Cox proportional hazards regression model. *J. Cancer Epidemiol Prev*. 7:155-164.

Maynard, R. 2003. Scientific information needs for regulatory decision making. *J. Toxicol. Environ. Health A.* 66:1499-1501.

Maynard, R., Krewski, D., Burnett, R.T., Samet, J., Brook, J.E., Granville, G., and Craig, L. 2003. Health and air quality: Directions for policy-relevant research. *J. Toxicol. Environ. Health A.* 66:1891-1904.

McDonnell, W.R., Nishino-Ishikawa, N., Petersen, F.F., Chen. J.H., and Abbey, D.E. 2000. Relationships of mortality with the fine and coarse fractions of long-term ambient PM_{10} concentrations in nonsmokers. *J. Expo. Anal. Environ. Epidemiol*. 10:427-436.

National Research Council (NRC). 1998. Committee on Research Priorities for Airborne Particulate Matter. Research Priorities for Airborne Particulate Matter: No. 1. Immediate priorities and a long-range research portfolio. Washington, DC: National Academy Press.

National Research Council (NRC). 1999.Committee on Research Priorities for Airborne Particulate Matter. Research Priorities for Airborne Particulate Matter: II. Evaluating Research Progress and Updating the Portfolio. Washington, DC: National Academy Press.

National Research Council (NRC), 2000. Acute Exposure Guideline Levels for Selected Airborne Chemicals, Volume 1. 1-201. 2000. Washington, DC, National Academy Press.

National Research Council (NRC), 2001. Committee on Research Priorities for Airborne Particulate Matter. Research priorities for airborne particulate matter III. Early research progress, National Academy Press, Washington, DC.

National Research Council (NRC). 2002. Acute Exposure Guideline Levels for Selected Airborne Contaminants, Volume 2. 1-276. 2002. Washington, DC, National Academy Press.

National Research Council (NRC). 2003. Acute Exposure Guideline Levels for Selected Airborne Contaminants, Volume 3. 2003. Washington, DC, National Academy Press.

National Research Council (NRC). 2004. Air Quality Management in the United States. Committee on Air Quality Management in the United States. January 2004. 1-314. Washington, DC, National Academy Press.

Oftedal, B., Nafstad, P., Magnus, P., Bjorkly, S., and Skrondal, A. 2003. Traffic related air pollution and acute hospital admission for respiratory diseases in Drammen, Norway 1995-2000. *Eur. J. Epidemiol.* 18:671-675.

Peters, J., Hedley, A.J., Wong, C.M., Lam, T.H., Ong, S.G., Liu, J., and Spiegelhalter, D.J. 1996. Effects of an ambient air pollution intervention and environmental tobacco smoke on children's respiratory health in Hong Kong. *Int. J. Epidemiol.* 25:821–828.

Peters, A., Liu, E., Verrier, R.L., Schwartz, D.R., Mittleman, M., Baliff, J., Oh, J.A, Allen, G., Monahan, K., and Dockery, D.W. 2000. Air pollution and incidences of cardiac arrhythmia. *Epidemiology* 11:11-17.

Pope C.A. 1989. Respiratory disease associated with community air pollution and a steel mill, Utah Valley. *Am. J. Pub. Health* 79:623–628.

Pope, C.A., Thun, M.J., Namboodiri, M.M., Dockery, D.W., Evans, J.S., Speizer, F.E., and Heath Jr, C.W. 1995. Particulate air pollution as a predictor of mortality in a prospective study of U.S. adults. *Am. J. Respir. Crit. Care Med.* 151:669-674.

Pope, C.A. 1996. Particulate pollution and health: A review of the Utah Valley experience. *J. Expos. Anal. Environ. Epidemiol.* 6:23–34.

Pope, C.A., Burnett, R.T., Thun, M.J., Calle, E.E., Krewski, D., Ito, K., and Thurston, G.D. 2002. Lung cancer, cardiopulmonary mortality, and long-term exposure to fine particulate air pollution. *J. Am. Med. Assoc.* 287:1132-1141.

Pope, C.A., Burnett, R.T., Thurston, G.D., Thun, M.J., Calle, E., Krishnan, K., and Godleski, J.J. 2003. Cardiovascular mortality and long-term exposure to particulate air pollution: Epidemiological evidence of general pathophysiological pathways of disease. *Circulation* 109:71-77.

Rabl, A., Nathwani, J., Pandey, M., and Hurley, F. 2005. Tools and strategies for improving policy responses to the risk of air pollution. *J. Toxicol. Environ. Health.* This issue.

Raizenne, M. 2003. Science and regulation – U.S. and Canadian overview. *J. Toxicol. Environ. Health A.* 66:1503-1506.

Ramsay, T., Burnett, R.T., and Krewski, D. 2003a. Exploring bias in a generalized additive model for spatial air pollution data. *Environ. Health Perspect.* 111:1283-1288.

Ramsay, T., Burnett, R.T., and Krewski, D. 2003b. The effect of concurvity in generalized additive models linking mortality to ambient particulate matter (with discussion). *Epidemiology* 14:18-23.

Reid, N., Misra, P.K., Amann, M., and Hales, J. 2005. Air quality modelling for policy development. *J. Toxicol. Environ. Health.* This issue.

Salehi, F., Krewski, D., Mergler, D., Normandin, L., Kennedy, G., Philippe, S., and Zayed, J. 2003. Bioaccumulation and locomotor effects of manganese phosphate/sulfate mixture in Sprague-Dawley rats following subchronic (90 days) inhalation exposure. *Toxicol. Appl. Pharmacol.* 191:264-271.

Samet, J.M., Dominici, F., Curriero, F., Coursac, I., and Zeger, S.L. 2000. Particulate air pollution and mortality: findings from 20 U.S. cities. *N. Engl. J. Med.* 343:1742-1757.

Schwartz, J. 1999. Air pollution and hospital admissions for heart disease in eight U.S. counties. *Epidemiology* 10:1-4.

Schwartz, J. 2000. Harvesting and long term exposure effects in the relation between air pollution and mortality. *Am. J. Epidemiol.* 151:440-448.

Stieb, D.M., Judek, S., and Burnett, R.T. 2002a. Meta-analysis of time-series studies of air pollution and mortality: effects of gases and particles and the influence of cause of death, age, and season. *J. Air Waste Manage. Assoc.* 52:470-484.

Stieb, D.M., Smith-Doiron, M., Brook, J.R., Burnett, R.T., Dann, T., Mamedov, A., and Chen, Y. 2002b. Air pollution and disability days in Toronto: Results from the National Population Health Survey. *Environ. Res.* 89:210-219.

Sunyer, J., Atkinson, R., Ballester, F., Le Tertre, A., Ayres, J.G, Forastiere, F., Forsberg, B., Vonk, J.M., Bisanti, L., Anderson, R.H., Schwartz, J., and Katsouyanni, K. 2003. Respiratory effects of sulphur dioxide: a hierarchical multicity analysis in the APHEA 2 study. *Occup. Environ. Med.* 60:2e.

UK Department of Health (UK). 2001. Statement and Report on Long-Term Effects of Particles on Mortality. HMSO, London, England. www.doh.gov.uk/comeap/ state.htm#state.

Utell, M.J., Frampton, M.W., Zareba, W., Devlin, R.B., and Cascio, W.E. 2002. Cardiovascular effects associated with air pollution: Potential mechanisms and methods of testing. *Inhal. Toxicol.* 14:1231-1247.

Villeneuve, P.J., Goldberg, M.S., Krewski, D., Burnett, R.T., and Chen, Y. 2002. Fine particulate air pollution and all-cause mortality within the Harvard Six-cities Study: Variations in risk by period of exposure. *Ann. Epidemiol.* 12:568-576.

Villeneuve, P.J., Burnett, R.T., Shi, Y., Krewski, D., Goldberg, M.S., Hertzman, C., Chen, Y., and Brook, J. 2003. A time series study of air pollution, socioeconomic status, and mortality in Vancouver, Canada. *J. Expo. Anal. Environ. Epidemiol*. 13: 427-435.

Vincet, R., Jumarathasan, P., Mukherjee, B., Gravel, C., Bjarnason, S., Urch, B., Speck, M., Brook, J., Tarlo, S., Zimmerman, B., and Siverman, F. 2001. Exposure to urban particles ($PM_{2.5}$) causes elevation of the plasma vasopeptides endothelin (ET)-1 and ET-3 in humans. [Abstract]. *Am. J. Respir. Crit. Care Med.* 163:A313.

White, W.H. and Suh, H.H. 2003. Monitoring exposure to ambient air pollutants. *J. Toxicol. Environ. Health A.* 66:1879-1882.

Wigle, D.T. 2003. *Children's Health and the Environment*. Oxford: Oxford University Press.

Willis, A.J., Jerrett, M., Burnett, R.T., and Krewski, D. 2003. The association between sulfate air pollution and mortality at the county scale: An exploration of the impact of scale on a long-term exposure study. *J. Toxicol. Environ. Health A.* 66:1605-1624.

World Health Organization (WHO). 2002. World Health Report. Geneva: World Health Organization.

Yang, Q., Chen, Y., Shi, Y., Burnett, R. T., McGrail, K., and Krewski, D. 2003. Association between ozone and respiratory admissions among children and the elderly in Vancouver, Canada. *Inhal. Toxicol.* 15:101-112.

Zanobetti, A., Wand, M.P., Schwartz, J., and Ryan, L.M. 2000. Generalized additive distributed lag models: quantifying mortality displacement. *Biostatistics* 1:279-292.

Zanobetti, A., and Schwartz, J. 2002. Cardiovascular damage by airborne particles: Are diabetics more susceptible? *Epidemiology* 13:588-592.

Zeger, S., Dominici, F., and Samet, J.M. 1999. Harvesting-resistant estimates of pollution effects on mortality. *Epidemiology* 10:171-175.

AMBIENT AIR POLLUTION AND POPULATION HEALTH: OVERVIEW OF HEALTH EFFECTS POSTERS PRESENTED AT THE 2003 AIRNET/NERAM CONFERENCE

Daniel Krewski, Daniel Rainham[1]

ABSTRACT

In November 2003 approximately 200 researchers, stakeholders and policy makers from more than 40 countries gathered to discuss the science and policy implications of air pollution and human health as part of the AIRNET/NERAM Strategies for Clean Air and Health initiative. The purpose of this paper is to review the more than 35 research posters presented at the conference, including exposure, toxicological and epidemiological studies of air pollution. Collectively, these papers support previous evidence that both short- and long-term exposures to particulate air pollution have adverse population health impacts, including effects on children. Cellular studies also suggest that air pollution can cause mutagenic and oxidative effects, raising concerns about carcinogenicity and cellular regeneration. Studies of biomarkers, such as Clara cell proteins and lymphocyte damage assessment, provide further evidence of air pollution effects at the cellular level. Other studies have focused on improvements to measurement and sources of air pollution. These studies suggest that particle mass rather than particle composition may be a more useful indicator of potential human health risk. It is well known that emissions from transportation sources are a major contributor to ambient air pollution in large urban centres. Epidemiologic researchers are able to reduce bias due to misclassification and improve exposure assessment models by allocating air pollution exposure according to distance from traffic sources or land-use patterns. The close association between traffic patterns and air pollution concentrations provides a potential basis for the development of transport policies and regulations with population health improvements as a primary objective. The results of the research presented here present opportunities and challenges for the development of policies for improvements to air quality and human health. However, there remains the challenge of how best to achieve these reductions.

[1] McLaughlin Centre for Population Health Risk Assessment, University of Ottawa, Ottawa, ON Canada K1N 6N5

INTRODUCTION

In November 2003 approximately 200 researchers, stakeholders and policy makers from more than 40 countries gathered to discuss the science and policy implications of air pollution and human health as part of the AIRNET/NERAM Strategies for Clean Air and Health initiative. AIRNET is a European-funded thematic network on pollution and health. NERAM is a Canadian Network for Environmental Risk Assessment and Management that fosters the integration of scientific knowledge and expertise to support efficient environmental protection practices and decision-making. The purpose of this paper is to review the more than 35 research posters presented at the AIRNET/NERAM conference. The studies have been organized into the following areas: studies of air pollution exposure, studies of the effects of air pollution on human and animal toxicology, epidemiological studies, and studies providing valuable insights using alternative methodologies and approaches. We conclude with a summary of the research findings and speculate on some of the implications for the development of policy to prevent population health impacts from air pollution exposures.

EXPOSURE ASSESSMENT

Sources of Variation

As part of the HEAPSS-project (Health Effects of Air Pollution on Susceptible Subpopulations), Aalto et al. (2003) used an ultrafine particle database with information from five European cities, to quantify the risk of hospitalization and of death due to air pollution. Particle measurement sites were selected to represent urban background concentrations. Initial reviews of the particle data suggest an increase in ultrafine particle concentrations is associated with decreasing latitude in the winter months and with temperature inversion events. Aarnio et al.'s (2003) adjunct research on traffic-specific particle exposures improves estimates of the actual concentrations experienced while commuting in urban areas. Particle samples were drawn from locations specific to commuting by train, metro, bus, or by foot. Preliminary conclusions reveal significant differences between urban background and traffic-specific locations in terms of concentration exposures and particle size.

Particle Composition

Exposure studies have also focused on the hypothesis that the physiological reaction to particulates will vary depending on particle composition. Cassee et al. (2003) examined the qualitative differences among particles collected from different sites throughout Europe. Aside from the usual increase in particle concentrations during the winter months, the study also found significant contrasts in chemical composition as a function of location. For example, metal concentrations were higher in Rome, whereas higher concentrations of magnesium and vanadium were found from samples collected in Amsterdam. In a similar study, De Berardis et al. (2003) studied the composition and physico-chemical characteristics of particles less than 2.1 $\mu g/m^3$ in Rome using scanning electron microscopy (SEM) and ion chromatographic techniques. Initial conclusions indicate a predominance of particles originating from vehicular sources, except during winter when particles a released through combustion of heating oil or methane gas for heating purposes. Exposure to specific chemicals associated with carbonaceous particles will typically vary with climatic and transportation patterns.

Source Apportionment

Urban populations experience heavier exposures to traffic-related air pollution concentrations. Frequency of exposures increases with the density of traffic and with traffic congestion, a trend common in many large urban centers, and magnitude of exposure increases among higher density populations. However, it is interesting to note that a majority of exposure to traffic and combustion related by-products takes place indoors. Ilacqua and Jantunen (2003) reported that volatile organic compounds (VOCs) from tailpipe emissions comprise the largest contributor to personal exposure with more than two-thirds of the exposure taking place indoors. Transportation is also a source of particulate air pollution and is highly variable depending on weather conditions and traffic flow densities. In the United Kingdom, for example, research has shown that an average of 25% of particulate pollution originates from road transportation and that this figure rises to 75-80% on high pollution days (O'Connell and Matthews, 2003).

Temporal Trends

Epidemiological assessment of the population health risks from air pollution could not be performed without the construction of high quality, long-term data sets. Of particular interest are ultrafine particles as research is beginning to reveal the inverse relationship between particle size and risk to health. Particle concentrations are time and location dependent. For example, Marconi et al. (2003) found that ultrafine particle concentrations were much greater in winter than in summer, were highly correlated with hours of high traffic volumes, and were lower on weekends than on weekdays. They also found that proximity to high traffic locations plays an important role in determining exposure. Background sites characteristically have particle concentrations up to 50% lower than with sites located near high traffic areas. Also of population health significance is the development of methods for the derivation of long-term air pollution concentration series. Metz (2003) correlated source emission contribution data with information on available air quality data to develop a model useful for forecasting future particle emissions and ambient concentrations. These types of series will be useful for estimating the impact of population health interventions associated with technological improvements, transportation policies, and strategies employed to reduce human exposure, especially among susceptible populations.

HUMAN TOXICOLOGY

Biomarkers

Biomarkers are used to improve assessments of biological plausibility in the relationship between air pollutants and human health effects. A promising biomarker called Clara cell proteins have been identified as sensitive markers of increased permeability of the lung epithelial barrier and an increase in serum levels has been associated with ambient ozone concentrations. Ozone exposures under controlled laboratory conditions among healthy adults resulted in a significant increase in serum levels of Clara cell proteins up to six hours after initial exposure (Blomberg et al., 2003). However, the influence of serum baseline diurnal variations is uncertain and further research is warranted. Research has also focused on identification of specific biological reactions from exposure to fine particulates. Research has also shown that fine particles may have direct or indirect effects on blood coagulation processes and thus physiological impacts affecting cardiac function. To evaluate the biological effects of fine particles Tarrona et al. (2003) examined a cohort of healthy adults between the ages of 20 and 55 working in clean industrial environments, such as the semiconductor and pharmaceutical industries, and who live in urban areas. Markers of blood coagulation were assessed at three points – before the work shift, after the work shift and after exposure to outdoor ambient air during commuting or social and domestic routines – with the goal of providing a plausible biological explanation for the ecologic link between exposure to particulate air pollution and myocardial infarction.

Genotoxicity

Genotoxic chemicals are able to cause damage to DNA but do not inevitably lead to the creation of cancerous cells (PTCL, 2004). Inorganic and organic compounds which dominate the composition of fine particulate matter air pollution may cause genotoxic health effects although the extent of genotoxicity is not well known. Brits et al. (2003) collected fine particulate matter from urban and industrial sites in Belgium to evaluate the mutagenic and cytotoxic properties. Particle mass concentrations of both PM_{10} and $PM_{2.5}$ were significantly higher in the industrial area. Human alveolar epithelial cells were exposed *in vitro* to particulate matter for a 48 hour period to investigate toxicological potential. Although particles from both size-fractions showed no significant toxicity, the micronucleus frequencies in binucleated cells were significantly increased indicating potential mutagenetic effects.

Gábelová et al. (2003) assessed the genotoxicity of the organic components associated with particulate air pollution. The organic components of particulate matter were extracted from particulates collected at three sites from three countries. The organic components induced a seasonal and dose-dependent *in vitro* increase of DNA damage in human cells. Effects were most pronounced for coarse particles in the 2.5-10μm diameter range.

Guastadisegni et al. (2003) investigated the relationship between the chemical composition of particulate matter air pollution and toxicity. An analysis of 32 samples collected from a variety of European metropolitan and rural areas sought to assess the oxidative activity of particles. The investigators created a synthetic model of the respiratory tract

lining fluid and measured the depletion rates of ascorbate, urate and reduced glutathione following a four hour exposure of 50μg/ml particle concentration. Inflammatory response was measured by evaluating the release of arachidonic acid, tumour necrosis factor alpha and interleukin-6 from a macrophagic cell line. The oxidative potential of particulate matter air pollution seems strongly related to a capacity to induce arachidonic acid from the macrophages and can be explained by the bioavailability of iron specific to every particle. However, these relationships were only apparent in the coarse fraction.

The aim of the EXPAH studies (Effects of polycyclic aromatic hydrocarbons (PAHs) in environmental pollution on exogenous and endogenous DNA damage) is to evaluate the role of PAHs as a source of genotoxic activities of organic mixtures associated with particulate air pollution. The effect of DNA damage is monitored by determination of chromosome aberration, and susceptibility in populations is studied by investigating the effect of metabolic polymorphisms in carcinogen metabolism and DNA repair. A case-control design was adopted to evaluate genotoxocity in exposed and less-exposed groups in the cities of Prague, Kosice and Sofia. The exposed group consisted of policemen (as well as bus drivers in Sofia) and the control groups were volunteers who spent more than 90% of their time in indoor environments. The role of competing risks factors such as diet and other lifestyle factors was also evaluated.

As part of the EXPAH studies, Cebulska-Wasilewska et al. (2003) investigated cellular susceptibility to polycyclic aromatic hydrocarbons (PAHs). Damage to DNA was evaluated as a function of lymphocyte repair competence. No significant differences were detected between the referent and exposed groups although residual damage to DNA and kinetic repair is less efficient among those with greater exposure to PAHs. Lymphocyte repair rates were also lower for those with less education implying a possible role of education as an effect modifier in the relationship between exposure to PAHs and genetic injury. Studies of the potential for oxidative damage to DNA were also performed to investigate the differences between exposed and less-exposed groups. Some significant results were detected between cities although most conclusions were mixed (Singh et al., 2003). Sram et al. (2003) used DNA adducts analysis and a common molecular cytogenetic technique called fluorescence in situ hybridization (FISH) to assess genotoxic risk.

The results are of interest for two reasons. First, the FISH technique seems to be an efficient tool to detect chromosomal aberrations and thus useful as a sensitive biomarker of exposure to PAHs. Second, the analysis detected significantly increased FISH cytogenetic parameters in nonsmoking policemen and bus drivers (exposed group) indicating an association between increased exposure to particulate air pollution with PAHs and potential genotoxic effects.

Source Toxicity

Exposure to air pollution, and fine particulate pollutants, can originate from either point or line (mobile) sources. As opposed to typical epidemiological investigations designed to assess health risk within a population, toxicological studies will assess the level and variability of chemicals within the blood stream or serum to determine the level of exposure to a specific pollutant source. From these studies it is then assumed that these chemicals will lead to measurable health impacts or possible mortality. Individuals in a toxicological study are generally grouped according to level of exposure for which distance can be used as a suitable proxy.

For example, Fierens et al. (2003) conducted a study to quantify the amount of dioxins, polychlorinated biphenyls (PCBs) and heavy metals in the bloodstreams of residents living in the vicinity of a municipal solid waste incinerator. A comparison or control group living in an unpolluted area was used for statistical comparison and information on lifestyle, dietary and occupational characteristics was also collected. The study concluded that exposure to emissions from can increase the body burden of dioxin and PCBs but that the increase is dependent on the consumption of animal fat products from the local food chain. The proposed interaction between exposure from the point source and dietary intake provides an indication of the difficulty involved when assessing toxic effects of pollution in a localized ecosystem.

Point sources of pollution may also have significant impacts on highly susceptible subpopulations as well as in the population. Lolova et al. (2003) investigated the relationship between air pollution emissions from a smelter plant

and blood lead levels in children as well as in placenta, maternal and cord blood. Air quality and determination of ambient lead and cadmium levels were determined from three fixed monitoring stations. Blood lead levels were analyzed from children residing close to the monitoring sites and from a control group not exposed to a specific point source, as well as from 76 maternal/infant pairs. The investigators found that lead and cadmium concentrations in the vicinity of the smelter were above national guidelines suggesting that the population is at risk of increased exposure to these chemicals. The researchers also found that blood lead levels in children living within the smelter area were significantly higher than among children living outside the area. Furthermore, the results also indicate a significant relationship between abnormal birth outcomes (pre-term births) and significantly higher than normal lead levels in maternal and cord blood.

Mobile sources of air pollution such as particulate pollution from road traffic also pose serious health risks. Since much of the air pollution in urban areas arises from motor vehicle emissions, recent research has begun to take interest in the variation of health effects associated with distance from road traffic sources as well as the variation in particle toxicity associated with density of road traffic. Gerlofs-Nijland et al. (2003) investigated how the toxic potency of particulate pollution varied with levels of traffic exhaust emissions. Particles were assessed for their toxic potential and grouped according to particle size and reactivity. Tests of toxicity were performed *in vitro* (in rats) and *in vivo* (in humans) using indicators of ascorbate depletion, arachidonic acid release, interleukin-6, and damage to DNA (genotoxicity). The researchers found that toxicity varies with particle size although both have adverse health effects. They also found that particles from locations with high traffic areas induce more effects than those from lower traffic areas.

New research is also probing into the toxicological characteristics of antioxidants as potential protection from vehicle-related air pollution. de Burbure et al. (2003) investigated antioxidant status as measure by selenium levels and lung response to NO_2 in rats. Selenium normal and selenium deficient rats were differentially exposed to NO_2 concentrations ranging from normal concentrations to acute exposures. The conclusions suggest that antioxidants play an important protective role in respiratory response to both acute and long term challenges from air pollutants such as NO_2. The results also support previous research of clara cell proteins (CC16) as excellent markers of increased lung permeability. Ironically, selenium may interact with CC16 antioxidants resulting in decreased antioxidant production thus depriving the lung of its natural capacity to protect against foreign pollutants.

EPIDEMIOLOGY

Cohort Studies

Cohort study designs afford an opportunity to investigate the long-term health effects from exposure to air pollution. In North America, large cohort studies such as the Harvard Six Cities study (Dockery et al., 1993), the American Cancer Society studies (Pope et al., 1995; Krewski et al., 2003a, b) and the ongoing Adventist Smog and Health study (Abbey et al., 1999), have consistently demonstrated associations between long-term exposure to particulate matter and mortality and have played a pivotal role in the establishment of national air quality objectives in both the United States and Canada (Greenbaum et al., 2001). Many newly developed European cohort studies have adopted advanced spatial techniques and will make an important contribution to improving the estimation of human health risk from ambient air pollution.

The European Community Respiratory Health Survey (ECRHS) consists of a nine-year follow-up prospective survey of more than 13,000 adults including data from 29 research centres located in 14 countries. As part of the air pollution working group, Götschi et al. (2003) are currently investigating correlations between surrogates of exposure to air pollution across 21 European centres and evaluating the influence of air pollution sampler location on air pollution exposure assessment. The SAPALDIA 2 (Swiss Cohort Study on Air Pollution and Lung Diseases in Adults) Swiss cohort study, part of the ECRHS study, is assessing associations of long-term air pollution exposure with the 10-year course of respiratory and atopic symptoms, diseases, markers of respiratory health and of atopy and cardiovascular health among 9,651 adults. Participants have been followed for address changes and mortality since 1991 and 83% of the original cohort is available for continued study. A GIS-based approach was used to derive individual exposure assignments from traffic-related air pollution exposures while also taking into consideration historical ambient pollutant levels, participant mobility patterns (Bayer-Oglesby et al., 2003). Although results are

still forthcoming, the SAPALDIA 2 study constitutes a novel approach to the creation of a cohort from an original cross-sectional study and represents a novel approach for detecting the long-term health effects of low-level air pollution.

Cohorts have also been created to assess the impacts of air pollution on morbidity outcomes. A study called HEAPSS (Health Effects of Air Pollution on Susceptible Subpopulations) was designed to assess whether air pollution increases the risk of first acute myocardial infarction (MI), or the subsequent risk of cardiac events among patients who have survived their first MI (Lanki et al., 2003). Using a common protocol, the cohort will consist of approximately 27,000 cases of non-fatal MI derived from hospital admission registries located in Germany, Spain, Sweden, Italy and Finland. Although pooled associations were but generally weak, the researchers concluded that acute exposure to carbon monoxide (CO) and ultrafine particles ($PM_{2.5}$) increases risk of hospitalization for new MI even after controlling for the potential confounding effects of weather and personal health characteristics.

Many cohort studies of health impacts from air pollution are limited by the quality and extent of exposure data available. For example, many older air pollution monitoring stations were activated to monitor specific point source pollutants resulting from manufacturing or industrial processes and not for the measurement of ambient exposures. In Scotland, however, long-term (>50 year) records exist for measurements of black smoke and sulphur dioxide (SO_2). Yap et al. (2003) have linked this valuable exposure data set to health outcomes data derived from three existing cohorts consisting of more than 26,000 subjects; they are also able to link each subject to individual hospital admission data and/or to mortality records using the Scottish Health Record Linkage System. Multi-level modeling and GIS techniques will allow for the investigation of potential confounding and effect modification by individual and aggregate level factors, and long term exposure data will enable and examination of exposure latency durations and temporal variation in pollutant concentrations. Results of this work are still forthcoming.

Cross-Sectional Studies

Cross sectional studies measure the prevalence of health outcomes or determinants of health in a population at a point in time or over a short period. This type of information can be used to explore etiology, for example the association between exposure to air pollution and any variety of health outcomes. Heinrich et al. (2003) in the Bitterfeld study utilized data from three cross-sectional surveys taken over a seven-year period to determine temporal changes in the impact of ambient pollution on children's health. The study included a total of 5,360 children ranging in age from 5 to 7 years. The children were administered a questionnaire, and subsequent tests of spirometry, histamine response and blood analysis were employed to assess physiological response. Children from more polluted areas showed poorer respiratory health and ambient particles were correlated with increased risk of allergenic sensitization.

Another cross-sectional study has also reported acute effects of air pollution on respiratory conditions among susceptible populations. The AUPHEP (Austrian Project on Health Effects of Particulates) project investigated the short term effect of particulate matter on lung function, morbidity and mortality in three urban centres (Vienna, Linz, and Graz) and one less polluted rural control area. The investigators found fine particles to be associated with cardiovascular mortality among the elderly of Vienna, an increase in respiratory-related hospital admissions, and association with lung function impairment among children in Linz. Thus, despite overall improvements to air quality in Austria, particulate matter and traffic-related air pollution still pose human health risks, especially for the elderly and children.

More recently epidemiological investigations have focused on health effects from traffic-related air pollution – a large source of urban ambient air pollution. The APHEIS (Air Pollution and Health: A European Information System) network has collected air pollution concentration information for 26 European cities of which two, located in Madrid and Rome, are dedicated traffic-based monitors. Mean concentrations among all sites ranged from 20 to 45$\mu g/m^3$ for fine particles; however, values from the traffic-related monitors revealed concentrations in the upper end of the distribution at 37 and 43$\mu g/m^3$ for Madrid and Rome respectively. Cross-sectional analysis of health and air pollution data from 1999 reveal increased chronic- and acute-related mortality in cities with higher vehicular particulate concentrations (Mücke and Medina, 2003). Vehicular traffic is also a major contributing source of ambient urban air pollution in Tblisi, Georgia. To investigate the potential for health impacts Samadashvili (2003)

examined the leading morbidity outcomes of children between the ages of 7 and 17 living in either an area with high traffic related pollution concentrations or in a suburban area with much lower concentrations. Children from the more polluted district were more likely to have some form of respiratory illness and were 2.5 times more likely to contract some form of skin disease.

Time-Series Studies

Evidence of associations between air pollution and human health impacts is derived primarily from time series studies. A time-series approach is used to associate daily fluctuations in ambient air pollution levels with daily rates of mortality or morbidity, and have consistently demonstrated increased risk to health on or shortly following day with increased ambient pollution levels in urban centres worldwide. The analysis of time series data also requires careful consideration of seasonal trends in the data, as well as adjustment for important covariates such as climate and gaseous co-pollutants (Burnett et al., 1995).

In Reggio, Italy, a time-series approach was adopted to evaluate the short term effects of urban air pollution levels on childrens' emergency visits from respiratory causes (Bedeschi et al., 2003). Researchers used generalized additive regression models to develop risk estimates of emergency visits after controlling for temporal and weather-related influences. Statistically significant and positive associations were found for NO_2 (11% increase in emergency visits) and for PM_{10} (3% increase). An analogous statistical approach was adopted by Vigotti et al. (2003) to investigate the association between ambient air pollution (CO, NO_2, and PM_{10}) and presentations to the emergency room for respiratory-related complaints by children and elderly patients in Pisa, Italy. A significant increase in emergency room visits was confirmed for children and the elderly resulting from particulate pollution exposure, after controlling for weather and temporal variability.

An important but often overlooked consideration in the evaluation of time-series studies linking particulate air pollution levels with health outcomes is the influence of specific metal concentrations located on the particle surface compared to the particle mass concentration alone. To examine this issue more thoroughly, Beverland et al. (2003) conducted a long-term study to investigate the relationship between elevated respiratory and cardiovascular mortality and morbidity rates and the variance of metal composition of PM_{10} in urban and rural locations in Edinburgh, Scotland. The researchers used a novel air mass trajectory approach to determine the particle characteristics of air masses common to the Edinburgh area. Air masses centered on the United Kingdom or originating from east/central Europe were associated with a 25% greater concentration of both PM_{10} and $PM_{2.5}$ when compared to air masses from other regions. A forward stepwise multiple regression modeling approach was adopted to examine the fraction of water soluble metal concentrations (Fe, Cu, Ni, V, Zn) as a constituent of particle mass. Generalized additive models were used to investigate the associations between health outcomes and either particle mass or metal fractions after controlling for particle mass. A positive, significant association was found between cardiovascular admission and a $10\mu g/m^3$ increase in total PM_{10} concentrations; although similar associations were found for specific metal fractions, the risk estimates became statistically insignificant after adjustment for total particle mass in the model.

Another novel development in the use of time-series approaches to investigate air pollution-related health effects is the integration of genotyping to identify susceptible subgroups and to determine the role of gene variation as a determinant of inflammatory response. In a proposed study, Peters et al. (2003) as part of the AIRGENE Study Group will use time series methods to determine the relationship between inflammatory response and air pollutants among survivors of myocardial infarction in six cities in Europe. Genotyping techniques will be used to provide insight into the mechanisms leading to inflammatory response in an effort to identify and define susceptible subpopulations and prevent the exacerbation of cardiovascular disease from ambient particle concentrations.

Several issues associated with the adoption of time-series approaches are noteworthy. First, although time series analyses provide important information about the effects of short term exposure on mortality, they are unable to identify effects with a lag period longer than several weeks (Dominici et al., 2000). Second, research has recently shown that time-series approaches can lead to overestimation of risk as well as overstatement of the precision of risk estimates due to convergence problems in the generalized additive model. A correction to this problem using more stringent convergence criteria confirms that mortality is associated with short-term fluctuations in particulate air pollution (HEI, 2003). Third, health effect estimates from time-series studies are sensitive to a number of

assumptions including model and parameter selection, comparison of health effect estimates to zero pollution levels, issues related to confounding from multi-pollutant mixtures, and non-standardized reporting methods (Sahsuvaroglu and Jerrett, 2003). It may be helpful in future research for investigators to explore more fully the consequences of uncertainty in the calculation and reporting of health risk estimates from exposure to air pollution.

OTHER STUDIES

Recent research contributions have illuminated the health impacts of air pollution at sub-regulatory concentrations, and have highlighted the need for researchers to consider the roles of human activity patterns and meteorology as influential in the air pollution and health risk relationship. An extensive review of peer-reviewed published research on the health effects from ozone suggests positive significant associations to health outcomes below the 40-80ppb 8-hour average range Walton (2003). Ambient concentrations of ozone below current Belgian air quality standards have been associated with inflammatory changes in the airways of children Nickmilder et al. (2003). Research has also revealed the importance of considering time-activity data and exposures to common chemicals as potential risks to the development of respiratory and cardio-pulmonary diseases such as asthma (Bernard et al., 2003). Moreover, attention should be directed towards a more thorough consideration of *atmospheric* risk factors than just air pollution. Air mass analysis and other meteorological approaches to controlling for the health impacts from variations in weather will help to elucidate the potential for weather/air pollution interactions and will enable public health authorities with the development of population health impact prediction systems from atmospheric risk factors (Michelozzi et al., 2003).

SUMMARY OF FINDINGS

The health effects of air pollution research posters represent important new findings from more than 41 cities and metropolitan areas across Europe. Consistent with previous air pollution and health research, many studies found children to be particularly susceptible to ambient concentrations of air pollution in urban areas resulting in decreased respiratory function and related morbidities. It is also clear that population health impacts are evident from both short- and long-term exposures to PM (Glorennec et al.). At the cellular level, studies reveal air pollution to have potential mutagenetic and oxidative effects which raises serious concern for the potential for oncogenicity and healthy cellular regeneration. Studies of biomarkers, such as Clara cell proteins and lymphocyte damage assessment, present the possibility of identifying air pollution effects at the cellular level and contribute to the evidence of air pollution as a causal factor for human health impacts.

At the other end of the spectrum, studies have focused on improvements to air pollution measurement and sources. For example, a few studies have shown that it maybe more prudent and efficient to assess particle mass rather than particle composition as an indicator of potential human health risk. Studies of the impact to health from air pollution are also moving upstream. It is well known that emissions from transportation sources are a major component of the total ambient air pollution concentration in large urban centres. From an epidemiological perspective, researchers are able to reduce bias due to misclassification and improve exposure assessment models by allocating air pollution exposure according to distance from traffic sources and/or land-use patterns. From a policy perspective, the close association between traffic patterns (frequency and mode) and air pollution concentrations leads to the development of transport policies and regulations with population health improvements as a primary objective.

POLICY IMPLICATIONS

The results of the research presented here present many opportunities and challenges for the development of policies for improvements to air quality and human health. Policymakers interested in morbidity and mortality related to air pollution should be made aware that the moderate and prolonged exposures, rather than peak exposures, are responsible for a majority of human health effects. Of particular concern are the critical pollutants – respirable forms of particulate matter, carbon monoxide, nitrogen oxides, and volatile organic compounds – that have been linked expressly to vehicle and transportation-related emissions. The results from epidemiological research suggest there is insufficient evidence to depart from assumptions of linearity in associations between air pollution and human health. Benefits to health may be realized from any reductions in air pollution *inter alia* pollution sources from transportation. Policymakers are thus presented with an opportunity to create population health improvements by

giving priority to reducing ambient pollutant levels from traffic sources, especially in residential areas. The challenge remains, however, on how best to achieve these reductions.

REFERENCES

Aalto, P., Paatero, P., Kulmala, M., Hämeri, K., Forastiere, F., Cattani, G., Marconi, A., Cyrys, J., Von Klot, S., Zetzsche, K., Peters, A., Bouso, L., Castaño, G., Palacio, J.A., Sunyer, J., Lanki, T., Pekkanen, J., Sjöval, B., Berglind, N., Bellander, T., and Nyberg, F. 2003. Aerosol number concentration measurements in five European cities during HEAPSS. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Aarnio, P., Kousa, A., Yli-Tuomi, T., Jantunen, M., Koskentalo, T., and Hillamo, R. 2003. Composition of and exposure to $PM_{2.5}$ while commuting in the metro and on the street. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Bayer-Oglesby, L., Ackermann-Liebrich, U., Downs, S., Kuna-Dibbert, B., Probst, N., Felber Dietrich, D., Staedele-Kessler, P., Schindler, C., Leuenberger, P., and SAPALDIA Team. 2003. SAPALDIA 2 – Swiss Cohort Study on Air Pollution and Lung Diseases in Adults. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Bedeschi, E., Campari, C., Candela, S., Caranci, N., Frasca, G., Collini, G., Galassi, C., and Vigotti, M.A. 2003. Urban air pollution and respiratory emergency visits at Paediatric Unit, Reggio Emilia, Italy. Preliminary results. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Bernard, A., Carbonelle, S., Nickmilder, M., Michel, O., Higuet, S., de Burbure, C., Buchet, J.-P., Hermans, C., Dumont, X., and Doyle, I. 2003. Lung hyperpermeability and asthma prevalence in schoolchildren: unexpected associations with the attendance of indoor chlorinated swimming pools. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Beverland, I.J., Heal, M.R., Agius, R.M., Hibbs, L.R., and Elton, R. 2003. The metal content of airborne particles: Application to epidemiological research. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Blomberg, A., Mudway, I., Forsberg, B., Nordberg, G., and Bernard, A. 2003. Clara cell protein (CC16) as a biomarker for ozone exposure in humans. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Brits, E., Schoeters, G., and Verschaeve, L. 2003. Evaluation of the mutagenicity of PM_{10} and $PM_{2.5}$ collected in an industrial and urban area of Antwerp. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Burnett, R.T., Dales, R., Krewski, D., Vincent, R., Dann, T., and Brook, J.R. 1995. Associations between ambient particulate sulfate and admissions to Ontario hospitals for cardiac and respiratory diseases. *Am. J. Epidemiol*. 142:15-22.

Cassee, F.R., Bloemen, H.J., Boere, A.J., Fokkens, P.H., Leseman, D.L., Catani, G., Johansen, B.V., Halatek, T., and Dybing, E. 2003. Qualitative differences in particulate air pollution at different locations throughout Europe (RAIAP). Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Cebulska-Wasilewska, A. Panek, A., Pawłyk, I., Farmer, P., and Popov, T. 2003. Influence of occupational exposure to PAHs on lymphocytes susceptibility to the induction of DNA damage (sampling in Sofia). Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

de Berardis, B., Arrizza, L., Inglessis, M., Mosca, M., and Paoletti, L. 2003. Seasonal trend of the Physico-chemical characteristics of $PM_{2.1}$: A study by SEM/EDX and XPS in an urban area of Rome. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

de Burbure, C.Y., Heilier, J.-F., Nève, J., Becker, A., Albrecht, C., Borm, P.J.A., Gromadzinska, J., Wasowicz, W., Rydzynski, K., and Bernard, A.M. 2003. Lung permeability, antioxidant status and NO2 inhalation: a selenium supplementation study in rats. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Dominici, F., Zeger, S.L., and Samet, J.M. 2000. A measurement error model for time-series studies of air pollution and mortality. *Biostatistics* 1:157-175.

Dominici, F., McDermott, A., Zeger, S.L., and Samet, J.M. 2003. Airborne particulate matter and mortality: time-scale effects in four US cities. *Am. J. Epidemiol*. 157:1055-1065.

Fierens, S., Mairesse, H., Focant, J.-F., Eppe, G., De Pauw, E., and Bernard, A. 2003. Impact of iron and steel industry and waste incinerators on human exposure to dioxins, PCBs and heavy metals: results of a cross-sectional study in Belgium. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Gábelová1, A., Valovičová, Z., Horváthová, E., Slameňová, D., Binková, B., and Farmer, P.B. 2003. EXPAH: Risk assessment of the air pollution mixtures -`in vitro study´. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Glorennec, P., and Monroux, F. 2003. Health impact assessment of PM_{10} exposures in the city of Caen, France. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Gerlofs-Nijland, M.E., Boere, J.F., Kooter, I.M., Leseman, D.L., Dormans, J.A., Donaldson, K., Mudway, I., Gustadisegni, C., Bloemen, H.J., van Bree, L., and Cassee, F.R. 2003. HEPMEAP: In vitro and in vivo toxic potency of ambient fine and coarse PM across Europe: the influence of traffic exhaust emissions. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Götschi, T., Bayer-Oglesby, L., Hazenkamp-von Arx, M.E., Luczynska, C., Sunyer, J., Villani, S., Heinrich, J., Norbäck, D., Forsberg, B., Jarvis, D., Künzli, N., and the ECRHS Working Group Air Pollution. 2003. ECRHS II – Correlations between surrogates of exposure to ambient air pollution ($PM_{2.5}$ mass, sulphur, NO_2, black smoke, silicon) across 21 European Centres. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Guastadisegni, C., Duggan, S., Mudway, I., Cassee, F.R., Pozzi1, R., and Kelly, F.J. 2003. Oxidative potential of environmental particulate matter from sites with varying traffic density: Correlation between antioxidants depletion and arachidonic acid release. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Health Effects Institute (HEI). 2003a. Revised Analyses of Time-Series Studies of Air Pollution and Health Revised Analyses of the National Morbidity, Mortality, and Air Pollution Study, Part II Revised Analyses of Selected Time-Series Studies Preprint version www.healtheffects.org/Pubs/TimeSeries.pdf.

Heinrich, J., Wichmann, H-E., and Bitterfeld study group 2003. Bitterfeld Study. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Ilacqua, V., and Jantunen, M. 2003. Exposure to traffic-generated VOCs: Total exposure in urban environments and the specific contribution of exposure while in traffic. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Lanki, T., Tiittanen, P., Forastiere, F., Nyberg, F., Paatero, P., Pekkanen, J., Peters, A.,

Sunyer, J., and HEAPSS group. 2003. Air pollution and hospitalisations for first myocardial infarction (MI) in the HEAPSS* cohort. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Lolova, D., Tabacova, S., and Petrov, I. 2003. Ambient air pollution and children's exposure to lead and cadmium in a smelter region in Bulgaria. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Marconi, A., Cattani, G., Cusano, M.C., Ferdinandi, M., Inglessis, M., Settimo, G., Viviano, G., and Forastiere, F. 2003. Two years fine and ultrafine particles measurements in Rome, Italy. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Metz, N. 2003. PM_{10} Exposure assessment for a city in Europe from 1950 – 2050. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Michelozzi, P. 2003. PHEWE: Assessment and Prevention of acute Health Effects of Weather conditions in Europe. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Mücke, H-G., Medina, S., and the Apheis Network 2003. APHEIS: Influence of traffic-related PM_{10} emissions on public health in 1999. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Neuberger, M., Schimek, M.G., Moshammer, H., and Hauck, H. 2003. Acute effects of particulate matter on respiratory diseases in Austria (AUPHEP). Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Nickmilder, M., Carbonnelle, S., de Burbure, C., and Bernard, A. 2003. Lung inflammation in children with short-term exposure to ambient ozone: evidence of a threshold. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

O' Connell, S., and Matthews, I.P. 2003. An Epidemiological Assessment of Individuals Biological Uptake of Particulate Air Pollution and Related Health Effects. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Peters, A., Brueske-Hohlfeld, I., Cyrys, J., Henneberger, A., Ibald-Mulli, A., Illig, T., Kirchmair, H., Kolz, M., Loewel, H., Meisinger, C., Rueckerl, R., Schaffrath, R.A, Wichmann, HE., Koenig, W., Forastiere, F., Picciotto, S., Perucci, C., Pistelli, R., Santarelli, P., Pekkanen, J., Lanki, T., Tiittanen, P., Salomaa, V., Eriksson, J., Kulmala, M., Aalto, P., Paatero, P., Bellander, T., Nyberg, F., Berglind, N., Pershagen, G., Sunyer, J., Marrugat, J., Jacquemin, B., Katsouyanni, K., Chrysohoou, C., Panagiotakos, D., and Antoniades, C. 2003. Air pollution and inflammatory response in myocardial infarction survivors: Gene-environment interaction in a high-risk group (AIRGENE). Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

PTCL (Physical and Theoretical Chemistry Laboratory). 2004. The University of Oxford, Oxford, UK. http://ptcl.chem.ox.ac.uk/MSDS/glossary/genotoxic.html.

Ramsay, T., Burnett, R.T., and Krewski, D. 2003a. Exploring bias in a generalized additive model for spatial air pollution data. *Environ. Health Perspect*. 111:1283-1288.

Ramsay, T., Burnett, R.T., and Krewski, D. 2003b. The effect of concurvity in generalized additive models linking mortality to ambient particulate matter (with discussion). *Epidemiology* 14:18-23.

Sahsuvaroglu, T., and Jerrett, M. 2003. Sources of uncertainty in calculating mortality and morbidity attributable to air pollution. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Singh, R., Kaur, B., Farmer, P.B., Sram, R.J., Kalina, I., Popov, T.A., Garte, S., and Taioli, E. 2003. EXPAH: Effects of polycyclic aromatic hydrocarbons (PAHs) in environmental pollution on exogenous and endogenous DNA damage – oxidative damage. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Samadashvili, K. 2003. Review of the Health Effects from Motor Vehicle Traffic in Tbilisi. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Sram, R.J., Beskid, O., Binkova, B., Dusek, Z., Smerhovsky, Z., Kalina, I., Popov, T.A., and Farmer, P.B. 2003. EXPAH: Chomosomal aberrations by fluorescence in situ hybridization (FISH)- biomarker of exposure to carcinogenic PAHs. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Sram, R.J., Binkova, B., Beskid, O., Chvatalova, I., Milcova, A., Stavkova, Z., Smerhovsky, Z., Rössner, P., Rössner Jr, P., and Farmer, P.B. 2003. EXPAH: Effects of polycyclic aromatic hydrocarbons (PAHs) on environmental pollution on exogenous and endogenous DNA damage – Czech Cohort. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Taronna, M., Bertazzi, P.A., Cavallo, D., Carrer, P., Maroni, M., Foà, V., and Mannucci, P.M. 2003. Short-term exposure to $PM_{2.5}$, $PM_{1.0}$ and blood coagulation in humans. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Vigotti, M.A., Chiaverini, F., Biagiola, P., and Rossi, G. 2003. Urban air pollution and emergency room visits for respiratory complaints. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Walton, H. 2003. Is there a threshold for associations between ozone concentrations and health outcomes? A review. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

Yap, C., Beverland, I.J., Agius, R.M., Hole, D., Robertson, C., Cohen, G., Henderson, D., Morris, G., Heal, M.R., Fowkes, G., and Elton, R. 2003. Health effects of long-term exposure to air pollution in Scotland. Poster presented at the 2nd AIRNET Annual Conference / NERAM International Colloquium 'Strategies for Clean Air and Health', November, 2003.

SOURCES OF UNCERTAINTY IN CALCULATING MORTALITY AND MORBIDITY ATTRIBUTABLE TO AIR POLLUTION

Talar Sahsuvaroglu, Michael Jerrett[1]

ABSTRACT

Assessing and quantifying the burden of illness and mortality from air pollution exposure relies on statistical estimates and other assumptions that have inherent uncertainties. Through an intensive study in Hamilton, Canada, this study illustrates for policymakers the sensitivity of health effect estimates to a wide range of possible uncertainties. Dose-response relationships were derived based on pooled and averaged estimates published in the scientific literature from 1997 to 2001. These estimates were applied to local air pollution, mortality and hospital admissions data for the years 1995-1999. The data were adjusted to reflect uncertainties in the current state of knowledge, including: (1) baseline pollution, (2) single versus multipollutant effects, (3) local or pooled estimates, and (4) chronic effects. The estimates of mortality ranged from 96-374 annual deaths, while admissions ranged from 139-607 respiratory and 479-2000 cardiovascular admissions. Chronic fine particle exposure resulted in 232 annual deaths. Conclusions: First, there should be an effort to reach a consensus on reporting scientific findings from air pollution studies using standardized study designs and reporting formats. Second, given the sensitivity of the estimates to underlying assumptions, an immediate need exists for widely accepted burden of illness and mortality estimation conventions. Third, many areas of air pollution research require considerable refinement before complete estimates can be ascribed.

[1] School of Geography and Geology, McMaster University, Hamilton, ON L8S 4K1 Canada

INTRODUCTION

Adverse health outcomes due to exposure to ambient air pollution exposure are a major public health issue. Assessing and quantifying these impacts, however, requires the application of estimations and uncertainties. By conducting an intensive analysis within one study location, we aim to illustrate the sensitivity of health effect estimates to a wide range of possible assumptions.

Specifically, we estimate mortality and morbidity associated with ambient air pollution exposure in Hamilton, Canada. Currently, Hamilton has some of the highest ambient air pollution in Canada, exceeding government objectives on about 20 days per year. The reasons for these high exposures include the following: (1) proximity to the Ohio River Valley, where coal-fired generating stations emit pollutants that travel hundreds of kilometres to Hamilton; (2) the Nanticoke coal-fired generating station located on the northern shore of Lake Erie, which also contributes considerably to local pollution; (3) increasing transportation emissions that result from automobile and truck traffic in and around the city; (4) local point source emissions from one of the largest industrial areas in Canada; and (5) topographic and meteorological conditions that often keep the pollution close to ground level (Jerrett et al., 2001; HAQI, 1997). All of these factors elevate ambient air pollution exposures and make the issue of health effects particularly important in Hamilton, thus making this a good location to assess the uncertainties in health effects assessments.

Clean Air Hamilton (CAH), a multi-stakeholder group tasked with advising Hamilton City Council on air pollution policy, requested an update on estimates of mortality and morbidity attributable to air pollution in the City previously prepared by Pengelly et al. (1997). The updated findings were calculated using the most recent research and data available at the time. Quantitative information from this new assessment can help local decision-makers to understand the size of the health effects from air pollution and to take action to improve population health in Hamilton. This paper summarizes our findings from this update, but more broadly gives policymakers elsewhere an appreciation of the challenges that underlie the estimation of mortality and morbidity attributable to air pollution exposure.

METHODS

Overview

To promote comparison with the Pengelly work, we followed a similar methodology. The methodology used in this paper followed a series of four sequential steps: (1) Identification of pollutants of interest through consultation with local officials and the scientific literature; (2) review of published results to identify risk coefficients for specific pollutants and conversion into comparable values; and (3) acquisition of relevant air quality and health outcome data; (4) estimation of the burden of illness. After completion of the basic burden calculations, we completed extensive sensitivity analyses, which are detailed below.

Identification of Pollutants of Interest

Based on consultations with the Health and Environmental Impacts Working Group for Clean Air Hamilton (CAH), we utilized the criteria pollutants that were indicated in the Hamilton-Wentworth Air Quality Initiative (HAQI) report in 1997. Specifically, we included particulate matter (PM_{10}), sulfur dioxide (SO_2), nitrogen dioxide (NO_2), carbon monoxide (CO), and ozone (O_3). Pengelly et al. (1997) also applied this methodology to Toronto data in 2000 (i.e., using the same pollutants except for the air toxics). In addition, we estimated the mortality attributable to fine particles (i.e., $PM_{2.5}$) because these have received increasing attention in the scholarly literature as particularly harmful to pulmonary function (Pope, 2000).

Literature Review

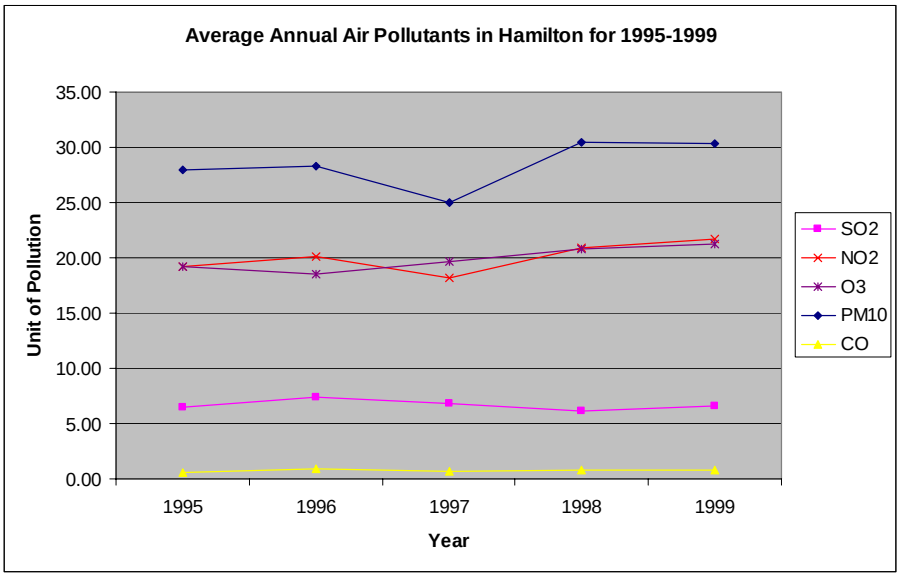
We conducted our literature review using the Medline and PubMed search engines. We searched combinations of the words "air pollution" with the following keywords – mortality, morbidity, health effects, time-series – for articles dated 1997 and onwards, until the beginning of October, 2001. Using Medline, 2067 related articles were identified, while the search in PubMed revealed about 6900 articles.

Subsequent review and selection of the articles was based on relevance, suitability of outcome measure, and significance of findings. We excluded articles that were not related to mortality or hospital admissions; those that focused on indoor air pollutants and tobacco smoke; those in languages other than English or French; and those that specifically identified elderly or infants as study groups. Articles that made use of multipollutant models were given priority to provide maximum control for co-pollutants. While findings from single pollutant models and significant associations with the elderly population were present and included in the literature review, they were not included in the average calculations. Studies including random effects and meta-analysis of previous studies as a comparative metric were selected. Research that used Hamilton estimates in particular was emphasized.

Chronic studies were also included in this analysis. Based on the limited number available, this literature review included a search as far back until 1993 when the earlier chronic effect literature was published. Recent reanalyses of these articles were included in the literature review. Additionally, the literature review was updated based on the recent discoveries of previously undetected problems in the statistical software used to apply generalized additive models (GAM) in time-series studies (Ramsey et al., 2003).

Air Quality Data

Annual averages for the identified pollutants were available for multiple locations in Hamilton, courtesy of the Ministry of Environment's monitoring network. Regional arithmetic averages from all the available stations were calculated to derive the city-wide average. Figure 1 illustrates the general trends in ambient pollution for the period 1995-99.



Note: SO_2, NO_2, and O_3 expressed in parts per billion (ppb), CO expressed in parts per million (ppm) and PM_{10} expressed in $\mu g/m^3$

Figure 1. Change of average annual air pollutants in Hamilton, 1995-1999.

Health Outcome Data

Annual morbidity and mortality data for Hamilton were supplied by the Central West Health Planning Information Network. The data were extracted from the Ontario "data warehouse," available through the Government of Ontario Network (GONET). The ICD-9 codes used were those indicated by the Pengelly studies, covering the area of the City of Hamilton. Complete mortality data sets were only available for 1995 to 1997, while hospital admissions data were available for a longer period (i.e., 1995 to 1999).

We observed a marked increase in the number of hospital admissions, especially for cardio-vascular (CV) admissions, between 1995 and 1996 (Table 1). We checked the acquired data for internal errors, but the difference seems to be due to other factors not reported by the Ministry of Health.

Table 1. Mortality and morbidity data for Hamilton used in the analysis.

	Non-traumatic mortality			**Morbidity**			
	All	CV	Resp	All	CV	CHF	Resp
1995	3,730	1,445	370	39,854	5,612	814	2,249
1996	3,694	1,422	367	41,149	7,702	1,123	3,085
1997	3,868	1,419	353	39,420	7,468	1,176	2,738
1998				40,044	7,322	1,108	3,266
1999				39,993	7,572	1,031	3,330
Average	3,764	1,429	363	40,092	7,135	1,050	2,934

NT= non-traumatic, CV = cardiovascular, CHF = congestive heart failure, Resp = respiratory

Estimating the Mortality and Hospital Admissions Associated with Air Pollution

Following the methodology set by the Pengelly et al. (1997 and 2000), we computed the relationship to estimate health outcomes as follows:

$$HO = B * \Delta H\% * P$$

where:

HO = annual health outcome
B = base number of outcomes per year
ΔH% = percent change in health outcome per unit increase of pollutant
P = annual pollution average

Similar methods have been used by Kunzli et al. (2000) and Mindel and Joffe (2004).

Example Calculation

The following data were utilized to calculate the premature mortality attributable to particulates (PM_{10}) for the year 1995:

- Total non-traumatic deaths in Hamilton for 1995 = 3730 deaths per year
- Percent increase in non-traumatic mortality for PM_{10}, averaged from literature values, per unit increase = 0.076 increase in deaths per 1 $\mu g/m^3$ increase in PM_{10} * 1/100
- Annual average of PM_{10} for Hamilton for 1995 = 27.9 $\mu g/m^3$

$$HO = 3730 \frac{deaths}{year} \times 0.076 \frac{deaths}{\mu g/m^3} \text{ per } 100 \text{ deaths} \times 27.9\ \mu g/m^3 = 79.09 \frac{deaths}{year}$$

The sample equation shows that the units cancel each other out to leave deaths per year as the final unit of analysis. Thus, following normal rounding rules, 79 premature deaths are associated with an increase of 10 $\mu g/m^3$ PM_{10} exposure in Hamilton for the year 1995.

Estimate Adjustments

Three adjustments were conducted on the original estimates based on the literature review and methods outlined above. The first involves a recent discovery of a statistical limitation in one of the software packages used in time-series analyses. The second adjustment pertains to an achievable baseline pollution estimate. The third adjustment was the application of both of the previous adjustments.

The time-series studies summarized in this paper typically have used GAM in their statistical analysis, as these models allow for control of time-varying factors through the incorporation of non-parametric smoothers of weather and other confounders. The findings from these studies are now in question due to recent research identifying a programming limitation in the statistical software used in these analyses (Ramsey et al., 2003). The statistical software provided biased risk estimates because it neither adequately accounted for concurvity nor assured convergence of its iterative estimation procedure. The Health Effects Institute's (HEI) Special Report summarized the reanalysed findings of 21 time-series analyses that were conducted using GAM models, and concluded that changes in estimates varied between less than 10% to above 40% (HEI, 2003). The reanalysis of the National Morbidity, Mortality and Air Pollution Study (NMMAPS) data, one of the largest pooled data sets in the U.S., revealed that the risk estimates have been overestimated by 36 - 42% (Dominici et al., 2002). These reanalyses showed that positive associations still exist, although in some cases they become insignificant.

Adjustments were made on the summarized findings of the average dose-response estimates in this paper. The values were adjusted to account for the maximum overestimation of 42%. This model is referred to as the "adjusted" model. We have also utilized recently published random effect estimates derived from meta-analyses (Stieb et al., 2003) overestimates resulting from the GAM estimation problems.

The second adjustment considers that in calculating risk estimates, the impacts are often sensitive to the range of values chosen to estimate population exposure. The World Health Organization (WHO) suggests that this sensitivity be quantified by conducting the analysis of health impacts under various exposure levels (WHO, 2001). The choice of range to use depends on realistic policy options, and can include theoretical zero concentrations, non-zero 'acceptable' levels, and up to concentrations determined by air quality standards. Estimates are often calculated in terms of comparison to the zero pollution level, which is considered to be unattainable and overly idealistic. We chose a more achievable estimate of a baseline of 20% of current pollution concentrations to emphasize this sensitivity. A separate estimate was calculated using annual pollution values of the mean minus the lower quintile, based on daily averages. These were calculated for 1997 to provide a comparison estimate. This adjustment is referred to as "baseline 20% model." Calculation of this 20% estimate required additional compilation of daily pollution data for a representative year (1997) to assess those in the lowest quintile.

Hamilton-specific Estimates

We also calculated estimates of studies conducted in Hamilton, using the research of Burnett et al. (1998a) for gaseous air pollutants and Jerrett et al. (2003) for the particulate metric, measured with the coefficient of haze (CoH). For these estimates, multipollutant models were used for the gaseous air pollutants, while single pollutant models were available for the particulate measures. The percent risks at the mean value for relevant years were computed.[2]

[2] Because the Poisson regression takes a log-linear form, we computed the risk estimates for each criteria pollutant as follows:

$$e^{(\beta \bar{x})}$$

where: e is the exponential function, β is the regression coefficient estimating the average increase in mortality associated with a unit increase in pollution, and $\bar{x}$ is the average of the air pollutant.

Random Effects Estimates

We also attempted to estimate pooled effects with a random effects model (Dersimonian and Laird, 1984). The relative risks (RR's) were extracted from the articles and reported as change in mortality/morbidity associated with an increase of 10 units of pollutant (except for CO, which was calculated per 1 unit for pollutant). Standard errors, 95% confidence intervals or t-ratios of the regression parameters were also extracted when available. As differences exist in reporting methods between authors, the same data was not present in every paper. Thus conversions between RR's and regression coefficients, and 95% confidence intervals and standard errors were applied to have comparable value formats.

The following two equations were used where required:
1. $RR = e^{(\Delta conc \times \beta)}$
2. $95\%\ CI = e^{[\Delta conc \times (\beta +/- 1.96 \times SE)]}$

where:
Δ conc = change in concentration of pollutant
CI = confidence interval
SE = standard error associated with estimate

RESULTS

Results of the Literature Review

A narrative and summary of our findings from this search is presented in Appendix 1. While single pollutant analyses are included in our commentary, they were not included in the calculations for final estimates. The detailed tables containing the literature review results are presented in Appendix 2. The tables include the study location, the modeled pollutants and the key results in a standardized format.

Results of Estimated and Adjusted Calculations

To calculate the final averages of the risk estimates from the literature, only multipollutant models were used. A simple averaging method for correlation studies was used to compute the overall effect from the literature (see Wolf, 1986). As well, the low and high ends of the findings are noted, as there are considerable differences in estimates of dose-response. Adjusted values were applied to the mean values. Recent pooled random effect estimates (Stieb et al., 2003) and estimates from chronic studies (Pope et al., 2002) were also included.

Notation in the following tables includes 'P1997' as the original HAQI report, Pengelly et al. (1997); 'P2000' as the City of Toronto report, Pengelly et al. (2000); 'CAH' as the current reanalysis of HAQI conducted for Clean Air Hamilton; 'Adjusted' as the current results with adjustment of 42% overestimate; 'M-min' (mean minus minimum 20%) represents the baseline 20% model; and 'M-min adj' indicates the baseline model adjusted for the 42% overestimate.

Relatively wide ranges can be observed within the estimated percent changes from increases in pollutants (Table 2). For a 10 $\mu g/m^3$ increase in PM_{10}, there was an increase ranging from 0.43% to 1.07% in non-traumatic deaths; 0.7-3.5% for respiratory admissions; and 0.5-2.3% in cardiovascular admissions. In the case of SO_2, the increase per 10 ppb resulted in a range of 0.84-3.89% increase in mortality; 1.3-6.1% for respiratory admissions; and 0.2-2.1% in cardiovascular admissions. The other pollutants follow similar ranges, with the higher ranges existing for morbidity results and lower ranges in mortality estimates. Adjusted mean values were slightly higher than the low end of the estimates, except for the association between O_3 and non-traumatic mortality.

Table 3 compares the average values for the risk estimates found in the literature after 1997 with the literature findings from the two previous studies and the adjusted values. This identifies the trends in literature values for the estimates. Current estimates were consistently higher than the 1997 estimates, except for PM_{10} estimates for non-

traumatic mortality and CO estimates for cardiovascular admissions. Adjusted values were lower than initial estimates for PM_{10} and O_3, but higher for SO_2, NO_2, and CO.

Table 2. Summary of percent changes per 10 units of pollutant: low, mean, high, and 42% adjusted mean estimates of calculated values.

Pollutant	NT mortality[a] (change per 10 units pollutant) range of estimates				Respiratory admissions[b] (change per 10 units pollutant) range of estimates				CV admissions[b] (change per 10 units pollutant) range of estimates			
	low	mean	high	adj mean	low	mean	high	adj mean	low	mean	high	adj mean
PM_{10} ($\mu g/m^3$)	0.43	0.76	1.07	0.44	0.7	2.1	3.5	1.22	0.5	1.4	2.3	0.8
$PM_{2.5}$(μ/m^3)	1.68	2.88	4.46	1.67								
SO_2 (ppb)	0.84	2	3.89	1.16	1.3	3.7	6.1	2.15	0.2	1.1	2.1	0.6
NO_2 (ppb)	1.5	1.9	2.3	1.10	1	4.9	9	2.84	4.4	6.55	8.7	3.8
CO (1 ppm)	2	3.68	4.95	2.13					0.4	1.95	2.5	1.1
O_3 (ppb)	0.94	1.38	1.7	0.80	1.5	2.8	4.9	1.62	1.6	4.5	7.5	2.6

NT= Non-traumatic; CV = cardiovascular;
[a] = Mortality values were calculated on the basis of 2 or 3 estimates
[b] = Morbidity values were calculated on the basis of 1 or 2 estimates; in the case of one estimate, 95% confidence intervals were used as the low and high range of estimates
adj mean = Mean estimate adjusted for 42% overestimate
Note: Because the ranges of data vary among pollutants, the 10-unit change is not directly comparable as a metric of severity in effects. For pollutants with a smaller range such as CO, a 10-unit change is proportionately larger than for PM_{10}, which has a larger range.

Table 3. Summary of percent changes per 10 units of pollutant, comparing average estimates of studies, adjusted and pooled estimates.

Pollutant	NT mortality (change per 10 units pollutant) average of estimates					Respiratory admissions (change per 10 units pollutant) average of estimates				CV admissions (change per 10 units pollutant) average of estimates			
	P1997	P2000	CAH	Adj	Pooled	P1997	P2000	CAH	Adj	P1997	P2000	CAH	Adj
PM_{10}	1	0.8	0.76	0.46	0.32	0.7	1.7	2.1	0.99	0.6	2.3	1.4	1.3
$PM_{2.5}$			1.9	1.10									
SO_2	0.6	2.25	2	1.16	0.85	0.4	2.76	3.7	1.60			1.1	0.0
NO_2	1.15	1.19	1.9	1.10	0.2	0.4	2.49	4.9	1.44		3.9	6.55	2.3
CO	1.1	3.48	3.68	2.13	0					5	6	1.95	3.5
O_3	0.3	0.4	1.38	0.80	0.3	0.8	1.1	2.8	0.64		4.52	4.5	2.6

Pooled = Pooled random effect model estimates (Stieb et al., 2003)
Adj = Mean CAH estimate adjusted for 42% overestimate

Table 4 presents the calculated mortality and morbidity estimates as incidences per year, using low, mean, high and adjusted risk estimates. Values ranged as in Table 2. Totals for all pollutants ranged from 248 to 567 annual deaths (using PM_{10} as a particulate estimate), to between 236 to 1252 respiratory and 993 to 3036 cardiovascular deaths. Adjusted mean totals were higher than the lower end estimates for all total counts.

Table 4. Summary of low, mean, high, and adjusted mean in the mortality and morbidity counts averaged for available years in current study.

Pollutant	NT mortality (incidences/year) calculated estimates				Respiratory admissions (incidences/year) calculated estimates				CV admissions (incidences/year) calculated estimates			
	low	mean	high	adj mean	low	mean	high	adj mean	low	mean	high	adj mean
PM_{10}	44	77	109	45	59	176	293	102	101	284	466	165
$PM_{2.5}$	108	185	286	107								
SO_2	22	51	100	30	30	72	140	42	10	52	100	30
NO_2	108	137	166	79	59	290	532	168	629	937	1244	543
CO	6	10	14	6					26	126	162	73
O_3	68	119	178	69	88	164	287	95	227	638	1064	370
Total	248	394	567	229	236	702	1252	407	993	2037	3036	1181
Total *	312	502	744	291								

* = total has been calculated with $PM_{2.5}$ instead of PM_{10}

Table 5 compares the estimates taken from the three studies and adjusted values, calculated on current air quality and health outcome data. Detailed calculations for these estimates can be found in Appendix 3. This table shows the differences in estimated mortality and morbidity counts according to the respective study values. The adjusted estimate is lower than any of the studies for mortality, at 229 annual deaths, but higher than the initial Pengelly study for morbidity at 407 annual respiratory and 1239 cardiovascular admissions.

Table 5. Summary and comparison of the mortality and morbidity counts using the average dose-response calculated in the three studies with adjusted values, applied to current Hamilton data.

Pollutant	NT mortality (average incidences/year) average of estimates				Respiratory admissions (incidences/year) average of estimates				CV admissions (incidences/year) average of estimates			
	P1997	P2000	CAH	Adjusted	P1997	P2000	CAH	Adjusted	P1997	P2000	CAH	Adjusted
PM_{10}	102	81	77	45	59	142	176	102	122	466	384	223
SO_2	15	58	51	30	22	81	72	42		629	52	30
NO_2	83	86	137	79	24	147	290	168	135	338	937	543
CO	3	10	10	6					20	50	126	73
O_3	97	29	119	69	53	66	164	95		641	638	370
Total	300	264	394	229	158	436	702	407	277	2124	2137	1239

Table 6 compares the original study, the current study, adjusted risk estimate values, baseline 20% adjustments, and application of both adjustments, all calculated for 1997 values. As the values show, there is a substantial difference in total mortality and morbidity counts, depending on the assumptions underlying the calculations. Our most conservative estimate, the application of both the 42% adjustment and the baseline 20% model, estimated 96 deaths in 1997 due to PM_{10}, compared to HAQI initial estimate of 298, our initial estimate of 374, and 217 deaths if the GAM discrepancy is taken into consideration. For respiratory admissions, the most conservative estimate is only a few admissions lower than HAQI estimates (139 compared to 144, respectively), while the highest estimate stands at 607 admissions. The highest estimate for cardiovascular admissions is our initial estimate of 2000 admissions, while the most conservative estimate is 479 admissions, still higher than the 257 admissions estimated by HAQI in 1997.

Table 6. Summary of the mortality and morbidity counts using the average dose-response in HAQI, CAH and both adjustments; applied to 1997 Hamilton data.

Pollutant	NT mortality (average incidences/year) average of estimates					Respiratory admissions (incidences/year) average of estimates					CV admissions (incidences/year) average of estimates				
	P1997	CAH	Adj	M-min	M-min adj	P1997	CAH	Adj	M-min	M-min adj	P1997	CAH	Adj	M-Min	M-Min adj
PM_{10}	97	73	43	24	14	48	144	83	46	27	112	280	157	84	49
SO_2	16	53	31	27	16	28	69	40	35	20		56	31	45	26
NO_2	81	134	78	46	27	20	244	142	83	48	125	888	497	303	176
CO	3	10	6	6	3						20	118	66	65	38
O_3	102	105	61	62	36	48	150	87	75	44		659	369	329	191
Total	298	374	217	119	96	144	607	352	239	139	257	2000	1120	826	479

M-min = Mean minus the minimum 20% (baseline 20% model). M-min adj = Adjusted value of M-min, for overestimate of 42%

Results of Hamilton-Specific Estimates

Hamilton-specific estimates revealed that, for NO_2 and CO, the values were comparable to the lower ranges of the literature estimates. For SO_2, estimates were slightly higher than the mean count from literature estimates, and Hamilton-specific O_3 estimates were at the higher end of the calculations (Table 7). Because the CoH estimate was derived from a non-GAM model, adjustments were not applied. For the remaining Hamilton-specific estimates, applying the adjustments brought their values closer to the mean of the literature estimates; however, the totals were not directly comparable to the remaining calculations because of the CoH component.

Table 8 summarizes all available calculations performed for non-traumatic mortality estimates.

Table 7. Comparison of the range of mortality counts using current estimates with averaged Hamilton-specific estimates and adjustments.

Pollutant	NT mortality (incidences/year) range in estimates						
	low	mean	high	Hamilton	Adj	$M\text{-}min_{1997}$	M-min adj
PM_{10}	44	77	109				
CoH				256	256	256	256
SO_2	22	51	100	73	42	37	21
NO_2	108	137	166	108	63	45	26
CO	6	10	14	5	3	4	2
O_3	68	119	122	122	71	81	47
Total	248	394	511	564	435	423	352

CoH= coefficient of haze (estimated from Jerrett (2003), using a non-GAM based model).

Table 8. Summary and comparison of mortality counts estimated for all available models, based on 1997 Hamilton pollution values.

Pollutant	NT mortality (average incidences/year) average of estimates								
	P1997	CAH	Adj	M-min	M-min adj	Pooled	Hamilton	Chronic	Chronic (M-min)
PM_{10}	97	73	43	24	14	31			
$PM_{2.5}$		110	64					348	140
CoH							256		
SO_2	16	53	31	27	16	22	73		
NO_2	81	134	78	46	27	14	108		
CO	3	10	6	6	3	0	5		
O_3	102	105	61	62	36	23	122		
Total	298	374	217	119	96	90		348	140
Total **		411	238				564		

Pooled = Pooled random effect model estimates (Stieb et al., 2003)
Hamilton = Hamilton-specific dose-response estimates
Chronic = Estimates based on chronic exposures to fine particulates (Pope et al., 2002)
CoH= Coefficient of haze (estimated from Jerrett (2003), using a non-GAM based model)
Total ** = Totals calculated with $PM_{2.5}$ or CoH as particulate measure

Results of Random Effects Models

Because there were no pooled estimates of morbidity analogous to the Stieb et al. (2003) article, we attempted to apply a random effects model to morbidity studies. In adhering to the constraints set in our literature review, with the emphasis placed on multipollutant models, we were only able to include two to three estimates per pollutant and outcome category. A random effects model was applied to outcomes with three estimates. The model reached convergence for only one of the pollutants (PM_{10} mortality), and this was run for estimates conducted before GAM adjustments. The model applied to morbidity estimates did not reach convergence, probably due to the small number of estimates and variability between them.

We also attempted to run the random effects model to include both respiratory and cardiovascular admissions to increase the number of estimates per category. These models reached convergence and had significant parameters. When calculating attributable morbidity, using 1997 data as the tables above, the results were comparable for PM_{10}, 28% lower for NO_2 and 21% lower for O_3 (Table 9).

Table 9. Comparison of combined morbidity counts estimated for initial and random effects models, based on 1997 Hamilton pollution values.

Pollutant	Combined morbidity (average incidences/year) average of estimates				Pooled RE estimates for combined morbidity (average incidences/year) average of estimates			
	CAH	Adj	M-Min	M-Min adj	CAH	Adj	M-min	M-min adj
PM_{10}	424	240	130	75	434	252	126	73
SO_2	125	71	80	46				
NO_2	1132	639	386	224	886	514	277	161
CO	118	66	65	38				
O_3	809	456	404	234	548	318	319	185

Combined morbidity = Respiratory and cardiovascular morbidity estimates. RE = Random effects model

DISCUSSION

This study has estimated mortality and hospital admissions associated with ambient air pollution in Hamilton, a mid-sized industrial city at the western tip of Lake Ontario. Dose-response relationships were derived based on exposure estimates published in the peer-reviewed literature. These estimates were applied to recent air pollution and health outcomes data available through routinely-gathered governmental sources.

Recent scientific discoveries identified software limitations in the GAM models used in time-series modeling. Applying the adjustments to account for a 42% overestimate lowered the average annual mortality rate to 229 from 394, respiratory admissions to 407 from 702, and cardiovascular admissions to 1181 incidences from 2137.

If further assumptions are taken into account by using the baseline 20% model of 1997 pollution values, annual mortality rates drop to 119 from 374, respiratory admissions to 239 from 607, cardiovascular admissions to 826 from 2000. Applying the 42% adjustment to these values revealed even lower mortality counts of 96 from 374, respiratory admissions at 139 from 607, and cardiovascular admissions at 479 from 2000. Similar baseline model adjustments and scenarios have been researched by Mindell and Joffe (2004). Differences in predicted premature deaths were estimated by applying four different theoretical models for pollution reduction in Westminster, England. Reductions to annual mean PM_{10} objectives to 24-hour PM_{10} objectives (current and 2009) as well as the effects of reducing PM_{10} to a zero-pollution level were considered. Adjustments to baseline pollution levels are important examples in emphasizing to policy-makers the sensitivities and uncertainties involved in the estimation of air pollution related health effects.

The Hamilton-specific estimates resulted in 352 annual deaths. This revealed that the total estimates of non-traumatic mortality were initially at the higher end of the range found in our literature review. With the adjustments, the values remained elevated but were not directly comparable to our other estimates due to differences in the particle metrics.

Pooled random effects model estimates from Stieb et al. (2003) resulted in 90 deaths, comparable to the 96 estimated by applying both GAM model and baseline 20% adjustments. Random effects models combining cardiovascular and respiratory morbidity estimates were also derived. These estimates revealed findings comparable for particulates, but lower than adjusted findings for the gaseous pollutants. The Dominici et al. (2002) and Stieb et al. (2003) GAM-adjusted findings were included in our mortality estimates for comparative purposes to cover the available range of model estimates. In future research, we will include the HEI (HEI, 2003) findings by incorporating updated morbidity and mortality estimates. Similar applications of random effects model estimates in calculating summaries for the effects of O_3 on a range of health outcomes have been utilized by the WHO (2003).

Chronic estimates of $PM_{2.5}$-related mortality produced an estimate of 348 deaths, and 110 using the baseline 20% adjustment. Both these values were higher than the adjusted acute-exposures summed for all other pollutants, despite being estimated for a single pollutant. Kunzli et al. (2001) have noted the likely pathways toward mortality burden from air pollution. Long-term exposure may contribute to the development of chronic disease that may occur through complex inflammatory and oxidative pathways over many years, such as the formation of atherosclerosis (Kunzli et al., 2004). Others work through the acute mechanisms, which may be more severe in susceptible individuals, who have underlying conditions that may or may not have been attributable to air pollution health effects. Thus the observation of chronic mortality effects probably represents both types of chronic and acute effects, some operating over many years and potentially leading to chronic conditions such as IHD or lung cancer, while others prey upon susceptible individuals with diseases such as diabetes. Thus, the larger chronic estimates reported here fit within the expected physiopathology of expected health effects.

Chronic estimates based on cohort studies are considered to be the “gold standard” for assessing health effects related to air pollution, due to their ability to assess life expectancy and incidence, course and remission of disease (Kunzli and Tager, 2000). The cohort study design provides the most accurate and comprehensive estimates of true health impacts as well as average reductions in lifespan due to pollution exposure. Thus, it captures the effects from both short-term and long-term outcomes, resulting in larger estimates (WHO, 2001). While researchers agree that time-series and cohort studies are methodologically different approaches for addressing the health effects of

exposure to pollution, disagreement still exists on how the results can best be used for estimations of total health burdens.

McMichael et al. (1998) suggest that time-series analyses are often inappropriately used to estimate longer-term effects. The time-series analyses report results of short-term exposures of individuals, not for sustained periods of exposure. Thus, the calculation of annual average mortality outcomes based on regression coefficients from the acute studies is criticized. The reason for this criticism stems from the notion that some of the deaths, while premature, may have occurred during the same year regardless of pollution exposure. This short-term displacement of deaths is also known as the harvesting effect. Following this reasoning, annual mortality may be overestimated due to the use of short-term estimates.

On the other hand, other researchers (Schwartz, 2000, 2001; Zeger et al., 1999), investigating the effect of harvesting on mortality estimates, found that time-series analyses often underestimate the exposure effect because the time lag usually employed did not account for effects occurring more than a few days after exposure. Kunzli et al. (2001) agree that time-series analyses underestimate the mortality attributable to air pollution exposure and that the results from cohort studies should be used instead. The WHO also maintains that time-series results are robust, both in terms of potential confounders and measurement error in exposure classification, and are able to provide estimates of premature mortality due to some recent exposure (WHO, 1996). Nevertheless, the time-series method still does not result in an accurate quantification of deaths due to air pollution exposure, and likely underestimates the total effects of air pollution (WHO, 2001, 2003). Thus, we caution that when interpreting results from our study (and similar studies), these limitations should be considered.

In our calculations, we did not include studies that concentrated on specific susceptible populations groups, nor did we attempt to incorporate inequalities in health. In Canada, we have evidence suggesting that persons with pre-existing conditions, such as diabetes (Goldberg et al., 2000) and persons of lower educational attainment may be more susceptible to the acute effects of ambient air pollution exposure (Jerrett et al., 2004). Estimates from different zones in Hamilton revealed effect modification by neighbourhood educational status and manufacturing employment. When these zonal estimates were pooled, however, the effects of the pooled model equalled those of a city-wide estimate (Jerrett et al., 2004). This may suggest that these effect modifiers have scale dependencies that negate the influence of susceptible populations in health effects assessments conducted at city-wide scales. For the estimates in this paper, we have assumed homogeneous susceptibilities across different strata of the population. Further research is needed to assess these heterogeneities in survival experience as they relate to air pollution exposure (Burnett et al., 2003).

A caveat is required with respect to the totals calculated in this paper and other similar efforts. They should be interpreted as general aids to decision-making rather than exact counts of death and illness. Researchers are often cautioned to avoid adding estimates of individual pollutants derived from single-pollutant models (WHO, 2001). If specific pollutants are not correlated, then adding single-pollutant effects may be justified; however, this must be done cautiously as pollutants often act in synergistic or antagonistic manners. As numbers of pollutants studied in multipollutant models increase, the estimates may become unstable due to collinearity (Samet et al., 1997).

Although we used multipollutant models to derive estimates, we used significant findings from estimates where collinearity between pollutants was accounted for. Some models, however, did not control for all criteria pollutants simultaneously. The uncontrolled confounding of co-pollutants may also influence the totals. In addition, each study may contain estimation error that is not accounted for in our simple averages of effect. Therefore, our totalled mortality estimates could exceed the actual number of deaths associated with air pollution and should be viewed with caution. Despite the limitations discussed, we summed the estimated effects to provide a direct comparison with the original Pengelly et al. (1997) document.

For an appreciation of the size of the uncertainties associated with the concept of summing estimates, we chose to separately assess several markers of independent aspects of the air pollution mixture. SO_2 can serve as a marker for localized industrial pollution, while NO_2 and CO are markers for traffic (Fenger, 1999). $PM_{2.5}$ accounts for long-range transportation and secondary sulfates from power plants (Brook et al., 2004; Burnett et al., 1997), while O_3 represents regional effects and reflect increases in the secondary photochemical pollution mixture (Bell et al., 2004).

Localized industrial pollution would then contribute 16 out of the 96 estimated total deaths, traffic indicators would result in 30 deaths, long range transportation would result in 22 deaths and the marker for secondary regional effects would produce approximately 36 deaths. Similar patterns were observed for respiratory and cardiovascular admissions. Thus, by estimating the separate markers, it appears that the largest contribution to adverse health outcomes is due to regional pollution effects, namely ozone exposure.

An alternative estimation would be to not sum pollutants that are known to be more correlated than others. Both O_3 and SO_2 (Bell et al., 2004; Katsouyanni et al., 2001; Gauderman et al. 2004) have been shown to be relatively uncorrelated with particulate matter, while NO_2 and CO have been identified as potential confounders due to high correlations (Burnett et al., 1997; Sarnat et al., 2001). This would indicate that perhaps effects of O_3, SO_2 and PM_{10} could be summed as they do not confound each other, but summing CO and NO_2 would be inappropriate. If we were to use our most conservative estimates, this would lead to 66 deaths, 91 respiratory admissions and 266 cardio-vascular admissions, compared to 96, 139 and 479 respectively, an average 37% decrease in total health estimates. These two applications again show the sensitivity of assessing health outcomes relating to the complex mix of pollutions in ambient air pollution exposure.

While we have adjusted for the GAM problem, which was a major statistical discovery that left scientists and policymakers questioning the magnitude of associations between acute exposures to air pollution and health, some continued uncertainty remains. Many time-series studies employed this method in their analysis, leading to many research groups reanalyzing their data in light of the new findings (e.g. Atkinson, 2004; Dominici et al., 2002). Reanalysis of both multi-city and single-city studies revealed that for the majority, the health effects of air pollution were still significant, but that the effects were slightly to substantially smaller. The WHO reported that an unpublished meta-analysis at the St. George Medical School (England) of 26 studies not using the GAM in their analysis averaged an increase of 0.4% per 10 $\mu g/m^3$ of PM_{10} (WHO, 2003). This was similar to both the lower end of our range of estimates (0.43%), and to our adjusted mean calculation (0.44%). This supports the use of our adjustment of the 42% decrease in observed effect (Dominici et al., 2002) to highlight the potential uncertainties that exist within the air pollution and health research. The close correspondence of our estimates with the new meta-analysis study by Stieb et al. (2003) lends further support to the validity of the 42% adjustment.

Another source of uncertainty is the "file drawer" problem, otherwise known as publication bias. Published research generally favours significant findings, while insignificant findings are rarely reported, leading to overestimates in the air pollution effect (Levy et al. 2000; WHO, 2004). While publication bias is a common problem in the general research culture (Simes, 1986; Begg and Berlin, 1989), it is only relatively recently being discussed specifically in the air pollution and health field (Anderson et al., 2002; Peacock et al., 2002). Since our study relies on published articles, there may be a bias in favour of positive findings and consequently inflated estimates.

Other considerations suggest our study may underestimate the total burden of illness due to air pollution in Hamilton. Our estimates only include mortality and acute health effects from air pollution. Other important health effects such as the development and exacerbation of asthma (Tenias et al., 1998; Yu et al., 2000), reproductive abnormalities (Bobak and Leon, 1999; Wang et al., 1997), elevated cancer rates (Beeson et al., 1998; Cohen, 2000) and less serious respiratory conditions such as infectious respiratory diseases (Kim et al., 1996) are excluded from this analysis.

As a final caveat, we emphasize that the different ways that the estimates were derived, calculated and discussed were to document and highlight the sensitivities and sources of uncertainty that exist in assessing air pollution health effects. While the literature review for our discussion is current, the original review for calculating the health effect estimates had to be limited to the end of 2001 with minor adjustments for later meta-analyses. These earlier estimates informed the policy process for air quality management in the City of Hamilton report. Thus, further updates would be needed to utilize this information for current policy making.

CONCLUSIONS

Four main points of concern arose from our research. First, there are no standardized methods for reporting the results from air pollution and health studies. This makes it difficult for individuals and groups working outside of academic structures to analyze the multitude of scientific findings. There are well-known limitations and difficulties

often associated with the interpretation of epidemiological studies (WHO, 1996, 2001). The necessity for thorough evaluation, accurate interpretation and appropriate presentation of uncertainties involved in impact estimates are especially highlighted. For example, the medical field utilizes the CONSORT (Consolidated Standards for Reporting Trials) statement to improve the quality of reporting results of a randomized control trial (Begg et al., 1996). This statement is a widely adopted requirement for medical journals such as JAMA, BMJ and The Lancet (Moher et al., 2001). While the WHO has developed a suggested protocol for conducting health impact assessments that calls for clear specifications and descriptions of purpose, approach, assumptions, methods, metrics and estimations, it does not specifically apply to reporting results (WHO, 2000). We recommend a standardized format for reporting of health effects that includes full disclosure of regression coefficients, standard errors, and significance tests. Such standardizations will ease the process of compiling and updating estimates.

Second, there are no widely accepted standards or conventions for dealing with important assumptions such as pollution baselines. In related fields such as economic and environmental accounting, conventions for dealing with normative issues have evolved through professional consensus (Jerrett et al., 1999). The calculations involved in estimating health effects attributable to air pollution can be compared to "health accounting" systems, where accepted conventions are utilized. To develop these conventions, accepted norms must be formalized to account for these uncertainties and limitations.

Third, a wide gap exists in the communication between scientists and policymakers. We repeatedly cautioned policymakers in Hamilton about the problem of summing estimates that may not have adequately controlled for co-pollutants. Yet the members of the CAH committee and policy representatives from the City emphasized the importance of "bottom line" estimates that could be used to inform the policy process and track ongoing progress. These differences can cause potentially complicated situations for users of scientific literature outside of the academic structure, such as policymakers and public health officials. Additionally, it indicates that much work remains in making scientific reporting formats more suitable and accessible to non-academic groups.

Fourth, we have learned that the over-reliance of scientific research on one method may result in situations similar to the GAM findings, in which results can change by as much as 42% literally overnight. While not all analyses of the acute effects of air pollution used this method, we feel that the impact was still significant enough to document as a source of uncertainty. In avoiding the dangers that exist in placing importance on one type of method or one type of estimate, we suggest that future studies should also incorporate a plurality of study research designs as well as methods, such as case control, cohort, ecologic and panel studies. Other gaps include the limited number of chronic effects studies, especially for morbidity, though this number is increasing both in European and North American contexts (e.g. Hoek et al., 2002; McConnell et al., 2002; Finkelstein et al., 2004; Nafstad et al., 2004).

Based on our analysis and experience with advising policymakers, we conclude with three suggestions for future research. First, there should be an effort to reach a consensus to report scientific findings using standardized or comparable methods. Second, given the sensitivity of the estimates to underlying assumptions, an immediate need exists for widely accepted health accounting conventions, particularly related to the baseline pollution level. Third, many areas of air pollution research require considerable work before complete estimates can be ascribed. Priority areas include studies on the chronic health effects of air pollution, multipollutant studies, and on health outcomes that are likely to have large population health impacts.

Acknowledgements:

We thank Dr. Tom Abernathy, Central West Health Planning Information Network, for supplying the mortality and morbidity data. Mr. Frank Dobroff, Ontario Ministry of the Environment, assisted with the air pollution data. We thank Drs. R. Burnett and S. Cakmak for assistance with programming the random effect models. We acknowledge helpful comments from Dr. Susan Elliott, Ms. Sonya Kapusin, Mr. Norm Finkelstein, Mr. Chris Giovis, Mr. Ric Hamilton and Mr. Pat DeLuca. We acknowledge funding from the City of Hamilton, Health Canada, the Canadian Institutes of Health Research, and the Southern California Environmental Health Sciences Center 5P30 ES07048 (funded by the National Institute of Environmental Health Science).

REFERENCES

Alberdi Odriozola, J.C., Diaz, J.J., Montero Rubio, J.C., Miron, P., Pajares Ortiz, I.M.S., and Ribera, R.P. 1998. Air pollution and mortality in Madrid, Spain: a time-series analysis. *Int. Arch. Occup. Environ. Health* 71:543-549.

Anderson, H.R., Atkinson, R.W., Peacock, J.L., and Sweeting, M. 2002. Publication bias in studies of ambient particulate pollution and daily mortality. *Epidemiology* 13:S149.

Atkinson, R.W., and APHEA2 Project Team. 2004. Acute effects of air pollution on admissions: reanalysis of APHEA 2. *Am. J. Resp. Crit. Care Med.* 169:1257-1258.

Beeson, W.L., Abbey, D.E., and Knutsen, S.F. 1998. Long-term concentrations of ambient air pollutants and incident lung cancer in California adults: results from the AHSMOG study. Adventist Health Study on Smog. *Environ. Health Perspect.* 106:813-823.

Begg, C., Cho, M., Eastwood, S., Horton, R., Moher, D., Olkin, I., Pitkin, R., Rennie, D., Schulz, K.F., Simel, D., and. Stroup, D.F. 1996. Improving the quality of reporting of randomized controlled trials. The CONSORT statement. *J. Am. Med. Assoc.* 276:637-639.

Begg, C.B., and Berlin, J.A. 1989. Publication bias and dissemination of clinical research. *J. Nat. Cancer Inst.* 81:107-115.

Bell, M., McDermott, L.A., Zeger, S.L., Samet, J.M., and Dominici, F. 2004. Ozone and short-term mortality in 95 US urban communities, 1987-2000. *J. Am. Med. Assoc.* 292:2372-2378.

Bobak, M., and Leon, D.A. 1999. Pregnancy outcomes and outdoor air pollution: an ecological study in districts of the Czech Republic 1986-8. *Occup. Environ. Med.* 56:539-543.

Borja-Aburto, V.H., Castillejos M., Gold D.R., Bierzwinski S., and Loomis D. 1998. Mortality and ambient fine particles in southwest Mexico City, 1993-1995. *Environ. Health Perspect.* 106:849-855.

Brook, J.R., Johnson D., and Mamedov, A. 2004. Determination of the source areas contributing to regionally high warm season $PM_{2.5}$ in eastern North America. *J. Air Waste Manage. Assoc.* 54:1162-1169.

Burnett, R.T., Cakmak S., Brook J.R., and Krewski, D. 1997. The role of particulate size and chemistry in the association between summertime ambient air pollution and hospitalization for cardiorespiratory diseases. *Environ. Health Perspect.* 105:614-620.

Burnett, R.T., Brook, J.R., Yung, W.T., Dales, R.E., and Krewski, D. 1997. Association between ozone and hospitalization for respiratory diseases in 16 Canadian cities. *Environ. Res*. 72:24-31.

Burnett, R.T., Cakmak S., and Brook, J.R. 1998. The effect of the urban ambient air pollution mix on daily mortality rates in 11 Canadian cities. *Can. J. Public Health*. 89:152-156.

Burnett, R.T., Cakmak S., Raizenne, M., Stieb, D., Vincent, R., Krewski, D., Brook, J.R., Philips, O., and Ozkaynak, H. 1998. The association between ambient carbon monoxide levels and daily mortality in Toronto, Canada. *J. Air Waste Manage. Assoc.* 48:689-700.

Burnett, R.T., Smith-Doiron, M., Stieb, D., Cakmak, S., and Brook, J.R. 1999. Effects of particulate and gaseous air pollution on cardiorespiratory hospitalizations. *Arch. Environ. Health* 54:130-139.

Burnett, R.T., Dewanji, A., Dominici, F., Goldberg, M.S., Cohen, A., and Krewski, D. 2003. On the relationship between time-series studies, dynamic population studies, and estimating loss of life due to short-term exposure to environmental risks. *Environ. Health Perspect.* 111:1170-1174.

Burnett, R.T., Dales, R., Brook, J., Raizenne, M., and Krewski, D. 2001. Association between ambient carbon monoxide levels and hospitalizations for congestive heart failure in the elderly in 10 Canadian cities. *Epidemiology* 8:162-167.

Cohen, A.J. 2000. Outdoor air pollution and lung cancer. *Environ. Health Perspect.* 108:743-750.

Daniels, M.J., Dominici, F., Samet, J.M., and Zeger, S.L. 2000. Estimating particulate matter-mortality dose-response curves and threshold levels: an analysis of daily time-series for the 20 largest US cities. *Am. J. Epidemiol.* 152:397-406.

Dersimonian, R., and Laird, N. 1986. Meta-analysis in clinical trials. *Control. Clin. Trials*. 7:177-188.

Diaz, J., Garcia, R., Ribera, P., Alberdi, J.C., Hernandez, E., Pajares, M.S., and Otero, A. 1999. Modeling of air pollution and its relationship with mortality and morbidity in Madrid, Spain. *Int. Arch. Occup. Environ. Health* 72:366-376.

Dominici, F., McDermott, A., Zeger, S.L., and Samet, J.M. 2002. On the use of generalized additive models in time-series studies of air pollution and health. *Am. J. Epidemiol.* 156:193-203.

Fenger, J. 1999. Urban air quality. *Atmos. Environ.* 33:4877-4900.

Finkelstein, M.M., Jerrett, M., and Sears, M.R. 2004. Traffic air pollution and mortality rate advancement periods. *Am. J. Epidemiol.* 160:173-177.

Garcia-Aymerich, J., Tobias, A., Anto, J.M., and Sunyer, J. 2000. Air pollution and mortality in a cohort of patients with chronic obstructive pulmonary disease: a time series analysis. *J. Epidemiol. Comm. Health* 54:73-74.

Gauderman, W.J., Avol, E., Gilliland, F., Vora, H., Thomas, D., Berhane, K., McConnell, R., Kuenzli, N., Lurmann, F., Rappaport, E., Margolis, H., Bates, D., and Peters, J. 2004. The effect of air pollution on lung development from 10 to 18 years of age. *New Eng. J. Med*. 351:1057-1067.

Goldberg, M.S., Bailar, J.C., III, Burnett, R.T., Brook, J.R., Tamblyn, R., Bonvalot, Y., Ernst, P., Flegel, K.M., Singh, R.K., and Valois, M.F. 2000. Identifying subgroups of the general population that may be susceptible to short-term increases in particulate air pollution: a time-series study in Montreal, Quebec. *Research Report - Health Effects Institute* 97:7-113.

Goldberg, M.S., Burnett, R.T., Bailar, J.C., III, Brook, J., Bonvalot, Y., Tamblyn, R., Singh, R., and Valois, M.F. 2001. The association between daily mortality and ambient air particle pollution in Montreal, Quebec. 1. Nonaccidental mortality. *Environ. Res.* 86:12-25.

Gouveia, N., and Fletcher, T. 2000. Time series analysis of air pollution and mortality: effects by cause, age and socioeconomic status. *J. Epidemiol. Comm. Health* 54:750-755.

Gwynn, R.C., Burnett, R.T., and Thurston, G.D. 2000. A time-series analysis of acidic particulate matter and daily mortality and morbidity in the Buffalo, New York, region. *Environ. Health Perspect.* 108:125-133.

HAQI (Hamilton-Wentworth Air Quality Initiative) 1997. Ambient air quality and effects on the environment in Hamilton-Wentworth - Environment Work Group Final Report. Hamilton, ON, Regional Municipality of Hamilton-Wentworth.

HEI (Health Effects Institute) 2003. Revised analyses of time-series studies of air pollution and health - Special Report. Boston, MA: Health Effects Institute.

Hoek, G., Brunekreef, B., Goldbohm, S., Fischer, P., and van den Brandt, P.A. 2002. Association between mortality and indicators of traffic-related air pollution in the Netherlands: a cohort study. *Lancet* 360:1203-1209.

Jerrett, M. 1999. Green costs, red ink: Determinants of municipal defensive expenditures in Ontario. *Prof. Geog*. 51:115-134.

Jerrett, M., Burnett, R.T., Kanaroglou, P., Eyles, J., Finkelstein, N., Giovis, C., and Brook, J.R. 2001. A GIS - environmental justice analysis of particulate air pollution in Hamilton, Canada. *Environ. Plan. A*. 33:955-973.

Jerrett, M., Burnett, R.T., Brook, J., Kanaroglou, P., Giovis, C., Finkelstein, N., and Hutchison, B. 2004. Do socioeconomic characteristics modify the short term association between air pollution and mortality? Evidence from a zonal time series in Hamilton, Canada. *J. Epidemiol. Comm. Health* 58:31-40.

Katsouyanni, K., Touloumi, G., Spix, C., Schwartz, J., Balducci, F., Medina, S., Rossi, G., Wojtyniak, B., Sunyer, J., Bacharova, L., Schouten, J.P., Ponka, A., and Anderson, H.R. 1997. Short-term effects of ambient sulphur dioxide and particulate matter on mortality in 12 European cities: results from time series data from the APHEA project. Air Pollution and Health: a European Approach. *Br. Med. J.* 314:1658-1663.

Katsouyanni, K., Touloumi, G., Samoli, E., Gryparis, A., Le Tertre, A., Monopolis, Y., Rossi, G., Zmirou, D., Ballester, F., Boumghar, A., Anderson, H.R., Wojtyniak, B., Paldy, A., Braunstein, R., Pekkanen, J., Schindler, C., and Schwartz, J. 2001. Confounding and effect modification in the short-term effects of ambient particles on total mortality: results from 29 European cities within the APHEA2 project. *Epidemiology*. 12:521-531.

Kelsall, J.E., Samet, J.M., Zeger, S.L., and Xu, J. 1997. Air pollution and mortality in Philadelphia, 1974-1988. *Am. J. Epidemiol.* 146:750-762.

Kim, P.E., Musher, D.M., Glezen, W.P., Rodriguez-Barradas, M.C., Nahm, W.K., and Wright, C.E. 1996. Association of invasive pneumococcal disease with season, atmospheric conditions, air pollution, and the isolation of respiratory viruses. *Clin. Infect. Dis.* 22:100-106.

Klemm, R.J., Mason, R.M., Jr., Heilig, C.M., Neas, L.M., and Dockery, D.W. 2000. Is daily mortality associated specifically with fine particles? Data reconstruction and replication of analyses. *J. Air Waste Manage. Assoc.* 50:1215-1222.

Kunzli, N., and Tager, I.B. 2000. Long-term health effects of particulate and other ambient air pollution: Research can progress faster if we want it to. *Environ. Health Perspect.* 108:915-918.

Kunzli, N., Jerrett, M., Beckerman, B., Mack, W., Gilliland, F., Thomas, D., and Peters, J. 2004. Association of subclinical atherosclerosis (CAROTH) intima media thickness) with residential ambient $PM_{2.5}$ in healthy adults. *Epidemiology* 15:S23-S24.

Kunzli, N., Medina, S., Kaiser, R., Quenel, R., Horak, F., Jr., and Studnicka, M. 2001. Assessment of deaths attributable to air pollution: Should we use risk estimates based on time series or on cohort studies? *Am. J. Epidemiol.* 153:1050-1055.

Levy, J., Hammitt, J., and Spengler, J. 2000. Estimating the mortality impact of particulate matter: what can be learned from between-study variability. *Environ. Health Perspect.* 108:109-117.

McConnell, R., Berhane, K., Gilliland, F., London, S.J., Islam, T., Gauderman, W.J., Avol, E., Margolis, H.G., and Peters, J.M. 2002. Asthma in exercising children exposed to ozone: a cohort study. *Lancet* 359:386-391.

McMichael, A.J., Anderson, H.R., Brunekreef, B., and Cohen, A.J. 1998. Inappropriate use of daily mortality analyses to estimate longer-term mortality effects of air pollution. *Int. J. Epidemiol.* 27:450-453.

Michelozzi, P., Forastiere, F., Fusco, D., Perucci, C.A., Ostro, B., Ancona, C., and Pallotti, G. 1998. Air pollution and daily mortality in Rome, Italy. *Occup. Environ. Med.* 55:605-610.

Mindell, J., and Joffe, M. 2004. Predicted health impacts of urban air quality management. *J. Epidemiol. Comm. Health* 58:103-113.

Moher, D., Schulz, K.F., Altman, D.G., and Consort, G. 2003. The CONSORT statement: revised recommendations for improving the quality of reports of parallel-group randomised trials. *Clin. Oral Invest.* 7:2-7.

Moolgavkar, S.H., Luebeck, E.G., and Anderson, E.L. 1997. Air pollution and hospital admissions for respiratory causes in Minneapolis-St. Paul and Birmingham. *Epidemiology*. 8:364-370.

Moolgavkar, S.H. 2000. Air pollution and hospital admissions for diseases for the circulatory system in three U.S. metropolitan areas. *J. Air Waste Manage. Assoc.* 50:1199-1206.

Morgan, G., Corbett, S., and Wlodarczyk, J. 1998. Air pollution and hospital admissions in Sydney, Australia, 1990 to 1994. *Am. J. Public Health* 88:1761-1766.

Morgan, G., Corbett, S., Wlodarczyk, J., and Lewis, P. 1998. Air pollution and daily mortality in Sydney, Australia, 1989 through 1993. *Am. J. Public Health* 88:759-764.

Nafstad, P., Haheim, L.L., Wisloff, T., Gram, F. Oftedal, B., Holme, I., Hjermann, I., and Leren, P. 2004. Urban air pollution and mortality in a cohort of Norwegian men. *Environ. Health Perspect.* 112:610-615.

Neas, L.M., Schwartz, J. and Dockery, D. 1999. A case-crossover analysis of air pollution and mortality in Philadelphia. *Environ. Health Perspect.* 107:629-631.

Peacock, J.L., Anderson, H.R., Atkinson, R.W., and Sweeting, M. 2002. Publication bias in studies of PM_{10} and children's lung function. *Epidemiology* 13:S149.

Pengelly, D., Szakolcai, A., Birmingham, B., Cole, D., Muller, P., Bailey, S., Bell, R., and Socha, A. 1997. Human Health Risk Assessment for Priority Air Pollutants - Hamilton-Wentworth Air Quality Initiative Human Health Working Group. Hamilton-Wentworth: Institute of Environment and Health, McMaster University.

Pengelly, D., Campbell, M., Ennis, S., Ursitti, F., and Li-Muller, A. 2000. Air Pollution Burden of Illness in Toronto. City of Toronto: Toronto Public Health.

Pope, C.A., III. 2001. Review: Epidemiological basis for particulate air pollution health standards. *Aerosol Sci. Tech.* 32:4-14.

Pope, C.A., III, Burnett, R.T., Thun, M.J., Calle, E.E., Krewski, D., Ito, K., and Thurston, G.D. 2002. Lung cancer, cardiopulmonary mortality, and long-term exposure to fine particulate air pollution. *J. Am. Med. Assoc.* 287:1132-1141.

Ramsay, T.O., Burnett, R.T., and Krewski, D. 2003. The effect of concurvity in generalized additive models linking mortality to ambient particulate matter. *Epidemiology* 14:18-23.

Saez, M., Figueiras, A., Ballester, F., Perez-Hoyos, S., Ocana, R., and Tobias, A. 2001. Comparing meta-analysis and ecological-longitudinal analysis in time-series studies. A case study of the effects of air pollution on mortality in three Spanish cities. *J. Epidemiol. Comm. Health* 55:423-432.

Samet, J., Zeger, S.L., Kelsall, J.E., Xu, J., and Kalkstein, L.S. 1997. Particulate air pollution and daily mortality. Analyses of the effects of weather and multiple air pollutants. The Phase I.B Report of the Particle Epidemiology Evaluation Project. Cambridge, MA. Health Effects Institute.

Samet, J.M., Dominici, F., Curriero, F.C., Coursac, I., and Zeger, S.L. 2000. Fine particulate air pollution and mortality in 20 U.S. cities, 1987-1994. *New Eng. J. Med.* 343:1742-1749.

Sarnat, J.A., Schwartz, J., Catalano, P.J., and Suh, H.H. 2001. Gaseous pollutants in particulate matter epidemiology: confounders or surrogates? *Environ. Health Perspect.* 109:1053-1061.

Schwartz, J. 1997. Air pollution and hospital admissions for cardiovascular disease in Tucson. *Epidemiology* 8:371-377.

Schwartz, J. 2000. Harvesting and long term exposure effects in the relation between air pollution and mortality. *Am. J. Epidemiol.* 151:440-448.

Schwartz, J. 2001. Is there harvesting in the association of airborne particles with daily deaths and hospital admissions? *Epidemiology* 12:55-61.

Simes, R J. 1986. Publication bias: the case for an international registry of clinical trials. *J. Clin. Onc.* 4:1529-1541.

Stieb, D.M., Judek, S., and Burnett, R.T. 2003. Meta-analysis of time-series studies of air pollution and mortality: update in relation to the use of generalized additive models. *J. Air Waste Manage. Assoc.* 53:258-261.

Tenias, M.J., Ballester, F., and Rivera, M.L. 1998. Association between hospital emergency visits for asthma and air pollution in Valencia, Spain. *Occup. Environ. Med.* 55:541-547.

Thurston, G., and Ito, K. 2001. Epidemiological studies of acute ozone exposures and mortality. *J. Expos. Anal. Environ. Epidemiol.* 11:286-294.

Wang, X., Ding, H., Ryan, L., and Xu, X. 1997. Association between air pollution and low birth weight: a community-based study. *Environ. Health Perspect.* 105:514-520.

WHO (World Health Organization) 1996. A Methodology for Estimating Air Pollution Health Effects. WHO/ENG/96.5. Geneva: WHO.

WHO (World Health Organization) 2000. Evaluation and Use of Epidemiological Evidence for Environmental Health Risk Assessment: Guideline document. EUR/00/5020369. Copenhagen, Denmark: Regional Office of Europe, WHO.

WHO (World Health Organization). Quantification of the Health Effects of Exposure to Air Pollution: Report of a WHO Working Group. EUR/01/5026342. 2001. Netherlands: WHO.

WHO (World Health Organization). Health Aspects of Air Pollution with Particulate Matter, Ozone and Nitrogen Dioxide. EUR/03/5042688. 2003. Copenhagen: WHO Regional Office for Europe.

WHO (World Health Organization) 2004. Health Aspects of Air Pollution - answers to follow-up questions from CAFE - Report on a WHO working group meeting Bonn, Germany, 15-16. Copenhagen: WHO Regional Office for Europe.

Wolf, F. 1986. Meta-analysis: Quantitative methods for research synthesis. Sage University Paper Series on Quantitative Applications in the Social Sciences, 07-059. Newbury Park, CA: Sage.

Yu, O., Sheppard, L. Lumley, T., Koenig, J., and Shapiro, G.G. 2000. Effects of ambient air pollution on symptoms of asthma in Seattle-area children enrolled in the CAMP Study. *Environ. Health Perspect.* 108:1209-1214.

Zeger, S.L., Dominici, F., and Samet, J. 1999. Harvesting-resistant estimates of air pollution effects on mortality. *Epidemiology* 10:171-175.

APPENDIX I

Carbon Monoxide (CO)

Non-traumatic mortality

Three studies have found significant associations between CO and non-traumatic mortality since the HAQI study in 1997. Burnett et al. (1998a, 1998b) and Gywnn et al. (2000) found an increase of 4.7%, 2.0%, and 4.13% per 1 ppm increase, respectively. The studies all used multipollutant models.

Respiratory hospital admissions

None of the literature reported significant associations between respiratory hospital admissions and CO.

Cardiac hospital admissions

CO was related to cardiac hospital admissions, specifically for hospitalization for congestive heart failure. Schwartz (1997) examined data for Tucson, US, and reported an increase of 1.4% in admissions per 1 ppm increase. Burnett et al. (1997c) calculated congestive heart failure admissions specifically for Hamilton and reported a 2.5% increase. Interestingly, the Toronto-specific estimate by the same researchers was comparatively higher at 6%.

Sulfur Dioxide (SO_2)

Non-traumatic mortality

Recent research shows a range of dose-response estimates for sulfur dioxide and total non-traumatic mortality. Garcia-Aymerich et al. (2000) found that in Barcelona, Spain, a 10 ppb increase in SO_2 led to a 4.2% increase in total mortality. Saez et al. (2001) found a 1.1% increase for three Spanish cities using a multipollutant model. In Madrid, Spain, Diaz et al. (1999) found a 2.1% increase in non-traumatic mortality with a single pollutant model. Taking 12 European countries into account, Katsouyanni et al. (1997) found an increase of 1.1%. Kelsall et al. (1997) considered a multipollutant model for Philadelphia, US, and found a 0.84% relative increase to the 10 ppm increase. Burnett et al. (1998a) studied SO_2 effects for 11 cities in Canada, using multipollutant models, and obtained a 3.89% increase in non-traumatic mortality for Hamilton.

Respiratory hospital admissions

Gywnn et al. (2001) associated an increase of 3.7% per 10 ppb increase in SO_2 in terms of respiratory hospital admissions. No other studies investigated this association.

Nitrogen Dioxide (NO_2)

Non-traumatic mortality

NO_2 has recently been significantly associated with non-traumatic mortality in a number of studies. In Rome, Italy, Michelozzi et al. (1998) found a 1.54% increase in a 10 ppb increase, while in Barcelona, Spain, Garcia-Aymerich et al. (2000) reported a 2.9% increase. Morgan (1998b) in Sydney, Australia indicated the value was closer to 1.5%. However, the latter study did not take multipollutant modeling into account. Burnett et al. (1998a) revealed a 1.5% increase in non-traumatic mortality associated with a 10 ppb increase in NO_2, specifically for Hamilton, while a 2.3% increase was estimated for Toronto.

Respiratory hospital admissions

Burnett et al. (1997a) found a 4.87% increase in respiratory admissions for Hamilton for a 10 ppb increase in NO_2.

Cardiac hospital admissions

Three studies found significant associations between NO_2 and cardiac hospital admissions. Burnett et al. (1997a) found an 8.7% increase for the 10 ppb increase in NO_2. Morgan et al. (1998a) found a lower value of 4.4%. However, a multipollutant model was not taken into account for this study. Moolgavkar (2000), in Los Angeles County, US, found a 1.7% increase, with a two-pollutant model (i.e., SO_2 and NO_2).

Ozone (O_3)

Non-traumatic mortality

There has been an increasing amount of research in ozone-related mortality. Recent studies showed significant associations between O_3 and non-traumatic mortality. Garcia-Aymerich et al. (2000) in Barcelona, Spain, estimated a 0.95% increase in non-traumatic mortality, while Gouveia et al. (2000) in Sao Paulo, Brazil, identified a 0.43% increase. In Philadelphia, US, Kelsall et al. (1997) found the relative risk to be at 0.94%, while in Santa Clara

County, California, US, Fairley (1999) estimated a much higher risk at 2.47%. Thurston and Ito (2001) calculated this value at 0.56% in a meta-analysis study based on 12 published estimates.

Respiratory hospital admissions

Moolgavkar et al. (1997) found a 4% increase in respiratory hospital admissions associated with a 10 ppb increase of ozone, while using a multipollutant model. Burnett et al. (1997b) found an increase of 1.5%; however, in his 1998 article (Burnett et al., 1998b), this value was estimated to be 4.9%. Gywnn et al. (2000) found this value closer to 2.0%.

Cardiac hospital admissions

Only one study, Burnett et al. (1997b) tested the ozone-admission association. They reported a 4.5% increase for cardiac hospital admissions. As this is the only study to find significant associations at such high values, this estimate should be considered preliminary.

Particulates

Non-traumatic mortality

Numerous studies have calculated the percent increase in daily mortality per 10 $\mu g/m^3$ increase in particulate matter, in the form of TSP, PM_{10}, $PM_{2.5}$, and SO_4^{2-}.

TSP: Alberdi Odriozola et al. (1998) and Diaz et al. (1999) conducted studies in Madrid, Spain, and found a 0.6% and 0.72% increase, respectively. In Rome, Italy, Michelozzi et al. (1998) calculated a comparable 0.66% increase. Neas et al. (1999) found a 0.56% increase in Philadelphia using a single pollutant study. Goldberg et al. (2001) calculated increases in non-traumatic mortality in Montreal and reported a value of 0.65% for a 10 $\mu g/m^3$ in TSP in single pollutant analysis. Kelsall et al. (1997) found a 0.3% increase in Philadelphia using a multipollutant model.

PM_{10}: Burnett et al. (1998b) estimated a 0.7% increase in non-traumatic deaths in Hamilton taking into account other pollutants, while in Montreal, Goldberg et al. (2001) calculated an increase of 0.69% in a single pollutant analysis.

In a meta-analysis, Daniels et al. (2000) found a 0.54% increase in non-traumatic deaths in 20 US cities. Samet et al. (2000) reported a 0.51% increase for 20 US cities considered. In their reanalysis of Schwartz et al. (1996) article on particulates in six US cities, Klemm et al. (2000) found a 0.8% increase associated with PM_{10}. Katsouyanni et al. (1997) reported non-traumatic mortality for PM_{10} increases equal 0.4% for the 12 European countries studied.

Primarily in European research, black smoke (BS) values were used as approximations to PM_{10} values. Saez et al. (2001) calculated a 0.64% increase for the three Spanish cities in the study, while Garcia-Aymerich et al. (2000) found this value closer to 1.1% in their single-pollutant analysis.

$PM_{2.5}$: Goldberg et al. (2001) found a 1.96% increase in non-traumatic mortality related to the increase in $PM_{2.5}$ in Montreal. Fairley (1999) calculated a 4.46% in Santa Clara County, US. Klemm et al. (2000) estimated this increase as 1.3% in a study of six US cities. In Mexico City, Mexico, Borja-Aburto et al. (1998) recorded a 1.68% in non-traumatic mortality associated with the fine particulates. Burnett et al. (1998b) reported a 2.5% increase in Hamilton.

Respiratory hospital admissions

PM_{10}: Moolgavkar et al. (1997) found a 1.7% increase in respiratory hospital admissions in Los Angeles County, US. Burnett et al. (1997b) calculated the relative risk at 2.1% in Hamilton, while Gywnn et al. (2000) found this value to be closer to 2.2% in New York, US.

$PM_{2.5}$: There were no studies that report significant associations within our literature time-frame and search specifications, mainly due to the concentration of studies reporting findings related to specific subpopulations of children and elderly.

SO_4^{2-}: Gywnn et al. (2000) estimated this to be 0.5% in New York, while Burnett et al. (1997b) reported 2.7% for Hamilton.

Cardiac hospital admissions

PM_{10}: Burnett et al. (1999) found a 0.5% increase in cardiac admissions in Toronto, Canada, while Morgan et al. (1998a) found this value closer to 0.76% in Sydney, Australia.

$PM_{2.5}$: Burnett et al. (1999) calculated a 0.75% increase. Again, this was the only study that found significance in our review, and it should be considered preliminary.

Appendix 2: Detailed Literature Summary Tables

Table A-1. Comparison of % increases in non-traumatic deaths in relation to increases of 10 units per pollutant.

			% change in daily mortality for each 10 unit increase in pollutant				
Reference	**Location**	**Multi Pollutant models**	**Particulates ($\mu g/m^3$)**	**SO_2 (ppb)**	**NO_2 (ppb)**	**CO (1 ppm)**	**O_3 (ppb)**
Borja-Aburto et al. (1998)	Mexico City, Mexico	$PM_{2.5}$, O_3, NO_2 (4 day lag)	$PM_{2.5}$ = 1.68 % (0.2, 3.14)				
Burnett et al. (1998a)	Estimates derived for Hamilton, Canada	CO, NO_2, SO_2, O_3		2.2%	1.5%	2.0%	1.7%
Burnett et al. (1998b)	Toronto	CO	PM_{10}= 1.5% (1.1,1.9) $PM_{2.5}$=2.5% (1.7,3.3)	3.89% (2.9, 4.86)	2.3% (1.6, 2.8)	4.95% (3.8, 6.1)	1.5% (1.2, 1.9)
Gwynn et al. (2000)	Buffalo, US	PM_{10}, CO	PM_{10} = 1.07% (0.02, 2.1)			4.1% (CI) (1.0, 7.2)	
Fairley (1999)	Santa Clara County, CA, USA	CO, NO_2, O_3, NO_3	$PM_{2.5}$ = 4.46%				2.47%
Kelsall et al. (1997)	Philadelphia, USA	TSP, SO_2, NO_2, O_3	TSP = 0.31% (0, 0.61)	0.84% (0.11. 1.57)			0.94% (0.35, 0.15)
Morgan et al. (1998)	Sydney, Australia	PM_{10}, NO_2, O_3	PM_{10}= 0.8%(0.0, 1.6)				
Saez et al. (2001)	3 Spanish Cities	SO_2, BS	BS= 0.64% (0.2, 1.1)	1.1% (0.2, 1.9)			
Chronic:							
Dockery et al. (1993)	6 US cities	yes	$PM_{2.5}$: 0.68% (0.5, 0.8)				
Reanalysis of 6 cities			$PM_{2.5}$: 0.69% (0.6, 0.8)				
Pope et al. (1995) ACS study	151 US cities	yes	$PM_{2.5}$: 0.48% (0.44, 0.51)				
Reanalysis of ACS (2000)		yes	$PM_{2.5}$: 0.48% (0.45, 0.52)				
Not used in the calculation of current estimate:							
Pengelly et al. (2000)	Toronto, Canada	depending on average calculation	PM_{10}= 0.8% $PM_{2.5}$= 1.5% (0.85, 2.2)	2.2%	1.19%	3.48% (24 hr)	0.4%

Table A-2. Comparison of % increases in non-traumatic deaths in relation to increases of 10 units per pollutant for studies using single-pollutant models and meta-analysis studies.

		% change in daily mortality for each 10 unit increase in pollutant				
Reference	**Location**	**Particulates ($\mu g/m^3$)**	**SO_2 (ppb)**	**NO_2 (ppb)**	**CO (1 ppm)**	**O_3 (ppb)**
Alberdi Odriozola et al. (1998)	Madrid, Spain	TSP = 0.6%				
Diaz et al.(1999)	Madrid, Spain	TSP = 0.72%	2.1%			
Garcia-Aymerich et al. (2000)	Barcelona, Spain	BS = 1.1% (0.5, 1.7)	4.2% (2.2, 6.1)	2.9% (0.7, 5.1)		0.95% (0.2, 1.6)
Goldberg et al. (2001)	Montreal	TSP = 0.65% $PM_{2.5}$= 1.96%				
Gouveia et al. (2000)	Sao Paulo, Brazil	PM_{10} = 0.51% (0.1, 0.9)	4.5 % (1.1, 7.9)			0.43% (0.00, 0.85)
Katsouyanni et al. (1997)	12 European cities	PM_{10}= 0.44% (0.2, 0.6)	1.1% (0.8, 1.3)			
Michelozzi et al. (1998)	Rome, Italy	TSP = 0.66% (0.31, 1.02)		1.54% (0.14, 2.97)		
Morgan et al. (1998)	Sydney, Australia			1.5% (0.2, 2.1)		0.7% (0.0,1..3)
Neas et al. (1999)	Philadelphia, USA	TSP= 0.56% (0.27, 0.86)				
Meta-analysis articles:						
Daniels et al. (2000)	20 US cities	PM_{10} = 0.54% (0.33, 0.76)				
Klemm et al. (2000)	6 US cities (reanalysis)	PM_{10}=0.8% $PM_{2.5}$=1.3% $SO_4^=$=1.6% (0.5, 1.1) (0.9, 1.7) (0.9, 2.4)				
Samet et al. (2000)	20 US cities	PM_{10} = 0.51% (0.07, 0.93)				
Thurston & Ito (2001)	Combined analysis					0.56% (0.32, 1.08)

Table A-3. Comparison of % increases in indicated morbidity values in relation to 10 unit increase per pollutant

				% change in daily morbidity for 10 unit increase in pollutant				
Reference	**Location**	**Multi-pollutants**	**Measure**	**Particulates ($\mu g/m^3$)**	**SO_2 (ppb)**	**NO_2 (ppb)**	**CO (1 ppm)**	**O_3 (ppb)**
Ballester et al. (2001)	Valencia, Spain	single pollutant	*card hosp adm*		1.1% (0.2, 2.1)			
Burnett et al. (1997a)	Hamilton	O_3, CO	*resp hosp adm*					1.5% (0.7, 2.2)
Burnett et al. (1997b)	Toronto	T, DP for PM_{10}, +SO_2, O_3 for NO_2 + PM, NO_2, CO for O_3	*card hosp admin* *resp hosp admin*	2.3% (0.3, 4.4) 2.1% (0.9, 3.3)		8.7% (3.2, 14.5) 4.9% (1.0, 9.0)	2.5% (0.2, 4.9)	4.5% (1.6, 7.5) 4.9% (2.7, 7.1)
Burnett et al. (1999)	Toronto	gaseous pollutants	*card hosp adm*	PM_{10}=0.50% $PM_{2.5}$=0.75%				
Gywnn et al. (2000)	Buffalo, NY	each gas against particulates	*resp hosp adm*	PM_{10} = 2.1% (0.7, 3.5) SO_4^{2-}= 0.5% (0.3, 0.7)	3.7% (1.3, 6.1)			2.0% (0.9, 3.0)
Morgan et al. (1998)	Sydney	single pollutant	*card hosp adm*	PM_{10} = 0.7 (0.2-1.3)		4.4% (3.06-5.8)		
Morris et al. (1998)	Chicago	PM_{10}, NO_2, SO_2, O_3	*chf hosp adm*				2.6% (1.0-3.9)	

URBAN AIR POLLUTION AND RESPIRATORY EMERGENCY VISITS AT PEDIATRIC UNIT, REGGIO EMILIA, ITALY

Emanuela Bedeschi, Cinzia Campari, Silvia Candela,[1] Nicola Caranci, Gabriella Frasca,[2] Giorgia Collini,[1] Claudia Galassi,[2] Maria Angela Vigotti[3]

ABSTRACT

Short-term effects of air pollution on daily mortality and hospital admissions for respiratory causes are well documented. Few studies however explore the association between exposure to air pollution and daily emergency room visits for respiratory disorders, particularly in Italy and particularly among children as a susceptible population. A time series analysis was conducted to explore the short term association between air pollutants (PM_{10}, TSP, NO_2, SO_2, CO, O_3) and Pediatric Emergency Room Visits (ER) in a small city of Northern Italy (Reggio-Emilia) during the period 03/01/2001 - 03/31/2002. 1051 ER visits were included in the study. Data were analyzed using Generalized Additive Models (GAM), adjusting for various confounding variables, including temperature, humidity and pollens (Graminaceae). The analyses were also stratified according to the nationality of children (Italians and Foreigners). In single pollutants models, the strongest associations were observed at lag 3 for a 10 $\mu g/m^3$ increase of TSP (2.7% increase in ER, 95%CI 0.7;4.6) and PM_{10} (3.0% increase, 95%CI 0.4;5.7), and at lag 4 for a 10 $\mu g/m^3$ increase of NO_2 (11.0% increase in ER, 95%CI 3.6;18.8). At lag 3, the percentage increase in ER visits is similar for the two groups of children (Italians and Foreigners) for TSP and PM_{10}. The results of the study support the findings that air pollution is a relevant determinant of deterioration of respiratory health among children.

[1] ASL di Reggio Emilia, Italy.
[2] Regional Agency for Health, Bologna, Italy.
[3] Universita di Pisa, Pisa, Italy.

INTRODUCTION

A large body of epidemiological research has focused on urban air pollution exposure and its consequences on respiratory health. Most of these studies were conducted measuring the admissions to hospital and mortality for respiratory diseases as main health effects (Atkinson et al., 2001; Atkinson, 2004; Brunekreef and Holgate, 2002; Samet et al., 2000). There are also several studies on emergency room (ER) visits all over the world, but mainly focused on asthmatic children (Norris et al., 1999; Thompson et al., 2001; Tolbert et al., 2000; von Klot et al., 2002). Fewer studies have analysed less severe end points such as ER visits focusing on respiratory symptoms and disorders in the general population and among young children (Atkinson et al., 1999; Delfino et al., 1997; Fusco et al., 2001). Recurrences of respiratory symptoms are certainly more frequent events than admissions to hospital. They enhance the demand for medical attention and, therefore, their increase can be considered as an indicator of deteriorating life quality.
Reggio Emilia is a city of northern Italy with almost 150 000 inhabitants, with only one hospital with a Pediatric Unit where all Emergency Room Visits (ER) are fully registered. Moreover, an air pollution monitoring station network has been in operation since 1989. During the winter of 2001-2002, the alarm threshold for air pollution episodes was often exceeded. Reggio Emilia offers high levels of welfare benefits and presents one of the lowest unemployment rates in Italy; therefore the immigration rate is quite high. Children of foreign parents, either born in town or immigrated very early, are exposed to the same air pollution as the Italian nationals. It is reasonable to expect similar effects on respiratory health, independent of parents' nationality, although socio-economic differences could play a role. The aim of this study was to evaluate the short-term effects of urban air pollution levels on ER visits due to respiratory symptoms among children in the city of Reggio Emilia (Northern Italy), and to investigate possible differences by nationality.

METHODS

Health Data

The number of ER visits for all causes among Reggio Emilia inhabitants under 15 years of age is about 13 000 per year.

At the registration desk of the Emergency Department children requiring an urgent visit are usually sent to the Pediatric Unit where, on a paper registry, information concerning the diagnosis, the applied therapy, any additional examination by a specialist or hospital admission are recorded, as well as children's personal data. Further demographic information was obtained using the Registry Office. A medical doctor (EB) collected and coded ER data from the Pediatric Unit registry concerning all the children under 15 years, residing in the Municipality of Reggio Emilia, for whom a diagnosis of respiratory disease was made.

The study period was from 03/01/2001 to 03/31/2002. Children with more than 5 ER visits due to influenza or influenza syndrome were not enrolled.

The respiratory disorders were analyzed as a whole and included the following diagnoses: "asthma or asthma like disorders" (if a diagnosis of asthma or bronchiolitis or of dyspnea/shortness of breath was recorded by the physician) and "other respiratory disorders" (upper and lower respiratory illness, including sinusitis, bronchitis and pneumonia).

"Non Italian children," i.e. children with a non Italian father, were identified either if the child was born outside of Italy or on the basis of father's surname, and hereafter we will refer to them as "Foreign children."

Air pollution and weather variables

Since 1989, Reggio Emilia Department of the Regional Environmental Protection Agency (ARPA) routinely monitors levels of air pollutants through a network of 6 monitoring stations located in different sites of the town. Allergenic pollens are also monitored through the network.

Pollutants analysed in this study were: NO_2, SO_2, CO, TSP, PM_{10}, O_3 and allergenic pollens. Weather variables (temperature, humidity, precipitation) were obtained from the Regional Meteorological Service (SIM).

Statistical methods

Data were analyzed using Generalized Additive Models (GAM) that take into account the effect of seasonal trend using semi-parametric smoothing function: penalized regression spline, with approximately 5 degrees of freedom (df), obtained specifying a smoothing parameter (sp). The choice of the number of df was made on the basis of minimisation of the AIC criterion and of observation of residual autocorrelation using ACF and PACF. Since GAM estimates using the *gam* function in the S-Plus statistical package may provide biased estimates (Dominici et al., 2002), the analyses were carried out with R software. The daily number of ER vists was considered distributed like a Poisson variable. The confounding factors included in the model were: weekday, dummy for festivity day, humidity, precipitation and temperature (current and delayed temperature, referred to the previous 3 days (Biggeri et al., 2001)), dummy for flu-epidemic (from the National Sentinel Surveillance System: FLU-ISS data base) and pollen concentrations (*Graminaceae*). The models were implemented including single pollutant a time, given the collinearity between the pollutants (Table 1). Associations have been studied for different time lags (from the same day up to 5 days before). Results are presented as the percentage variation of the Relative Risk of having ER visits computed as usual as the: RR=100*(exponential (βI) –1) where β is the coefficient obtained for that pollutant and I is a specific increment of pollutant level. We have used I=10 μg/m^3 for all pollutants except CO (I=1 mg/ m^3) in order to make results comparable with most of the recent analyses.

Table 1: Matrix of linear correlation coefficients between air pollutants and weather variables; Reggio Emilia, Italy, January 2001-March 2002

	PM_{10}	**TSP**	**SO_2**	**NO_2**	**CO**	**O_3**	**Temp**	**Hum**
PM_{10}	1							
TSP	0.89	1						
SO_2	0.57	0.57	1					
NO_2	0.57	0.58	0.56	1				
CO	0.61	0.61	0.71	0.77	1			
O_3	-0.52	-0.52	-0.55	-0.50	-0.63	1		
Temp	-0.51	-0.49	-0.59	-0.63	-0.70	0.83	1	
Hum	0.38	0.41	0.22	0.33	0.37	-0.67	-0.50	1

RESULTS

Data on 1 051 ER visits were included in the study. 854 children (497 males, 357 females) were involved: 715 children visited once and only 13 visited 4 times during the study period. 210 children were Foreign (24.6%) and accounted for 271 (25.8%) visits. The median age was 2 years in both groups. ER visits were higher for males (611=58.1%) especially among Foreign children (176=64.9%). 254 asthma-related visits were included in the group (24%) and those included in the group of other respiratory disorders were 797 (76%). Only 26 ER visits (2.5%) were followed by a hospital admission; 17 of these admissions were among children included in the "Asthma group." ER visits for asthmatic disorders were much more frequent among Italians (207 = 26.5%) than among the Foreigners (47= 17.3%). ER characteristics such as the hour (of the day), the weekday, the month, as well as the mean number of ER visits per subject did not differ between the two groups of Italians and Foreigners. Most of ER visits were during the week-end and in the winter period (December to March). The rate of ER visits (number of ER visits per 1 000 inhabitants), stratified by nationality, was 43.2% among Italian children and 144.8% among Foreigners.

As shown in Table 2, daily values of airborne fine particulate were quite high during the study period: daily mean of PM_{10} concentration was 51.2 μg/m^3 (air quality standard: 40 μg/m^3 annual mean EC, 1999).

A significant association was found between ER visits for all respiratory disorders and particulate matter daily levels on previous days, either measured as TSP or as PM_{10}, i.e. particulate with a dynamic diameter below 10 microns.

Table 2. Statistics for air pollutants, meteorological variables and ER in Reggio Emilia, Italy, January 2001-March 2002 (daily values).

Pollutants	***unit***	**Mean**	***s.d.***	**min.**	**median**	**max.**
PM_{10}	$\mu g/m^3$	51.2	*30.6*	5	44.7	196.8
TSP	$\mu g/m^3$	60.4	*34.3*	3.7	52.4	272.9
SO_2	$\mu g/m^3$	9.3	*2.3*	4.6	8.9	20.9
NO_2	$\mu g/m^3$	49	*13.8*	21.6	47.5	107.5
CO	mg/m^3	1.4	*0.7*	0.4	1.1	4.6
O_3	$\mu g/m^3$	68.6	*46.7*	4.5	66.2	200.5
Summer-O_3	$\mu g/m^3$	107.5	*32.2*	30.9	105.4	200.5
Weather variabales						
Temperature	*°C*	12.8	*8.2*	-3.8	13.0	27
Relative Humidity	.	77.5	*10.8*	45.7	79.4	97.5
Precipitations	*mm*	1.5	*4.3*	0.0	0.1	37.8
Respiratory ER visits						
All	*count*	2.7	*2.5*	0	2	17
Italians	*count*	2.0	*2.0*	0	2	12
Foreigner	*count*	0.7	*1.0*	0	0	5

For a rise of 10 μg/m^3 in the daily level of TSP on the 3rd previous day (lag 3), the percentage variation in the risk for an ER visit for any respiratory disorder was 2.7% (95%CI 0.7;4.6) among all children residing in Reggio Emilia; among the group of Italian children it was 2.5% (95%CI 0.4;4.7) and among the Foreign children it was 3.7% (95%CI 0.2;7.3) always at lag 3, but in this group the most significant increase was 5% (95%CI 1.1;9.1) at lag 1. Figure 1 reports the results for the entire range of examined lags.

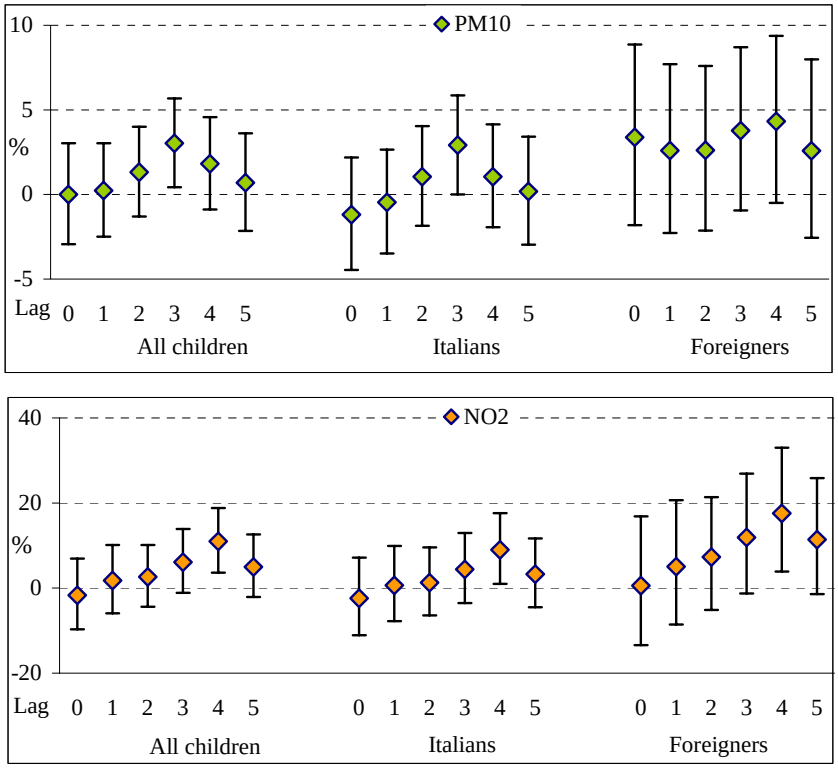


Figure 1: Single pollutant model results for all respiratory disorders. Associations are expressed as a percentage change (95% Confidence Interval) in the number of ER for an increase of 10 μg/m^3 in each pollutant (PM_{10}, TSP and NO_2), at different lags (0-5), and separately for Italians and Foreign children. Reggio Emilia.

The same increase in daily levels of PM_{10} at lag 3 was associated with an increase of 3.0% (95%CI 0.4;5.7) in the risk of ER visits for any respiratory disorder considering all children, and with a rise of 2.9% (95%CI 0.0;5.9) for the Italian children. No statistically significant results were seen among Foreign children (see Figure 1), although the higher values were observed at lag 4 (RR% 4.3, 95%CL: -0.5;9.4).

Finally, the rise of 10 $\mu g/m^3$ in the NO_2 levels was found to be correlated with an increase in the risk of ER visits for all groups and always at the same time lag (lag 4): among Italians the percentage increase was 9% (95%CI 1.0;17.6), among the Foreigners it was 17.6% (95%CI 3.9;33.0), while overall the percentage increase was 11.0% (95%CI 3.6;18.8). In Figure 1 the lower graph clearly shows the time trend of the increase of the risk of ER visits associated with NO_2 levels on the previous days, for each group of children.

No significant associations were found with daily levels of O_3, SO_2 or CO.

DISCUSSION

The study shows significant associations between urban air pollutants and Pediatric ER visits due to respiratory disorders. These associations are independent of the effect of temperature, humidity and pollen trend (Galan et al., 2003). The main pollutants involved are those linked to road traffic such as particulate matter (PM_{10} or TSP) and NO_2, as already suggested (Ciccone et al., 1998), and their effects are quite time delayed. The delayed time (3 or 4 days) may suggest that their main effect is to start a process of deterioration of respiratory health in children and that as a consequence after a few days they require urgent medical attention.

Another possible explanation of the delayed time is that parents bring children to the Emergency Units after having first consulted their own family practitioner, and if symptoms still persist and the family practitioner could not easily be reached (in the evening or at night, during the week-end).

The more frequent use of the Pediatric Emergency Unit observed for Foreigners compared to Italians, as well as the significant association found for TSP at a shorter lag (lag 1) among the Foreigners, could reflect a more inappropriate use of the Emergency Unit by the Foreigners, meaning that they first go to the emergency room rather than going to the family practitioner. However, this group is supposed to be characterized by a lower socio-economic status; for that reason an effect of air pollution as a modifier among children belonging to the lower socio-economic classes cannot be excluded. This could partially explain the slightly larger percentage increases among Foreigners' children. This matter certainly requires further investigation.

The observed effects of TSP, PM_{10} or NO_2 exposure are quite similar for each of the two nationality groups of children although results for the Foreigners exhibit larger confidence limits (for PM_{10} not significant), most likely as a consequence of the smaller size of the group. These results support the hypothesis that children experience similar effects in term of respiratory symptoms caused by the exposure in the town of residence, independently from parents' nationality.

Reggio Emilia, in Italy, is considered a town with a high level of welfare, good public health services, including a well functioning, easily accessible hospital. However, Reggio Emilia's weather is quite ugly, with long humid winters, frequent fog and hot humid summers. These climate conditions should likely enhance the noxious effects of air pollution, mainly due to urban traffic.

Although the short-term effect of air pollution on daily mortality and hospital admissions for respiratory causes are well known, few studies have explored the association between exposure to air pollution and emergency room visits, particularly in Italy and particularly among children as a susceptible population (Atkinson et al., 1999; Ribeiro, 1989; Ribeiro and Cardoso, 2003; Orazzo et al., 2001). Our study supports the finding that air pollution is a relevant determinant of deterioration of respiratory health among children.

REFERENCES

Atkinson, R.W., Anderson, H.R., Strachan, D.P., Bland, J.M., Bremner, S.A., and Ponce de Leon, A. 1999. Short term associations between outdoor air pollution and visits to accident and emergency departments in London for respiratory complaints. *Eur. Respir. J.* 13:257-265.

Atkinson, R.W., Anderson, H.R., Sunyer, J., Ayres, J., Baccini, M., Vonk, J.M., Boumghar, A., Forastiere, F., Forsberg, B., Touloumi, G., Schwartz, J., and Katsouyanni, K. 2001. Acute effects of particulate air pollution on

respiratory admissions: results from APHEA 2 project. Air pollution and health: a European approach. *Am. J. Respir. Crit. Care Med*. 164:1860-1866.

Atkinson, R.W. 2004. APHEA 2 Project. Acute effects of air pollution on admissions: reanalysis of APHEA 2. *Am. J. Respir. Crit. Care Med*. 169:1257-1258.

Biggeri, A., Bellini, P., and Terracini, B. 2001. Metaanalisi italiana degli studisugli effetti a breve termine dell'inquinamento atmosferico. *Epidemiol. Prev.* 25:1-72.

Brunekreef, B., and Holgate, S.T. 2002. Air Pollution and health. *Lancet* 1233:42.

Ciccone, G., Forastiere, F., Agabiti, N., Biggeri, A., Bisanti, L., Chellini, E., Corbo, G., Dell'Orco, V., Dalmasso, P., Volante, T.F., Galassi, C., Piffer, S., Renzoni, E., Rusconi, F., Sestini, P., and Viegi, G. 1998. Road traffic and adverse respiratory effects in children. SIDRIA Collaborative Group. *Occup. Environ. Med*. 55:771-778.

Delfino, R.J., Murphy-Moulton, A.M., Burnett, R.T., Brook, J.R., and Becklake, M.R. 1997. Effects of air pollution on emergency room visits for respiratory illnesses in Montreal, Quebec. *Am. J. Respir. Crit. Care Med.* 155:568-576.

Dominici, F., McDermott, A., Zeger, S.L., and Samet, J.M. 2002. On the use of generalized additive models in time-series studies of air pollution and health. *Am. J. Epidemiol.* 156:193-203.

EC 1999. Council directive 1999/30/EC of April 1999 relating to limit values for sulphur dioxide, nitrogen dioxide and oxide of nitrogen, particulate matter and lead in ambient air. *Official Journal of the European Commission (26.6.1999)*. L 163/41-60.

Fusco, D., Forastiere, F., Michelozzi, P., Spadea, T., Ostro, B., Arca, M., and Perucci, C.A. 2001. Air pollution and hospital admissions for respiratory conditions in Rome, Italy. *Eur. Respir. J*. 17:1143-1150.

Galan, I., Tobias, A., Banegas, J.R., and Aranguez, E. 2003. Short-term effects of air pollution on daily asthma emergency room admissions. *Eur. Respir. J.* 22:802-808.

Norris, G., Young Pong, S.N., Koenig, J.Q., Larson, T.V., Sheppard, L., and Stout, J.W. 1999. An association between fine particles and asthma emergency department visits for children in Seattle. *Environ. Health Perspect.* 107:489-493.

Orazzo, F., Rossi, G., Tassinari, D., DallaCasa, C., Dell'Erba, G., and Giardina, A. 2001. Air pollution, aeroallergens and emergency room visits for wheezing in very young children. *Am. J. Resp. Crit. Care Med*. 163, A562, Suppl.

Ribeiro, S.H. 1989. Air pollution and respiratory diseases in children in Sao Paulo, Brazil. *Soc. Sci. Med*. 29:959-964.

Ribeiro, S.H., and Cardoso, M.R. 2003. Air pollution and children's health in Sao Paulo (1986-1998). *Soc. Sci. Med.* 57:2013-2022.

Samet, J.M., Zeger, S.L., Dominici, F., Curriero, F., Coursac, I., Dockery, D.W., Schwartz, J., and Zanobetti, A. 2000. The National Morbidity, Mortality, and Air Pollution Study. Part II: Morbidity and Mortality from Air Pollution in the United States. Research Report 94. Cambrtidge MA: Health Effects Institute.

Thompson, A.J., Shields, M.D., and Patterson, C.C. 2001. Acute asthma exacerbations and air pollutants in children living in Belfast, Northern Ireland. *Arch. Environ. Health* 56:234-241.

Tolbert, P.E., Mulholland, J.A., MacIntosh, D.L., Xu, F., Daniels, D., Devine, O.J., Carlin, B.P., Klein, M., Dorley, J., Butler, A.J., Nordenberg, D.F., Frumkin, H., Ryan, P.B., and White, M.C. 2000. Air quality and pediatric emergency room visits for asthma in Atlanta, Georgia, USA. *Am. J. Epidemiol.* 151:798-810.

von Klot, S., Wolke, G., Tuch, T., Heinrich, J., Dockery, D.W., Schwartz, J., Kreyling, W.G., Wichmann, H.E., and Peters, A. 2002. Increased asthma medication use in association with ambient fine and ultrafine particles. *Eur. Respir. J.* 20:691-702.

URBAN AIR POLLUTION AND EMERGENCY VISITS FOR RESPIRATORY COMPLAINTS IN PISA, ITALY

Maria Angela Vigotti,[1] Francesca Chiaverini,[2] Piero Biagiola, Giuseppe Rossi[3]

ABSTRACT

Emergency room visits for respiratory complaints, considered an indicator of deterioration of respiratory health, have been positively correlated to traffic-related air pollution. This is an ecological study for evaluating the association between daily levels of urban air pollutants and emergency visits for respiratory complaints, in children and elderly residing in Pisa, Italy. Daily records on respiratory complaints visits to emergency department were selected from computerized registries, and paper medical reports were identified. Pollutant daily means were computed from hourly values obtained from public network. Poisson regression, allowing for over-dispersion and auto-correlation was used to evaluate the percent change in daily visits associated to variations of air pollution measure. Among children an increase in emergency visits of 10% (95%CL: 2.3;18.2) was associated with a 10 $\mu g/m^3$ increase of PM_{10} air level of the previous day and an increase of 11.8% (95%CL:1.4;23.3) was associated with an analogous increment for NO_2 of two days before. Among elderly the same increment of PM_{10} level of two days before was correlated with a 8.5% (95%CL:1.5;16.1) increase in the risk of recurring to emergency department for respiratory complaints; this risk increased to 26.5% (95%CL: 3.4;54.8) if the daily level of CO of four days increased by 1 mg/m^3. Evidence of such less severe health outcomes highlighted by this small study but also by larger ones, on short and long term time units, in different climatic countries suggest the need for actions to improve air quality in any urban context.

[1] Pisa University, Pisa, Italy.
[2] LHU-Empoli, Toscana, Italy.
[3] CNR-Pisa, Italy.

INTRODUCTION

Increase in the levels of air pollution may start and/or aggravate respiratory symptoms and may result in an urgent visit to the local hospital emergency room department. Since only the more serious Emergency Room Visits (ERVs) are likely to be hospitalised, ERVs can be considered as an indicator of deterioration of public respiratory health. To date, few studies have analysed health outcomes such as family doctor visits or ERVs due to respiratory symptoms. Children and the elderly are mainly investigated as they are considered the most susceptible to these exposures. Results indicate that proximity of residences to roadsides, exposure to high rates of road traffic, particularly to truck traffic, increases the risk of respiratory symptoms (Ciccone et al., 1998). Recent papers, based either on short-term or long-term effects designs (Atkinson et al., 1999; Martins et al., 2002; Gehring et al., 2002; Orazzo et al., 2001), found positive associations among traffic-related air pollutants and the risk of respiratory symptoms. In children living in non-urban towns in the Netherlands, respiratory symptoms were not associated with air pollution although they were correlated with a reduced respiratory function (Hoek et al., 1999). Traffic air pollution seems to be more dangerous to respiratory health of children if added to other environmental or socio-economic risk factors. In Sao Paulo, Brazil prevalence of respiratory symptoms among children of low-income families was similar to all other children in the low pollution area but it was much higher in areas with medium-high pollution levels (Ribeiro, 1989). Twelve years later in 1998 in the area of Sao Paulo, Brazil where both PM and SO_2 levels decreased, there was a reduction in the prevalence of respiratory symptoms (Ribeiro and Cardoso, 2003). This is an ecological study with the aim of evaluating the association between daily levels of urban air pollutants and ERVs for respiratory complaints, in children and elderly residing in the town of Pisa, Italy.

METHODS

Health data

Pisa hospital is centrally located in town, near the leaning tower. At the registration desk data, on patient's name, visit date and symptoms are routinely recorded on computer. Records on daily ERV for respiratory complaints were selected, by a medical doctor, from the ERV computerized registries for the year 2000. For each patient the paper medical report was identified to collect data on age, residence and presence of an anamnesis. Only residents of Pisa including children up to 10 years, and elderly over 65 years were considered.

Patients with influenza or influenza syndrome were excluded as well as those who visited the department six times or more in the year. Complaints were later coded by a co-author (FC) following the ICD 9 (International Classification of Disease, Ninth Revision), as due to asthmatic attack (ICD 9=493), dry cough (ICD 9=468), and acute bronchitis (ICD 9=466). However the ERV where analyzed as a whole, and not divided by groups of symptoms. Elderly with dyspnoea were excluded from the analyses as this symptom was associated to various pathologies; moreover the daily trend of dyspnoea was quite different from those of the other symptoms taken all together. Weekly data on influenza epidemics for the whole Tuscany Region were obtained from the Regional Health Agency and were used to estimate the daily curve of influenza incidence. The influenza epidemics occurred between December 1999 and January 2000.

Pollutants and meteorological data

Hourly data (from 0:00 to 24:00) on particulate matter up to 10μ in aerodynamic diameter (PM_{10}), nitrogen dioxide (NO_2), carbon monoxide (CO), temperature (T), relative humidity (RH) were obtained from the monitoring network system of Pisa. Missing temperature and relative humidity values were obtained from the Botanic Garden of Pisa University. The Regional Agency for Agricultural & Park Development and Innovation (ARSIA) provided data on daily maximum rain which measures millimetres of water collected during the longest period of continuous rain in the same day. Data were available from the 6 monitoring stations as follows: 3 stations for NO_2 (range of Pearson correlation coefficient ρ=0.43-0.44), 2 stations for PM_{10} (ρ=0.86), 2 for CO (ρ=0.81). Daily averages were computed if 75% of the hours were present. PM_{10} was measured by β-radiometry, NO_2 by chemioluminescence, and CO by infrared.

Statistical methods

Robust Poisson regression in a GAM model, allowing for over-dispersion (McCullagh, 1989), was used to evaluate the percent change in the daily number of ERV associated with the variations of the air pollution

measure. Non parametric smooth functions (loess) of the day of study, mean temperature, mean humidity, maximum rain and influenza epidemics were used. Other confounding variables included in the model were indicator variables for day of the week and holidays. Final core models included: time trend, holidays, days of the week, influenza cases on the previous day, millimetres of maximum rain two days before, temperature and relative humidity both at two days before. The window size for time was selected in order to minimize the Portmanteau statistic of the firth 60 partial autocorrelations of residuals. The span selection for the other covariates was based on the Akaike Information Criterion (AIC) while the selection of lags to introduce in the model was based on the Likelihood Ratio statistic. At the end we again examined the span for time and minimized the partial autocorrelations of the core model residuals. For time component four d.f./year were selected to fit the model of each age group. Finally the autocorrelations and partial autocorrelations of the core model residuals were checked. A linear term for pollutant was used and pollutant effects were investigated up to five days lags. Parameters were estimated using restrictive criteria as suggested by Dominici et al. (2002).

RESULTS

Approximately 94 000 citizens reside in Pisa including 6 200 children below 10 years of age and 20 800 adults over 65 years. 966 ERVs due to respiratory symptoms were recorded: 533 among children and 433 among elderly, excluding the symptom of dyspnoea. In both age groups more than 94% of the patients visited the ER department only once. Among children, respiratory complaints were mainly due to dry cough, while among elderly all the symptoms were almost equally represented. Pollutant concentrations were moderate and time trends, not reported here, do not show pronounced peaks. PM_{10} and CO decreased in summer while NO_2 levels did not vary along the year. Table 1 shows the distribution of all studied variables while correlation among pollutants and meteorological variables are presented in Table 2.

Table 1. Pollutants, meteorological and health variables, Pisa, 2000.

Pollutant		**unit**	**mean**	s.d.	**Min**	Median	**Max**
PM_{10}	24h	$\mu g/m^3$	**35.4**	15.8	**9.5**	31.6	**100.1**
NO_2	24h	$\mu g/m^3$	**45.6**	11.0	**21.3**	44.8	**74.0**
CO	24h	mg/m^3	**1.5**	0.7	**0.3**	1.4	**3.5**
Meteo							
Temp	24h	°C	**15.8**	6.1	**1.1**	15.6	**27.0**
Rel.Hum	24h	%	**82.7**	11.1	**42.8**	83.6	**99.6**
Max Rain	24h	mm	**2.2**	7.1	**0.0**	0.0	**90.6**
E.R.visit							
Children	24h	count.	**1.4**	1.3	**0**	1	**6**
Elderly	24h	count	**1.2**	1.1	**0**	1	**6**

Table 2: Correlations between daily values of pollutants and meteorological variables, Pisa, 2000.

	PM_{10}	**CO**	**R.H.**	**Temp.**	**Max Rain**
NO_2	**0.58**	**0.62**	*-0.01*	**-0.40**	**-0.19**
PM_{10}	**1**	**0.70**	**0.22**	**-0.42**	**-0.23**
CO		**1**	**0.42**	**-0.77**	*-0.03*
R.H.			**1**	**-0.40**	**0.36**
Temp.				**1**	*-0.08*

Bolded number are statistically significant $p<0.05$.

Uni-pollutant analyses were carried out and statistically significant associations with ERVs were found among children for PM_{10} and NO_2 concentrations and among elderly for PM_{10} and CO. Among children the Relative Risk (RR) of an ERV, for respiratory complaints, increases by 10% (95%CL: 2.3;18.2) if the PM_{10} air level of the previous day increases by 10 $\mu g/m^3$; the RR rises by 11.8% (95%CL:1.4;23.3) for the same increment in the air level of NO_2 of two days before. No significant increase was found for CO air levels of the previous day (a variation of +18.6%; 95%CL:-6.9;51.1 for an increment of 1 mg/m^3). Among elderly, the RR of a recurring ER, for respiratory complaints, rises by 8.5% (95%CL:1.5;16.1) for the same increment in the air levels of PM_{10} of

two days before; no significant association was found with same increment in NO_2 of two days before (6.0%; 95%CL:-3.3;16.2); a significant rise of 26.5% (95%CL: 3.4;54.8) was found in the RR for an increase of 1 mg/m^3 in the daily level of CO of four days before. Results are represented in Figure 1.

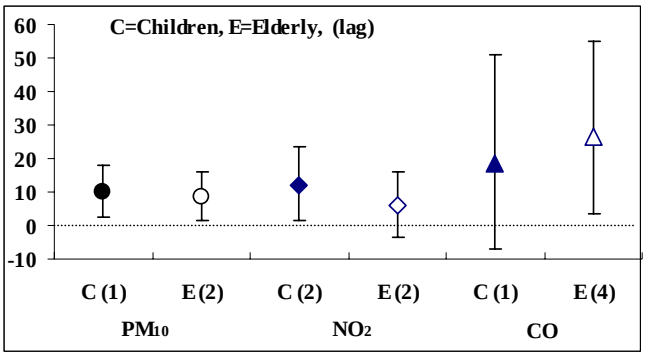


Figure 1. Percent variations of the relative risk of emergency room department visits for respiratory complaints in children (<10yrs) and in elderly (65+), for an increment of 10μg/m^3 in the daily air level of PM_{10} or NO_2 and of 1mg/m^3 of CO recorded on previous days, reported between parenthesis, Pisa, 2000.

An extensive sensitivity analysis was performed to investigate the effect on the final risk estimates of the type (loess with restricted criteria, natural spline and penalized spline) and the amount (degrees of freedom) of smoothing. This analysis showed that the results obtained from the selected model were very little sensitive to the type and to the amount of smoothing.

DISCUSSION AND CONCLUSION

This study has found a positive association between PM_{10} concentration in the urban air and emergency room visits due to respiratory complaints in children and in elderly. Limitations of the study are the shortness of the observational period and the size of the population at risk, which can explain the wide confidence intervals, and did not allow more disaggregated analyses, e.g. by sex, or age or specific respiratory complaint. These results confirm that increases of air pollution starts a process that after one or few days deteriorates the respiratory health status of children and elderly so far to require immediate medical attention. The Atkinson et al. (1999) study of 12 London hospitals in 1992-1994 found a stronger positive association between ERVs among children and SO_2, but still significant with PM_{10} or CO, while the association was not statistically significant for NO_2 and it was negative with ozone. Our results are consistent with these findings although the statistical power of our study was clearly lower. The Gehring et al. (2002) study of long term effects of exposure to traffic related air pollution on respiratory health, in a birth cohort of children below age two, found a positive association among the reports of cough without infection or dry cough at night and exposure to traffic related air pollution ($PM_{2.5}$, and NO_2). As the authors declared, although the study was designed to assess long-term effects it likely also included short-term effects. Hwang et al. (2002) studied daily clinic visits for lower respiratory tract illness by a small area design. They found that people over age 65 were the most susceptible, and estimated pollution effects decreased as the exposure time lag increased. The Italian multi-centres study on children respiratory health found a higher risk of chronic respiratory impacts among children living nearby roads with high rates of traffic particularly truck traffic (Ciccone et al., 1998). Moreover, deficits in lung function growth rate of children have been associated with their exposure to ambient air pollutant. In the second cohort of fourth grade children, larger deficits in lung function growth rate were observed in children who reported spending more time outdoors (Gauderman et al., 2002). Evidence of these less severe health outcomes highlighted by large and small studies, on short and long term time units, in different countries should be considered sufficient to start serious interventions to drastically improve the air quality in any urban context.

REFERENCES

Atkinson, R.W., Anderson, H.R., Strachan, D.P., Bland, J.M., Bremner, S.A., and Ponce de Leon, A. 1999. Short term associations between outdoor air pollution and visits to accident and emergency departments in London for respiratory complaints. *Eur. Respir. J.* 13:257-265.

Ciccone, G., Forastiere, F., Agabiti, N., Biggeri, A., Bisanti, L., Chellini, E., Corbo, G., Dell'Orco, V., Dalmasso, P., Volante, T.F., Galassi, C., Piffer, S., Renzoni, E., Rusconi, F., Sestini, P., and Viegi, G. 1998. Road traffic and adverse respiratory effects in children. SIDRIA Collaborative Group. *Occup. Environ. Med*. 55:771-778.

Dominici, F., McDermott, A., Zeger, S.L., and Samet, J. 2002. On the use of generalized additive models in tim-series studies of air pollution and health. *Am. J. Epidemiol.* 156:193-203.

Gauderman, W.J., Gilliland, G.F., Vora, H., Avol, E., Stram, D., McConnell, R., Thomas, D., Lurmann, F., Margolis, H.G., Rappaport, E.B., Berhane, K., and Peters, J.M. 2002. Association between air pollution and lung function growth in southern California children: results from a second cohort. *Am. J. Respir. Crit. Care Med.* 166:76-84.

Gehring, U., Cyrys, J., Sedimeir, G., Brunekreef, B., Bellander, T., Fischer, P., Bauer, CP., Reinhardt, D., Wichmann, H., and Heinrich J. 2002. Traffic-related air pollution and respiratory health during the first 2 yrs of life. *Eur. Resp. J.* 19:690-698.

Hoek, G.**,** Wypij, D.**,** and Brunekreef, B. 1999. Self-reporting versus parental reporting of acute respiratory symptoms of children and their relation to pulmonary function and air pollution. *Int. J. Epidemiol*. 28:293-299.

Hwang, J.S., and Chan, C.C. 2002. Effects of air pollution on daily clinic visits for lower respiratory tract illness. *Am. J. Epidemiol.* 155:1-10.

Martins, L.C., Latorre, M.R., Saldiva, P.H., and Braga, A.L. 2002. Air pollution and emergency room visits due to chronic lower respiratory diseases in the elderly: an ecological time-series study in Sao Paulo, Brazil. *J. Occup. Environ. Med.* 44:622-627.

McCullagh, P., and Nelder, J.A. 1989. *Generalized Linear Models.* 2nd Edition. Chapman & Hall: London.

Orazzo, F., Rossi, G., Tassinari, D., DallaCasa, C., Dell'Erba, G., and Giardina, A. 2001. Air pollution, aeroallergens and emergency room visits for wheezing in very young children. *Am. J. Resp. Crit. Care Med*. 163:A562.

Ribeiro, S.H. 1989. Air pollution and respiratory diseases in children in Sao Paulo, Brazil. *Soc. Sci. Med*. 29:959-64.

Ribeiro, S.H., and Cardoso, M.R. 2003. Air pollution and children's health in Sao Paulo (1986-1998). *Soc. Sci. Med.* 57:2013-2022.

INCREASE OF EXHALED NITRIC OXIDE IN CHILDREN EXPOSED TO LOW LEVELS OF AMBIENT OZONE

Marc Nickmilder, Claire de Burbure, Sylviane Carbonnelle, Xavier Dumont,[1] Alain Derouane,[2] Alfred Bernard[1]

ABSTRACT

Ozone (O_3) is known to induce lung function impairment and airways inflammation during episodes of photochemical smog. The aim of the present study was to assess the inflammatory effect of ambient O_3 in healthy children using nitric oxide in exhaled air (eNO) as a non invasive test. The study was performed on six groups of children (n = 11-15), aged 6.5 to 15 years who attended summer camps in rural areas of the South of Belgium in 2002. Ambient O_3 concentrations continuously monitored in the camps ranged from 48 to 221 µg/m^3 (1-hour maximal concentration). Children remained outdoors during the experimental days doing various recreational activities but no sports. Lung function tests (FEV_1 and FVC) and eNO were measured twice in each child in the morning and in the evening. Whilst lung function tests did not show any consistent pattern of decrease at these O_3 levels, a highly significant increase in eNO was found in all subjects from an ambient 1-hour O_3 level of 167 µg/m^3. A multivariate analysis did not reveal any influence of age, gender, height, weight and BMI of the children. The threshold for this O_3-induced increase in eNO estimated benchmark dose analysis was 135 µg/m^3 for 1-hour exposure and 110 µg/m^3 for 8-hour exposure. These observations suggest that ambient ozone produces early inflammatory changes in the airways of children from levels slightly below current air quality standards.

[1] Catholic University of Louvain, Brussels, Belgium.
[2] Interregional Cell of the Environment, Brussels, Belgium.
Supported by the European Union, Quality of Life and Management of Living Resources (HELIOS project, QLK4-CT-1999-01308).

INTRODUCTION

Ozone (O_3), the main oxidant of photochemical smog can reach high concentrations during hot summer days in industrial countries. Episodes of O_3 pollution affects non urban as well as urban areas, O_3 peaks tend even to be lower in large cities because of the scavenging of other pollutants. Depending on the inhaled dose and the sensitivity of subjects, O_3 produces a variety of adverse effects, including decreased lung function, inflammatory reactions, increased airways permeability and resistance, and aggravation of pre-existing respiratory diseases (Bylin et al., 1996; Committee of the Environmental and Occupational Health Assembly of the American Thoracic Society, 1996). Several air quality guidelines and standards are currently in application to protect the population from these harmful effects. The World Health Organisation has established a guideline value of 120 $\mu g/m^3$ for O_3 in ambient air for a period of 8 hours/day below which acute effects on public health are likely to be small (WHO, 2000). The National Ambient Air Quality Standards (NAAQS) of the US Environmental Protection Agency for O_3 are 0.12 ppm (235 $\mu g/m^3$) for 1-hour exposure and 0.08 ppm (157 $\mu g/m^3$) for 8-hour exposure (US EPA, 1997), respectively.

Research into the biomarkers field has much progressed during the last years with the development of several non-invasive tests or approaches allowing to evaluate lung inflammation or damage without specific restriction imposed by exposure conditions and the characteristics of examined subjects. Some of these tests such as serum specific Clara cell protein have been validated to monitor airways permeability changes during O_3 exposure (Blomberg et al., 2003; Broeckaert et al., 1999). Other tests using mediators measurable in exhaled air or breath condensate are also available to assess lung inflammation. Among these tests, one of the most validated is based on determination of nitric oxide in exhaled air (eNO). Nitric oxide produced by the inducible NO-synthase (iNOS) in bronchial epithelium is considered as a sensitive marker of proinflammation and oxidative stress in the lung (Barnes and Kharitonov, 1996; Kharitonov and Barnes, 2000; Kharitonov and Barnes, 2002; Saleh et al., 1998).

The aim of the present study was to evaluate whether exhaled NO can be used to monitor non invasively O_3-induced lung inflammation, by comparing morning and evening exhaled NO levels of children exposed to increasing concentrations of ambient O_3 during summer camps.

METHODS

Subjects and Study Design

A total of 72 children (17 girls) aged 6.5 to 15 years (mean, 10.5 years) were recruited for the study. All subjects were healthy (non asthmatic) and participated in the study after a written approval by their parents. The protocol of the study was approved by the Ethical Committee of the Faculty of Medicine of the University.

The study was conducted in six groups of children (n=11-15) attending different summer camps in rural areas in the South of Belgium between 26th July and 14th August 2002. Children were studied under varying O_3 exposure conditions that included cloudy days and episodes of photochemical smog. They were examined twice, first in the morning between 10 and 12 am prior to the ozone peak and a second time in the evening between 6 and 8 pm. Children remained outdoors doing normal outdoor recreational activities with no sport and running. Examination included a spirometric and an exhaled NO test and the measurement of height and body weight. The six groups of children were referred hereafter by a letter of A to F corresponding to increasing O_3 exposure levels.

Lung Function and Exhaled Nitric Oxide

The NO concentration in exhaled breath was measured on-line by chemiluminesence using the NIOX™ analyser (Aerocrine AB, Sweden). The test was performed in compliance with the guidelines of the American Thoracic Society (Slutzky et al., 1999). The FVC and FEV_1 were then measured with a Vitalograph-Compact (Vitalograph Ltd., Buckingham, England) according to the American Thoracic Society standards (ATS, 1995). All these tests were performed indoors to avoid confounding by weather conditions.

Air monitoring

Ambient O_3 was continuously monitored in each camp using an UV photometric O_3 analyser Model 427 (Signal Instrument Company Limited, England). The equipment was calibrated just before the beginning of study by the official laboratory of the national air monitoring network (Interregional cell for the environment, Belgium). Hourly concentrations of O_3 and other air pollutants (NO, NO_2, SO_2, $PM_{2.5}$ and PM_{10}) were obtained from the nearest local monitoring station. Temperature and relative humidity were also recorded hourly by nearest meteorological stations.

Statistical Analysis

All statistical tests were performed after checking the normality of data which were normalised by log transformation when necessary. Differences between morning and evening exhaled NO values were assessed by the paired Student's test (two-sided). One-way analysis of variance (ANOVA) followed by the Dunnett's multiple comparison test was used to compare the diurnal variations of exhaled NO or lung function parameters between the six studied groups exposed to increasing ambient O_3 levels. The associations between exhaled NO concentration and possible explanatory variables (age, height, weight, BMI, sex and ambient O_3 concentration) were tested using multiple stepwise regression. The benchmark dose (BMD) analysis of the increased of eNO was performed using the Weibull Model of the US-EPA Benchmark Dose Software, version 1.3.2. In this analysis, the percentages of children with an increased eNO were calculated using a cut off level of 4.30 ppb defined as the 95th percentile of the absolute diurnal eNO changes observed in children of groups A, B and C with the lowest exposures to O_3. Results were expressed as the mean ± standard error (SEM). The statistical package StatView® 5, Release 5.0.1, a business unit of SAS, third edition, Cary, NC: SAS Institute Inc., 2001 was used for all analyses. The level of significance was assigned at $p < 0.05$.

RESULTS

Meteorological data and concentrations of O_3 and other air pollutants during the six study days are summarised in Table 1. The maximum 1-hour concentrations of O_3 varied from 48 to 221 µg/m^3 and the 8-hour O_3 concentrations varied from 37 to 159 µg/m^3. Levels of other air pollutants remained low and stable, as illustrated in Figure 1 for NOx, even decreased during the studied days. Figure 1 also shows that O_3 concentrations peaked between approximately 14:00 and 18:00.

Table 1. Meteorological data and concentrations of air pollutants during the studied days.

	A	**B**	**C**	**D**	**E**	**F**
Date	07/26/2002	08/09/2002	08/14/2002	08/08/2002	07/29/2002	07/30/2002
Camp location	Pessoux	Marche-en-Famenne	Mozet	Marche-en-Famenne	Graide	Grandglise
Mean temperature (°C)	19.4	16.1	24.9	17.9	26.1	28.8
Max. temperature (°C)	22.1 (16 h)	17.1 (15 h)	26.4 (18 h)	20.4 (16 h)	27.8 (17 h)	32 (15 h)
Mean relative humidity (%)	77	81	64	75	40	50
Max. relative humidity (%)*	92 (10 h)	93 (10 h)	77 (10 h)	99 (10 h)	49 (10 h)	67 (10 h)
1-hour [O_3] (µg/m³)	48.3 (15 h)	71 (16 h)	96.2 (15 h)	127.3 (14 h)	166.6 (16 h)	221.2 (15 h)
8-hour [O_3] (µg/m³)	37.2	65.9	78.7	109.9	135	159
1-hour [NO] (µg/m³)*	10 (14 h)	5 (10 h)	43 (10 h)	3 (11 h)	3 (10 h)	6 (10 h)
8-hour [NO] (µg/m³)	5.3	3.1	10.9	2.2	2	3
1-hour [NO_2] (µg/m³)	26 (12 h)	23 (10 h)	42 (10 h)	13 (11 h)	6 (10 h)	35.3 (10 h)
8-hour [NO_2] (µg/m³)	20.1	13.7	19.4	9.8	4	17.4
1-hour [SO_2] (µg/m³)	4	4	8 (10 h)	4	4	17.6 (12 h)
8-hour [SO_2] (µg/m³)	4	4	4.9	4	4	11.6
8-hour PM 2.5 (µg/m³)	16.1	9.3	23.5	17.9	14.9	25.6
8-hour PM 10 (µg/m³)	17.9	14.8	44.9	30.1	34.6	47.6

*between brackets: time in the day corresponding to the maximum concentration (peak)

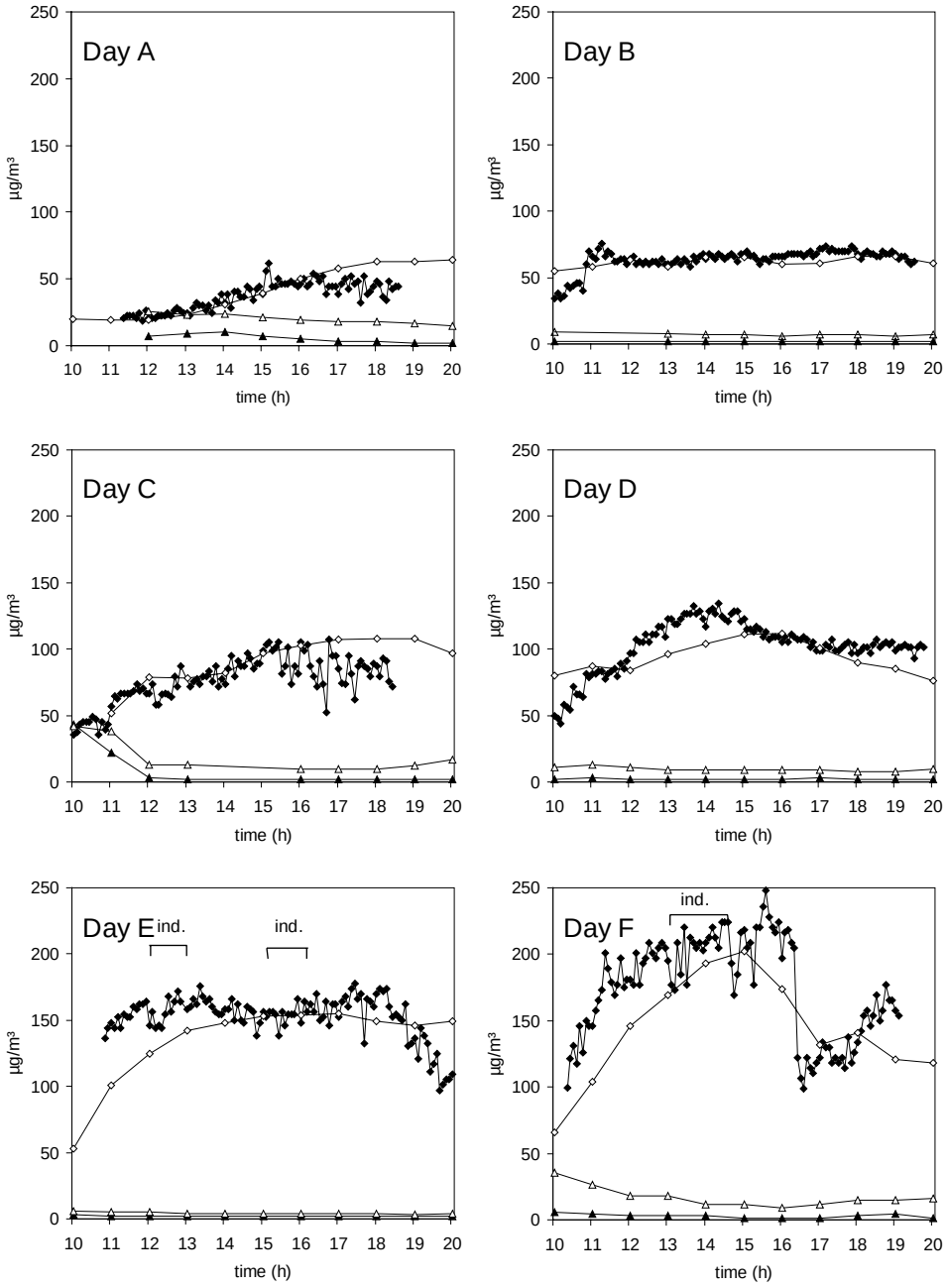


Figure 1. Diurnal variations of O_3 (dark diamonds for in-house measurements and white diamonds for station measurements), NO (dark triangles) and NO_2 (white triangles).

As shown in Table 2, the six groups were similar with regard to age and anthropometric parameters (ANOVA test, age, p = 0.4889; height, p = 0.0889; weight, p = 0.1461; BMI, p = 0.2016). Spirometric parameters (FVC, FEV_1 and FEV1/FVC ratio) did not show any consistent change with the increase of O_3 exposure levels. No significant difference in the diurnal variations of these lung function parameters was found between the studied groups by ANOVA. By contrast, as shown in Figure 2, the diurnal variation of eNO shows clearly distinct patterns between children exposed to low O_3 concentrations and those exposed to the highest O_3 levels. At O_3 levels below 100 µg/m^3, a statistically significant decrease of eNO is observed in the three studied groups (A, B

and C) whereas those exposed to the highest levels O_3 showed a marked elevation of their eNO levels (groups E and F). The breakpoint between the two patterns appears to lie around O_3 leve1s of 110 (8-hour) or 127 (1-hour) µg/m^3 when the diurnal decrease of eNO appears abolished by the rising O_3 exposure. No correlation was found between diurnal variations of exhaled NO and the concentrations of other air pollutants (PM_{10}, $PM_{2.5}$, SO_2, NOx).

Table 2. Clinical characteristics, lung function and exhaled NO concentrations of children in the morning and in the evening of the studied days. For spirometric parameters we discarded the results of three children (two in group A and one in group F) who did not perform properly the tests.

Group		**A**	**B**	**C**	**D**	**E**	**F**
Number, male/female		11/0	14/1	8/3	14/0	5/6	6/4
Age, yr		10.5 ± 0.2	10.6 ± 0.3	11.8 ± 0.9	9.7 ± 0.3	10.3 ± 0.7	11.1 ± 0.9
Height, m		1.46 ± 0.02	1.43 ±0.03	1.52 ± 0.05	1.37 ± 0.02	1.43 ± 0.04	1.43 ± 0.04
Weight, kg		34 ± 1	37 ± 2	43 ± 4	31 ± 1	34 ± 2	37 ± 4
BMI, kg/m²		15.8 ± 0.5	17.9 ± 0.5	17.9 ± 0.9	16.6 ± 0.5	16.4 ± 0.6	17.7 ± 1.5
FVC, % pred	AM	101 ± 3	97 ± 2	89 ± 3	98 ± 2	93 ± 3	91 ± 3
	PM	97 ± 4	102 ± 4	85 ± 4	90 ± 2*	97 ± 3†	88 ± 3
FEV_1, % pred	AM	90 ± 2	90 ± 3	85 ± 3	94 ± 3	91 ± 3	85 ± 3
	PM	86 ± 4	91 ± 3	79 ± 3‡	87 ± 3†	94 ± 3*	81 ± 3
FEV_1/FVC, % pred	AM	90 ± 1	93 ± 2	96 ± 2	97 ± 3	99 ± 2	94 ± 4
	PM	89 ± 3	94 ± 2	93 ± 2	96 ± 2	97 ± 2	92 ± 3
eNO, ppb	AM	13.7 ± 2.5	11.9 ± 1.7	14.2 ± 2.2	12.8 ± 2.8	9.1 ± 1.3	13.0 ± 2.5
	PM	12.1 ± 2.3*	9.9 ± 1.5‡	11.7 ± 2.0*	13.1 ± 2.5	29.2 ± 3.7‡	37.6 ± 2.1‡

Values are expressed as absolute numbers or as means ± SEM.
Significant difference between morning and evening values by paired Student't test: * $p < 0.05$; † $p < 0.01$; ‡ $p < 0.001$.

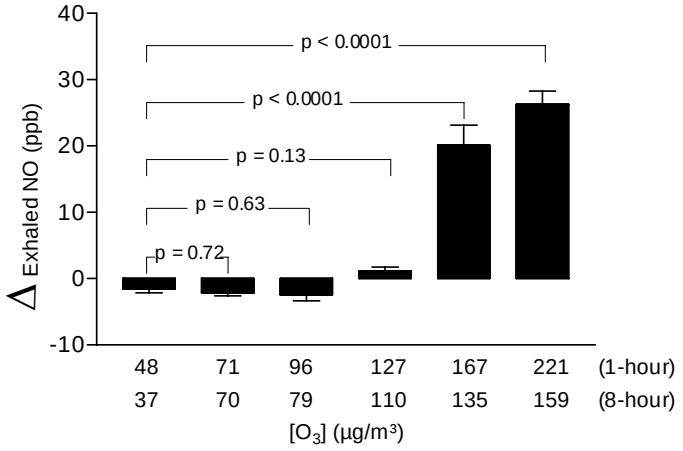


Figure 2. Diurnal variations of exhaled NO in children exposed to increasing concentrations of ambient ozone. Bars represent SE.

Multiple stepwise regression analysis shows that the increase of eNO induced by ambient O_3 in groups of children D, E and F is independent of age, sex and BMI but is only closely correlated with the maximal 1-hour ($r'' = 0.659$, $p < 0.0001$) or 8-hour O_3 concentrations ($r'' = 0.694$, $p < 0.0001$ for 8-hour). No predictor of the diurnal decrease of eNO in groups A, B and C could be identified. The O_3 threshold for increase of eNO was more precisely estimated by the calculation of the benchmark dose (BMD) and its lower-bound confidence limit (BMDL). A BMD of 134.5 µg/m^3 was derived for the maximal 1-hour O_3 concentration (BMDL, 119.2 µg/m^3) whilst the BMD for the maximal 8-hour O_3 concentration was estimated at 110.4 (BMDL, 101.5 µg/m^3).

DISCUSSION

The present study is the first to provide evidence of airways inflammation in children exposed to ambient ozone under field conditions. The inflammation induced by O_3 was detected by applying the eNO measurement, a non invasive test used to monitor inflammatory reactions in asthma and other lung diseases (Kharitonov and Barnes, 2002). At O_3 levels below 100 µg/m^3, the exhaled NO concentration shows a diurnal decrease confirming previous observations on the circadian variations of this indicator (Mattes et al., 2002). At higher O_3 concentration, the increase of eNO was not accompanied by lung function decrements, which is not really surprising given the relatively low O_3 levels in our study compared to previous reports and the well established lack of correlation between the inflammatory and functional responses of the lung to O_3 (Kinney et al., 1996; Balmes et al., 1996). These results are in agreement with the observations made in bleachery workers exposed to high peaks of O_3. Levels of eNO were found to be significantly increased among workers exposed to ozone gassings compared with those not exposed to such incidents. Like in our study, these changes were not associated with a significant decrease of FVC or FEV_1, suggesting that eNO is an early marker of airway inflammation produced by O_3 (Olin A.C. et al., 2004; Olin et al., 1999).

Although children were not exercising, the increase of eNO was observed from levels which are lower than current standards. The increase was already statistically significant from a maximum 1-hour concentration of 167 µg/m^3, which is lower than both the US and EU population information levels (235 and 180 µg/m^3, respectively, EU, 2002; US EPA, 1997). The corresponding 8-hour concentration was 135 µg/m^3, a concentration lower than the US National Ambient Air Quality Standard (8-hour ozone concentration of 157 µg/m^3). The BMD estimated (134 and 110 µg/m^3 for the 1-hour and 8-hour O_3 concentrations, respectively) from the dose-response relations suggest that the inflammatory response is triggered by even slightly lower concentrations. The 8-hour BMD is even lower than the air quality guideline recommended by the WHO and the European Union (120 µg/m^3).

In conclusion, the present study indicates that ambient O_3 induces in children an early lung inflammation that passes undetected with spirometric tests. This non invasive test, easily applicable under field conditions, represents an efficient tool which undoubtedly should improve the assessment of health risks of O_3 and the subsequent derivation of health-based air quality standards.

REFERENCES

ATS 1995. Standardization of Spirometry - 1994 Update. *Am. J. Respir. Crit. Care Med.* 152:1107-1136.

Balmes, J.R., Chen, L.L., Scannell, C., Tager, I., Christian, D., Hearne, P.Q., Kelly T., and Aris, R.M. 1996. Ozone-induced decrements in FEV1 and FVC do not correlate with measures of inflammation. *Am. J. Respir. Crit. Care Med.* 153:904-909.

Barnes, P.J., and Kharitonov, S.A. 1996. Exhaled nitric oxide: A new lung function test. *Thorax* 51:233-237.

Blomberg, A., Mudway, I., Svensson, M., Hagenbjork-Gustafsson, A., Thomasson, L., Helleday, R., Dumont, X., Forsberg, B., Nordberg, G., and Bernard, A. 2003. Clara cell protein as a biomarker for ozone-induced lung injury in humans. *Eur. Respir. J.* 22:883-888.

Broeckaert, F., Arsalane, K., Hermans, C., Bergamaschi, E., Brustolin, A., Mutti, A., and Bernard, A. 1999. Lung epithelial damage at low concentrations of ambient ozone. *Lancet* 353:900-901.

Bylin, G., Cotgraeve, I., Gustafsson, L., Nyber, G.F., Pershagen, G., Sundell, J., Victorin, K., and Zuber A. 1996. Health risk evaluation of ozone. *Scand. J. Work Environ. Health* 22:5-104.

Committee of the Environmental and Occupational Health Assembly of the American Thoracic Society 1996. Health effects of outdoor air pollution. *Am. J. Respir. Crit. Care Med.* 153:3-50.

EU Directive 2002/3/EC of the European Parliament and the Council of 12 February 2002 relating to ozone in ambient air. Off. J. L67, 14-30.

Kharitonov, S.A., and Barnes, P.J. 2000. Clinical aspects of exhaled nitric oxide. *Eur. Respir. J.* 16:781-792.

Kharitonov, S.A. and Barnes P.J. 2002. Biomarkers of some pulmonary diseases in exhaled breath. *Biomarkers* 7:1-32.

Kinney, P.L., Thurston, G.D., and Raizenne M. 1996. The effects of ambient ozone on lung function in children: A reanalysis of six summer camp studies. *Environ. Health Perspect.* 104:170-174.

Mattes, J., Storm Van's Gravesande, K., Moeller, C., Moseler, M., Brandis, M., and Kuerh, J. 2002. Circadian variation of exhaled nitric oxide and urinary eosinophil protein X in asthmatic and health children. *Pediatr. Res.* 51:190-194.

Olin, A.C., Andersson, E., Andersson, M., Granung, G., Hagberg, S., and Toren, K. 2004. Prevalence of asthma and exhaled nitric oxide are increased in bleachery workers exposed to ozone. *Eur. Respir. J.* 23:87-92.

Olin, A.C., Ljungkvist, G., Bake, B., Hagberg, S., Henriksson, L., and Toren, K. 1999. Exhaled nitric oxide among pulpmill workers reporting gassing incidents involving ozone and chlorine dioxide. *Eur. Respir. J.* 14:828-831.

Saleh, D., Ernst, P., Lim, S., Barnes, P.J., and Giaid, A. 1998. Increased formation of the potent oxidant peroxynitrite in the airways of asthmatic patients is associated with induction of nitric oxide synthase: effect of inhaled glucocorticoid. *Faseb J.* 12:929-937.

Slutzky, A.S., Drazen, J.M., Silkoff, P.E., Gaston, B.M., Holden, W., Romero, F.A., Alving, K., Baraldi, E., Barnes, P.J., Bratton, D., Chatkin, J.M., Cremona, G., De Gouw, H.W.F.M., Deykin, A., Djupesland, P., Douglas, J., Erzurum, S., Gustafsson, L. E., Haight, J., Hogman, M., Irvin, C., Joerres, R., Kissoon, N., Lanz, M.J., Lundberg, J.O.N., Massaro, A.E., Mehta, S., Olin, A., Permutt, Qian S.W., Robbins, R., Rubinstein, I., Sylvester, J.T., Townley, R., Weitzberg, E. and Zamel, N. 1999. Recommendations for standardized procedures for the online and offline measurement of exhaled lower respiratory nitric oxide and nasal nitric bride in adults and children - 1999. *Am. J. Respir. Crit. Care Med.* 160:2104-2117.

US EPA 1997. National Ambient Air Quality Standards for Ozone. Fed. Reg. 62, 38855-38896.

WHO 2000. Ozone and Other Photochemical Oxidants. In *Air Quality Guidelines for Europe*, Theakston, ed., 2nd ed. Copenhagen: WHO Regional Publications, European Series, No. 91.

PART III - MECHANISM OF EFFECT

LUNG PERMEABILITY, ANTIOXIDANT STATUS AND NO_2 INHALATION: A SELENIUM SUPPLEMENTATION STUDY IN RATS

C.Y. de Burbure, J.-F. Heilier,[1] J. Nève,[2] A. Becker, C. Albrecht, P.J.A. Borm,[3] J. Gromadzinska, W. Wasowicz, K. Rydzynski,[4] A.M. Bernard[1]

ABSTRACT

Little is known about antioxidant status, selenium status in particular, and lung response to NO_2 which acts as proinflammatory air pollutant. The effects of a low selenium diet (1.3 µg Se/day) with or without selenium supplementation were therefore studied in 128 two-month-old male Wistar rats exposed to either acute (50 ppm, 30 minutes), intermittent subacute (5 ppm, 6 h/d, 5 days), intermittent long-term NO_2 (1 ppm, 10 ppm, 6 h/d, 5 d/wk, 28 days) or normal atmospheric air (controls). Following sacrifice, measurements of lipid peroxidation (thiobarbituric acid reactive substances, chemiluminescence), antioxidative protective enzymes (glutathione peroxidase GPx, superoxide dismutase SOD, glutathione S-transferase GST, ceruloplasmin), lung damage (lactate dehydrogenase, alkaline and acid phosphatases), lung permeability (total protein, albumin), inflammation (cell populations), along with the determination of new biomarkers such as CC16 (Clara cell protein) were performed in serum and broncho-alveolar lavage fluid (BALF). While selenium supplemented animals had increased GPx activity in serum prior to inhalation experiments, they also had decreased BALF-CC16, blood-SOD and GST levels. Nevertheless, the protective role of normal selenium status with respect to NO_2 lung toxicity was evident both for long-term and acute exposures, as the increase in BALF-total proteins and corresponding decrease in serum (indicating increased lung permeability) was significantly more pronounced in selenium-deficient animals. During the various inhalation experiments, serum CC16 demonstrated its key role as an early marker of increased lung permeability. These findings corroborate the important role of selenium status in NO_2 oxidative damage modulation, but also indicate, in view of its negative impact on CC16, a natural anti-inflammatory and immunosuppressor, that caution should be used prior to advocating selenium supplementation.

[1] Université Catholique de Louvain, Brussels, Belgium.
[2] Université Libre de Bruxelles, Brussels, Belgium.
[3] Institut für Umweltmedizinische Forschung gGmbH (IUF), Düsseldorf, Germany.
[4] The Nofer Institute of Occupational Medicine, Lodz, Poland.

These studies are supported by the European Union (BIOART project IC15CT980336).

INTRODUCTION

Although the effects of NO_2 on the lung of both humans and animals have been widely documented, little is known about antioxidant status and particularly selenium status on the lung response to NO_2, which acts as a proinflammatory air pollutant. The objective of the present study was therefore to first evaluate the pulmonary effects of a low selenium diet (1.3 μg Se/day) with or without selenium supplementation in 128 two-month old male Wistar rats, then expose them to various concentrations of NO_2 and analyze the changes in lung response.

Nitrogen dioxide (NO_2) has become a major cause of concern not only as one of the most common man-induced air pollutants, but also more critically as a key mediator of the photochemical formation of ozone. This combustion product is mainly released into the environment in diesel exhaust fumes, tobacco smoke, by industries of nitric acid, fertilizers or explosives, or simply by cooking and heating with unventilated gas appliances. Its outdoor concentrations usually average 0.04-0.06 mg/m^3 i.e. 0.02-0.03 ppm, rising amidst dense traffic (0.05-0.1 ppm), while indoor concentrations average 0.02-0.08 mg/m^3, peak values being found in homes with gas stoves (average 0.5 ppm, peak values 1-2 ppm (Blomberg et al., 1999)). The occupational exposure limit (OEL) advised by the European Commission for NO_2 is 0.4 mg/m^3, i.e. 0.2 ppm (time weighted average TWA, 8h), its short-term exposure limit (STEL) being 1 mg/m^3 i.e. 0.5 ppm (TWA, 15 min). The WHO air quality guidelines currently recommend a maximal mean annual value of 40 $\mu g/m^3$ and a one-hour value of 200 $\mu g/m^3$ (WHO, 2000).

The pulmonary effects of NO_2 exposure are mainly reported on the lower respiratory tract, and are due to its capacity to react with water to produce HNO_3 and/or HNO_2, highly corrosive and irritant substances. Acute responses described vary from small dose-dependent reversible morphological changes in the trachea (reduced mucociliary transport, altered permeability, Kakinoki et al., 1998), bronchioles (changes in Clara cell shape, (Barth and Muller, 1999)) and alveoli (increased number of macrophages and type II epithelial cells (Chang et al., 1986), enhanced permeability (McElroy et al., 1997)) to important inflammatory responses resembling those of acute bronchitis at higher doses (influx of neutrophils and eosinophils, microvascular leakage, smooth muscle hyper-responsiveness (Papi et al., 1999)) and finally death from pulmonary edema reported in all species at much larger doses (Gray et al., 1954). A degree of morphological adaptation of the airways to oxidative stress has also been reported following prolonged exposure (Barth and Muller, 1999) or even repeated exposure (Blomberg et al., 1999) with the conclusion that NO_2 acts as a proinflammatory air pollutant.

The reported biochemical responses to NO_2 exposure have mainly concentrated on changes in lipid peroxidation and antioxidative protective systems, demonstrating wide interspecies differences in sensitivity to NO_2. Lipid peroxide (thiobarbituric acid reactant substances, TBARS) levels thus appear to be inversely related to the ability of the species to induce antioxidative protective factors in the lung such as the enzymes glutathione peroxidase, Glucose-6-phosphate dehydrogenase, Glutathione S-transferase, superoxide dismutase or ceruloplasmin and to regenerate reducing factors including non-protein sulfhydryls, Vitamin E and Vitamin C. Hence a direct interest in the potential protective role that antioxidant status may play in influencing the lung response to air pollutants such as NO_2 and ozone, which have indeed often been studied jointly (Ichinose et al., 1988; Ichinose and Sagai, 1989; Sagai et al., 1987; Sagai and Ichinose, 1991).

Interestingly, although the roles of vitamins A, C and E have already been investigated in the lung exposed to NO_2 (Dogra et al., 1983; Tom et al., 1985; Hatch et al., 1986; Sevanian et al., 1982), with the exception of one study on respiratory pentane (Dillard et al., 1980), the influence of selenium (Se) on the lung response to NO_2 exposure has, to the best of our knowledge, not yet been reported, although numerous selenium supplementation studies can be found in the literature. The antioxidant role of selenium is dependent on the action of three selenium-containing enzyme classes, cytosolic glutathione peroxidase, membrane-bound phospholipid hydroperoxide glutathione peroxidase and thioredoxin reductase, which in turn control the redox tone and the levels of cellular hydroperoxides and may also be responsible for the effects of selenium on the immune system. Its various and complex qualitative and quantitative effects on the cell cycle, viability, protein synthesis, DNA integrity and cellular metabolism are, however, mediated by a whole variety of other cellular selenium compounds which also interact with the antioxidant pathways (Spallholz, 2001). As selenium is already well known to play a protective role in vitro against oxidative damage to several organs (Chen and Tappel, 1995) as well as against apoptosis induced by superoxide anion (Guo et al., 2001) the hypothesis that selenium may play a protective role in vivo in the case of NO_2 exposure was therefore investigated in this inhalation study on Wistar rats, which are known to be able to induce antioxidant enzymes in response to NO_2.

The other innovative aspect of this study lies in the determination of the newly discovered and applied biomarker of lung response, the Clara cell secretory protein (CC16), which has been shown to be a major secretory product of human and animal bronchiolar Clara cells and controls lung sensitivity to pollutants (Bernard et al., 1992). CC16 is indeed a natural immunosuppressor and anti-inflammatory agent inhibiting both monocyte and polymorphonuclear neutrophil chemotaxis and phagocytosis *in vitro* (Lesur et al., 1995). It also appears to decrease the activity of interferon-γ, a potent multifunctional cytokine produced in the course of viral infections and inflammation (Dierynck et al., 1995) and its synthesis is stimulated by tumor necrosis factor-α (TNF-α) (Yao et al., 1998). As CC16 has already been shown to be a sensitive non-invasive biomarker of lung epithelium damage in case of ozone exposure (Broeckaert et al., 2000) we therefore included its study along standard measurements of lipid peroxidation (TBARS, chemiluminescence), antioxidative enzymes (Glutathione peroxidase, Superoxide Dismutase, Glutathione S-Transferase, Ceruloplasmin), lung damage (lactate dehydrogenase, alkaline and acid phosphatases), lung permeability (total protein, albumin) and inflammation markers (cell populations), both in serum and in broncho-alveolar lavage fluid (BALF).

MATERIALS AND METHODS

Study design

Following approval of the Ethical Committee of the Nofer Institute of Occupational Medicine, two month-old male Wistar rats (Outbred IMP, Wist, n=128) were divided at the start of the study into two separate groups and fed for 14 weeks either a special low selenium diet (group Se-, n=64) or the same low selenium diet supplemented with selenium as sodium selenite in drinking-water (Na_2SeO_3, 225 μg/l) *ad libitum* (group Se+, n=64). The low Se diet, containing 0.016 μg Se/g of fodder, was composed of bread (37%), cottage cheese (14%), sunflower oil (3%), vegetables (43.5%) and flax grain (2.5%). Fodder and water intake was checked daily. Estimated intake of the Se- group fed this diet was 1.3 μg Se/day, whereas the Se+ group animals were provided with 6.0 μg Se/day. Glutathione peroxidase (GPx) activity in erythrocytes was measured in blood at the beginning of the experiment and then at 3-4 week intervals for 14 weeks, as an indicator of Se status. Once a week the animal body weight gain was controlled. When GPx activity was stable and differentiated in both groups, NO_2 exposure began.

Groups of Se-normal and Se-deficient Wistar rats were placed in dynamic inhalation chambers and were exposed to 1 or 10 ppm NO_2 6 h/day, 5 days/week for 4 weeks (Groups I and II respectively, long-term exposures), 5 ppm for 6 h/day for 5 days (Group III, sub-acute exposure), 50 ppm for 30 minutes (Group IV, acute exposure) or used as controls breathing normal atmospheric air (Group V). Rats were sacrificed immediately after the experiment or 48 hours later in order to estimate recovery (see Table 1). Actual NO_2 concentrations were 1.04 ± 0.40 ppm, 5.45 ± 1.02 ppm, 10.40 ± 1.60 ppm and 45.55 ± 5.62 ppm. The control groups were fed either the low selenium or the selenium enriched diet. They were also kept in toxic chambers but breathed atmospheric air.

Table 1. Exposure conditions to NO_2 and Selenium (Se) status of animals in the inhalation study.

Exposure conditions to NO_2	**Groups**	**Total exposure to NO_2**		**Se-status**	**Recovery**
		ppm	**ppm.days**		
1 ppm, 28 days, 6 hrs/day, 5 d/wk	I (4x n=8)	28	5	Se+/Se-	0/48 hrs
10 ppm, 28 days, 6 hrs/day, 5 d/wk	II (4x n=8)	280	50	Se+/Se-	0/48 hrs
5 ppm , 5 days, 6 hrs/day	III (4x n=8)	25	6.25	Se+/Se-	0/48 hrs
50 ppm, 30 min	IV (2x n=8)	1.04	1.04	Se+/Se-	0 hrs
Controls	V (2x n=8)	0	0	Se+/Se-	0 hrs

Se-, diet with low selenium (1.3μg Se/day); Se+, diet with normal selenium content (6.0μg Se/day).

At the end of their exposure time, half of the rats of each exposed group were injected intraperitoneally with sodium pentobarbital (20 mg/250 g body weight) immediately after exposure and the other half 48 hours later in order to be able to evaluate potential recovery responses. The rats were exsanguinated via the abdominal aorta. Blood was collected into heparinised tubes, 200 μl of heparinised whole blood was immediately placed into a cold box away from sunlight and kept there no longer than 2 h (Habig et al., 1974) prior to analysis, while the remaining blood was centrifuged to separate plasma. In order to collect the bronchoalveolar lavage fluid (BALF)

from each rat, the trachea was cannulated, lungs were lavaged twice with 5 ml of 0.9% NaCl and lavage fluid collected from the 2 washes. BALF recovery in all investigated groups of animals was about 70%. 500 µl of BALF samples were stored in a cold box away from sunlight until analysis. Smears were prepared from BALF samples, stained according to May-Grünwald and Giemsa, and the total and differential cell counts were calculated with the use of light microscopy (Chapin, 1995). The rest of BALF samples were centrifuged at 600 g for 10 min to separate BALF cells. The remaining supernatant was stored at -20°C for further biochemical analysis which was conducted within six months of collection.

Biochemical analysis

Concentrations of nitric oxides in the inhalation chambers were measured using the colorimetric method after reduction of nitrates to nitrites, diazo reaction and conjugation of diazo compounds with chloride N-(1-naphthyl)-ethylenediamine to diazo dye. Selenium concentration was measured by graphite furnace atomic absorption spectrometry with Zeeman background correction (Neve et al., 1987). Quality assessment included participation in interlaboratory comparison trials and determination of commercial reference samples (Seronom trace elements from Sero AS).

The activity of GPx in BALF, plasma and red blood cell (RBC) lisate was measured using t-butylhydroperoxide as substrate (Paglia & Valentine, 1967). Superoxide dismutase (SOD) activity was determined with the use of xanthine oxidase and nitroblue tetrasolium (Beauchamp & Fridovich, 1971). Glutathione S-transferase (GST) activity was measured using 1-chloro2,4,-dinitrobenzene (Habig et al., 1974). Oxidase activity of ceruloplasmin (Cp) in BALF was measured by the spectrophotometric method (Sunderman, Jr. and Nomoto, 1970). The extent of lipid peroxidation was determined by measuring thiobarbituric acid reactive substances (TBARS) both in plasma and in BALF by the method optimized by Wasowicz et al. (1993).

The measurement of chemiluminescence (CL) was performed using a Luminometer 1251 linked to an IBM PC AT. After the addition of whole blood or BALF, CL was defined as the area under the emission curve as a function of time, calculated over 30 min (Kantorski and Tchorzewski, 1992). The value of CL was obtained from the following: measurement **x** value of hemoglobin / per 10^3 neutrophils in blood or macrophages in BALF.

Clara cell protein (CC16) was determined by latex immunoassay both in serum and in BALF (Halatek et al., 1998). Total protein and albumin, lactate dehydrogenase (LDH), acid phosphatase (ACP) and alkaline phosphatase (ALP) activities were determined in BALF (Technicon RA system, Bayer diagnostic, Domont, France), epithelial lining fluid volume was calculated as follows: ELF volume = total BALF urea/plasma urea.

Statistical analysis

Statistical analysis was performed with JMP, Version 5. (SAS Institute Inc., Cary, NC, 1989-2002). Data are described by mean and standard error on the mean. Data were checked for normality and by means of the Dixon Q test (Dixon, 1950; 1951), outliers were removed. Comparison between exposed and unexposed control animals was performed by Student t test. A systematic comparison of the groups for biomarkers of NOx exposure (BALF and serum CC16 level) was done in an ANOVA (one way), separating unexposed and NOx exposed and normal and Se-deficient rats (Sall et al., 2001). Statistical significance (*, $p<0.05$) was tested using ANOVA.

RESULTS

Effects of selenium status on lung parameters in control rats

The two different selenium concentrations in the fodder (Se-: 1.3 µg Se/day and Se+: 6 µg Se/day) affected the studied lung parameters in several ways. As expected and monitored during the 14 weeks prior to the inhalation studies, blood selenium concentrations rose significantly in parallel with GPx concentrations in plasma and red blood cells in those rats fed the 6 µg Se/ day diet compared with those fed the 1.3 µg Se/day (each $p<0.0001$). In BALF, however, GPx concentrations showed the reverse tendency, the low selenium control group tending to have higher levels than the normal selenium group, though not significantly so ($p=0.12$). Interestingly, some other parameters also varied unexpectedly between control groups fed a different selenium diet, such as for example the Clara cell secretory protein (CC16), and Alkaline phosphatase activity in BALF. The Clara cell protein showed a lower concentration in serum ($p<0.0001$) and a higher concentration in BALF in Se-animals. Alkaline phosphatase activity in BALF was also found to be lower in selenium deficient animals. Other

antioxidant parameters such as the zinc and copper dependent SOD enzyme also showed unexpected statistically significant variations in serum, increasing in Se- animals just as GST activity, demonstrating thereby compensatory opposite tendencies to GPx in serum (see Figure 1). None of the other studied parameters showed any impact of selenium status (plasma TBARS, BALF TBARS, total protein, albumin, ACP, LDH and Cp, CL and stimulated CL in blood and in BALF, hematocrit, hemoglobin, blood and BALF leukocyte, polymorphonuclear, neturophil, eosinophil, lymphocyte and monocyte counts, ELF and BALF volume).

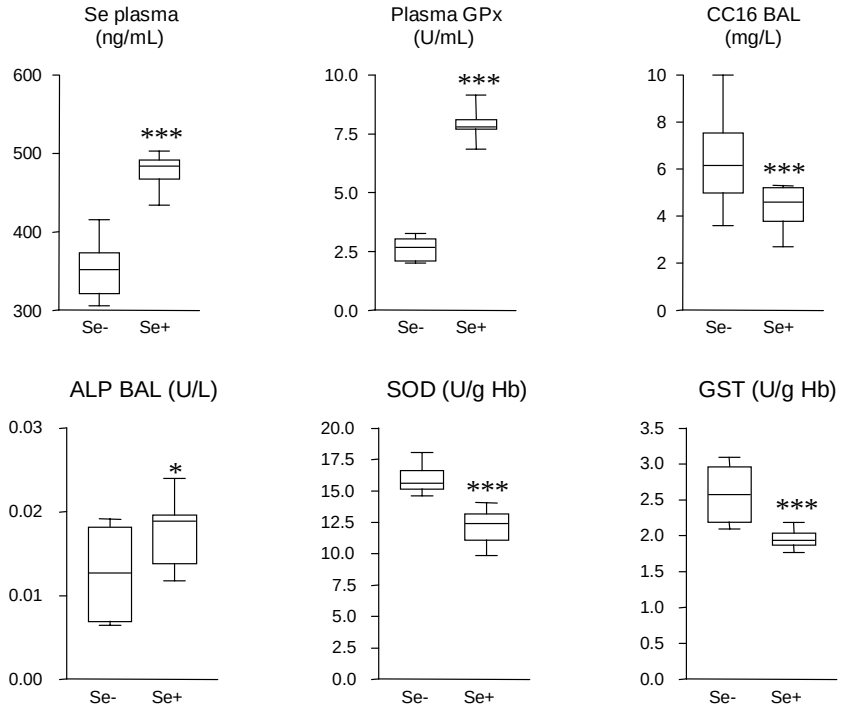


Figure 1. Changes in plasma and bronchoalveolar lavage (BAL) composition according to Selenium status in control rats (n=8 in each group). Comparison between Se+ and Se-: *: $p<0.05$; **: $p<0.01$; ***: $p<0.001$. Se: selenium (Se-: 1.3 µg Se/d, Se+: 6.0 µg/d), GPx glutathione peroxidase, CC16: Clara cell protein, ALP: alkaline phosphatase, SOD: superoxide dismutase, GST: glutathione S-transferase, Hb: hemoglobin.

NO_2 inhalation studies

The changes observed in three categories of studied end-points during the various inhalation experiments and their degree of statistical significance in comparison to control animals can all be found in Table 2.

A. Biomarkers of increased lung permeability and cellular damage

Total proteins and albumin. As illustrated in Figure 2, the increase in BALF-total proteins, indicating increased permeability of the lung epithelial barrier, was significantly more pronounced in Se- animals compared to rats receiving adequate selenium both for long-term exposure to 10 ppm and acute exposures to 50 ppm. As expected, changes in total protein and albumin concentrations were mainly and markedly significant for the acute exposure experiments at 50 ppm NO_2, increasing by a factor of 4 in both Se- and Se+ animals in the case of albumin, reflecting acute pulmonary damage and leakage of plasma into the airways. Interestingly, this did not seem to apply for the subacute exposure to 5 ppm, where total protein in BALF curiously actually decreased significantly in both Se- and Se+ animals, whereas albumin levels were only significantly reduced by 48 h ($p=0.0195$).

Table 2. Table of comparison to control values of the various parameters within each set of NO_2 inhalation. Increases compared to control values are indicated by +, decreases by -. NS: not statistically significant. Statistically significant variations are indicated as follows: +/-: $p<0.05$; ++/--: $p<0.01$; +++/---: $p<0.001$.

NO_2	1ppm (intermittent, 28 days)				5ppm (intermittent, 5 days)				10ppm (intermittent, 28 days)				50ppm (½hr)	
Diet	Se-		Se+		Se-		Se+		Se-		Se+		Se-	Se+
Vs. Ctl	Imm	48h	Imm	48h	Imm	48h	Imm	48h	Imm	48h	Imm	48h	Imm	Imm
A. Biomarkers of increased lung permeability and cellular damage														
CC16-S	NS	++	--	NS	+	+	NS	NS	+	++	---	NS	+	NS
CC16-BAL	NS	-	NS	++	---	NS	NS	NS	NS	NS	NS	+++	NS	NS
Tprot-BAL	NS	NS	NS	NS	--	NS	--	-	++	NS	NS	NS	+++	+++
Alb-BAL	NS	NS	NS	NS	NS	NS	NS	-	NS	NS	+	NS	+++	+
ACP-BAL	NS	NS	NS	NS	NS	NS	NS	NS	-	NS	NS	NS	---	NS
ALP-BAL	++	+++	+++	+	NS	NS	NS	NS	NS	NS	+	NS	+	NS
LDH-BAL	NS	NS	++	NS	NS	NS	NS	NS	NS	NS	++	NS	++	++
B. Selenium levels, antioxidant enzymes and oxidant production														
Se-Pl	--	NS	---	---	NS	NS	---	---	NS	NS	---	---	NS	---
GPx-Pl	NS	NS	---	---	NS	NS	---	---	+	NS	NS	+	NS	---
GPx-RBC	+++	+++	NS	+	++	+	++	+++	NS	+++	+++	+	+++	++
GPx-BAL	NS	NS	+++	+++	+++	NS	+++	+++	+++	+++	+++	+	+	+++
CP-BAL	+	NS	NS	NS	NS	NS	NS	NS	NS	+	NS	NS	NS	NS
SOD	+	---	+++	NS	---	---	NS	+++	--	NS	+++	NS	---	NS
GST	+	+++	+++	+++	NS	+++	+++	+++	NS	+	+++	+	NS	NS
CL-Bd	+	---	NS	NS	NS	+++	+++	NS	NS	NS	+	NS	NS	NS
CL-Bd+	NS	NS	NS	NS	NS	++	NS	NS	NS	NS	+	NS	NS	NS
CL-BAL	NS	+	-	NS	NS	+	NS	NS	NS	NS	+++	-	NS	NS
CL-BAL+	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	++	NS	NS	NS
TBAR-Pl	++	NS	++	-	NS	NS	NS	NS	NS	NS	NS	-	NS	NS
TBAR-BAL	NS	---	NS	NS	+++	NS	+++	---	NS	NS	++	---	NS	++
C. Parameters of blood and BALF cell populations														
Ht-Bd	NS	NS	---	-	NS	NS	---	NS	NS	NS	NS	---	-	NS
Hb-Bd	NS	NS	---	-	NS	NS	---	NS	NS	NS	NS	--	-	NS
Leu-Bd	NS	+	NS	NS	NS	--	NS	-	NS	NS	NS	-	-	NS
PMN-Bd	-	NS	NS	NS	NS	-	-	NS	NS	NS	NS	NS	NS	NS
Neu-Bd	NS	NS	NS	NS	NS	NS	NS	NS	-	-	NS	NS	NS	NS
Eosino-Bd	-	NS	NS	NS	NS	NS	NS	NS	NS	-	NS	NS	NS	NS
Lymph-Bd	++	NS	NS	NS	NS	+	+	NS	NS	NS	NS	NS	NS	NS
Mono-Bd	NS	NS	NS	NS	NS	NS	NS	NS	NS	-	NS	NS	NS	NS
Leu-BAL	NS	NS	NS	NS	NS	NS	NS	-	NS	NS	NS	NS	NS	NS
PMN-BAL	NS	NS	NS	NS	NS	NS	+	NS	NS	NS	NS	NS	NS	NS
Lymph-BAL	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	+	NS	NS	NS
Mono-BAL	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	-	NS	NS	NS

BAL: broncho-alveolar lavage, Bd: blood; Pl: plasma; RBC: red blood cells, CC16 (µg/l): Clara cell protein, Tprot (µg/ml): total protein in BALF; Alb (µg/ml): albumin in BALF; ACP (IU/l): acid phosphatase, ALP (IU/l): alkaline phosphatase, LDH (IU/l): lactate dehydrogenase, CL: chemiluminescence in area under curve, Se (ng/ml): Selenium; GPx (IU/ml): glutathione peroxidase, GST (IU/gHb): glutathione S-transferase, SOD (IU/gHb): superoxide dismutase, TBAR (nmol/ml): thiobarbituric acid reactive substances, Cp (mg/l): ceruloplasmin, Ht (%): hematocrit, Hb (g/dl): hemoglobin, Leu (10^6/ml): leukocytes, PMN (%): polymorphonuclear neutrophils, Neu (%): neutrophils; Lymph (%): lymphocytes; Mono (%): monocytes.

CC16. The changes in CC16 in both serum and BALF in rats exposed to intermittent 1 ppm NO_2 6 hrs/day, 5 days/week for 28 days can be visualized in Figure 3, with the statistically significant variations compared to control values and their exact p values. The significant decrease in BALF-CC16 was mirrored by an increase in serum CC16 in the Se- rats, whereas those rats with normal selenium levels actually showed the reverse tendency. These changes even continued to increase in Se- rats at 48 hrs post-inhalation, whilst rats with

sufficient selenium intake appeared to recover faster, as serum CC16 levels returned to normal at 48 hrs although production of CC16 still appeared increased in the lung according to the higher BALF concentrations. Similar significant variations in CC16 were to be found in the short term exposure experiments at 5 ppm for 6 hrs/day for 5 days. However, at higher intermittent chronic exposure to 10 ppm NO_2 for 28 days, CC16 variations in BALF in the Se- rats indicated a different type of response. Indeed, CC16 in BALF did not decrease as for 1 ppm or 5 ppm, it appeared to have adapted in the same way as in the Se+ rats, its concentration tending to remain the same if not higher than prior to the inhalation experiments, whilst still leaking significantly into serum. Interestingly, in the acute exposure to 50 ppm NO_2, CC16 only appeared very markedly in serum in Se- rats, indicating major leakage across the lung epithelial barrier, whilst its levels did not diminish in BALF. Two-way analysis of variance for CC16 in BALF in the control, 1 ppm and 10 ppm NO_2 inhalation experiments indicated that the effects of exposure to NO_2 took precedence ($p=0.0493$) over the effects of the selenium status ($p=0.5390$) in affecting the BALF-CC16 variations in the model, both immediately after inhalation and at 48h ($p=0.0166$ and $p=0.0537$ respectively).

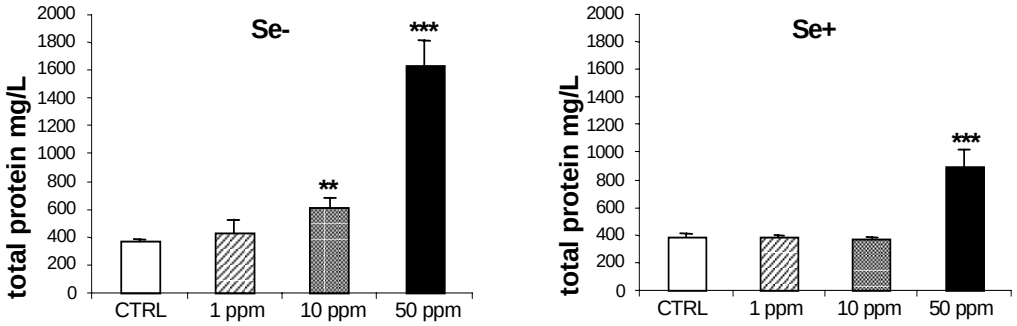


Figure 2. Total protein concentrations found in BALF immediately following NO_2 inhalation experiments at 1 ppm and 10 ppm NO_2 for 28 days (6 hrs/d, 5 d/week), and at 50 ppm NO_2 for 30 minutes, according to Selenium status (n=8 in each group except Se- 50 ppm n=6). Comparison to controls: *: $p<0.05$; **: $p<0.01$; ***: $p<0.001$.

CC16. The changes in CC16 in both serum and BALF in rats exposed to intermittent 1 ppm NO_2 6 hrs/day, 5 days/week for 28 days can be visualized in Figure 3, with the statistically significant variations compared to control values and their exact p values. The significant decrease in BALF-CC16 was mirrored by an increase in serum CC16 in the Se- rats, whereas those rats with normal selenium levels actually showed the reverse tendency. These changes even continued to increase in Se- rats at 48 hrs post-inhalation, whilst rats with sufficient selenium intake appeared to recover faster, as serum CC16 levels returned to normal at 48 hrs although production of CC16 still appeared increased in the lung according to the higher BALF concentrations. Similar significant variations in CC16 were to be found in the short term exposure experiments at 5 ppm for 6 hrs/day for 5 days. However, at higher intermittent chronic exposure to 10 ppm NO_2 for 28 days, CC16 variations in BALF in the Se- rats indicated a different type of response. Indeed, CC16 in BALF did not decrease as for 1 ppm or 5 ppm, it appeared to have adapted in the same way as in the Se+ rats, its concentration tending to remain the same if not higher than prior to the inhalation experiments, whilst still leaking significantly into serum. Interestingly, in the acute exposure to 50 ppm NO_2, CC16 only appeared very markedly in serum in Se- rats, indicating major leakage across the lung epithelial barrier, whilst its levels did not diminish in BALF. Two-way analysis of variance for CC16 in BALF in the control, 1 ppm and 10 ppm NO_2 inhalation experiments indicated that the effects of exposure to NO_2 took precedence ($p=0.0493$) over the effects of the selenium status ($p=0.5390$) in affecting the BALF-CC16 variations in the model, both immediately after inhalation and at 48h ($p=0.0166$ and $p=0.0537$ respectively).

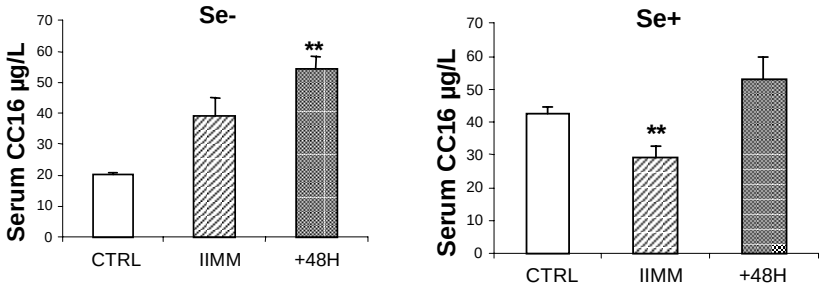


Figure 3. CC16 levels in BALF and serum following exposure to 1 ppm NO_2 for 28 days (6 hrs/d and 5 d/week) and according to Selenium diet. Comparison to controls: *: $p<0.05$; **: $p<0.01$; ***: $p<0.001$.

Lactate dehydrogenase (LDH), alkaline phosphatase (ALP) and acid phosphatase (ACP). Interestingly, none of these markers indicated significant changes during the subacute exposure experiment at 5 ppm in either sets of rats. While ACP decreases significantly at higher levels of exposure (10 & 50 ppm) in selenium deficient rats, curiously LDH only increases significantly in rats with normal selenium levels, except during the acute high level of exposure to 50 ppm where LDH rises significantly in both groups. ALP appears much more sensitive to NO_2 inhalation, increasing significantly in all rats at 1 ppm exposure levels, only in Se+ animals at 10 ppm and in Se- animals at 50 ppm.

B. Selenium levels, antioxidant enzymes and oxidant production

Selenium levels. Selenium levels were found to fall dramatically during NO_2 exposure in all acute, short or long-term inhalation studies, but curiously only in Se+ animals, receiving adequate amounts of selenium in their diet. These levels even continued to fall significantly over the next recovery period of 48 hrs following inhalation ($p<0.0001$ for all exposures compared to controls). In those rats on very low selenium diets, however, the levels only fell significantly for the least aggressive regimen, the long-term 1 ppm NO_2 exposure ($p=0.0081$). The exact reason for selenium levels to drop only in one group of rats and not the other would require further investigations, as in humans there appear to be no threshold values for selenium, some populations, for example in China, being known for their extremely low and almost non-existent selenium levels.

GPx in BALF, in plasma and in red blood cells. GPx levels rose in BALF in all animals during all experiments, strikingly more so in Se+ animals. The increases observed in GPx in BALF were mirrored in the Se+ animals by corresponding significant decreases in GPx in plasma at all experimental levels, implying leakage across the lung epithelial barrier of this selenodependent antioxidant in response to the oxidative stress. The Se- animals never quite reached the levels of GPx attained by selenium-supplemented rats, even in the 50 ppm acute exposure experiment, and these animals did not show any significant GPx changes in either BALF or serum at 1 ppm. GPx in RBC, however, increased significantly in almost all animals at all inhalation exposure levels.

Ceruloplasmin. Cp levels, which as we saw above, were unaffected by Selenium status in the controls, did however show some changes in response to the oxidative stress caused by NO_2 inhalation. This was particularly the case for the chronic experimental protocols, where Cp levels appeared elevated in Se- rats immediately following the intermittent 28-day exposure to 1 ppm and at 48 hrs post-exposure to 10 ppm, but not in Se+ rats, which maintained their levels constant in BALF.

SOD. Red blood cell superoxide dismutase activity showed statistically significant changes at all levels of NO_2 exposure, indicating thereby its sensitivity to oxidative stress. The changes were all consistent with a significant increase in activity in response to NO_2 inhalation at 1 ppm, 5 ppm and 10 ppm in the Se+ rats. Surprisingly, the response was quite the opposite in Se- rats. Indeed, except at 1 ppm where SOD increased, SOD dropped significantly in Se- animals at 5 ppm, 10 ppm and 50 ppm.

GST. Glutathione S-Transferase levels increased significantly in Se+ and Se- animals during the 1 ppm and 5 ppm exposure experiments and continued to rise for 48 hrs after exposure. The increase in GST levels was significantly greater in Se+ than in Se- rats immediately following exposure (1 ppm: Se- $p=0.0422$, Se+ $p<0.0001$; 5 ppm: Se- $p=0.0753$ while Se+ $p<0.0001$). The same effect of selenium status was observed at 10 ppm NO_2, where GST levels rose immediately post-inhalation in Se+ but not in Se- animals whilst all 48 h-levels

post-exposure were significantly raised. There were, however, no GST-level variations at 50 ppm acute NO_2 exposure.

TBARS. In plasma, TBARS rose significantly for both Se- and Se+ animals during the long-term 1 ppm inhalation studies, but at 48 h returned to normal (Se-) or significantly below control levels (Se+). There were no significant changes in plasma TBARS at 5 and 50 ppm, and only a significant decrease in Se+ rats immediately after exposure to 10 ppm for 28 days. In BALF, there were significant increases in TBARS at 5 ppm. The only other significant changes in TBARs in BALF were increases observed in Se+ rats at 10 ppm and 50 ppm.

Chemiluminescence (CL). Both spontaneous and phorbol-myristate acetate (PMA)-stimulated oxidant release were measured in BALF-cells and blood leukocytes. The responses observed to the various levels of oxidative stress appeared to move in similar ways for both Se- and Se+ rats, demonstrating an overall increase in oxidant formation in response to NO_2. However, if oxidant formation measured in blood significantly increased after 28 days 1 ppm NO_2 inhalation then rapidly decreased over 48 hours in Se- and not in Se+ animals (increase $p=0.0380$, 48h decrease $p<0.0001$), at 5 and 10 ppm exposures it only rose in Se+ rats ($p=0.0147$ and $p=0.0206$ respectively), returning to normal levels at 48 hours whilst Se- rats then showed a significant increase in oxidant formation ($p=0.0006$). The increases in both blood and BALF CL observed at 50ppm did not reach statistical significance in either Se+ or Se- rats. Increases in BALF CL were significant for Se+ animals during the long-term exposures at 1 ($p=0.0121$) and 10 ppm ($p=0.0008$) but not at 5 ppm, whilst they were only significant in Se- animals at 48 h post-exposure to 1 ($p=0.0217$) and 5 ppm (0.0123). Stimulated CL variations were not significant.

C. Parameters of blood and BALF cell populations

Results in polymorphonuclear counts (PMN) were interesting, particularly following short-term 5 ppm NO_2 inhalation, where PMN count was reduced in blood, and correspondingly increased in BALF for both groups of rats, significantly so in Se+ rats (decrease in blood PMN $p=0.0481$; increase in BALF PMN $p=0.0490$). For Se+ rats, however, the PMN count showed signs of recovering, both in BALF and in blood, 48 hrs post-exposure, while still diminishing from the circulation in Se- rats. Another interesting finding concerned the number of lymphocytes recovered in BALF which increased in Se- rats following 5 ppm NO_2 exposure ($p=0.0535$) and in Se+ rats following the chronic 10 ppm exposure experiments ($p=0.0133$). There were no consistent findings that would otherwise suggest a role of selenium on cell counts and differentials in animals exposed to NO_2.

DISCUSSION

Selenium diet. When assessing daily amounts of fodder consumed by rats, daily Se intake in the Se- group was estimated at 1.3 μg Se/day/animal. According to the literature, a truly Se-deficient diet provides <0.12 μg Se/day/animal, so the animals used in this experiment were not entirely deficient in selenium but simply on a low Se diet. A Selenium-free diet is based on Torula yeast, cultivated on special selenium-free cultures, as a main source of protein (Beilstein & Whanger, 1988), while a standard Murigran diet for rodents contains 0.176 μg Se/g/fodder and provides rats with about 3.5 μg Se/day. According to the literature data this amount of selenium is recognized as appropriate and thus used in most experiments (Jenkinson et al., 1987). The range of Se concentrations applied in experiments with rats is very wide, and the supply of even 20 μg Se/day falls within limits of correct values. However, at such high doses of selenium, concentration and activity of selenoproteins do not increase proportionally to the applied dose (Sun et al., 1998). The changes observed on the various lung parameters in this study in control rats are most interesting and demonstrate the importance of appropriate selenium supply in the diet. Furthermore, most markers studied indicated a positive protective effect to NO_2 in Se+ compared to Se- animals. The unexpected decreased CC16 secretion in BALF observed in Se+ rats, further corroborated in the human studies of the BIOART project (unpublished data), indicates however that one should exert caution when advocating selenium supplementation and that further studies on the complex role played by selenium on the lung and the Clara cell in particular are recommended.

Lung permeability markers. Changes in total protein and albumin in BALF clearly indicate an enhanced permeability of the lung epithelial barrier, known to increase following NO_2 inhalation studies (McElroy et al., 1997). Selenium status, however, also had considerable effects on the concentrations observed in the various groups: as clearly shown in Figure 1, Se- rats had a statistically significant increase in BALF-total protein concentrations at 10 ppm, when this was not the case for the Se+ rats. Moreover, the mean total protein in BALF observed in Se+ rats were much lower than those observed in Se- animals, indicating a greater protection of their

epithelial barrier to oxidative stress. Interestingly, this did not seem to apply for the subacute experimental conditions, where total protein increased in both Se+ and Se- animals, indicating increased permeability of the lung epithelium to some proteins at least, such as demonstrated by the mirrored responses in serum and BALF of CC16. Concerning CC16, it is remarkable to notice how the changes in BALF were mostly mirrored by those in serum, confirming the increased permeability (Hermans and Bernard, 1999). The unchanged total protein and albumin concentrations in BALF during the chronic 1 ppm experiment, whilst CC16 moved from BALF into serum, demonstrated once more the selective increase in permeability changes previously described which characterize CC16 as an early marker of increased permeability (Broeckaert et al., 2000). At 10 ppm exposure, the higher CC16 levels in BALF in both Se+ and Se- animals could sign an adaptative response to the higher oxidative stress (increased CC16 production, increased Clara cell numbers) with or without increased cell destruction and spilling of CC16 contents. These results are consistent with the findings reported by Barth and Muller, 1999 who described in rats maximal proliferation of the bronchiolar epithelium following short term exposure to 5 ppm (three day exposure), when long term exposure to 5 ppm did not increase epithelial proliferation. Exposure to higher doses NO_2 long term (10 and 20 ppm for 25 days) increased bronchial and bronchiolar cell proliferation in a linear dose-response fashion (Barth and Muller, 1999). However, the increased serum CC16 levels indicate that the epithelial barrier still has an increased permeability, as it had for 1 and 5 ppm, as evidenced by the countercurrent of total protein leakage into BALF in Se- rats.

Why the CC16 levels are not diminished in BALF at 50 ppm could be explained by the acute exposure circumstances to the high level of NO_2 and hence by the destruction of the epithelial and Clara cells in the respiratory tree, inducing raised levels of CC16 originating from the destroyed Clara cells. These changes could indeed be linked to the morphological report by Kawakami et al. (1989) describing almost total loss of apical projections of Clara cells in terminal bronchioles following inhalation of 50 ppm for 5 hours. Although the other epithelial cells showed signs of recovery from day 3 onwards, new cilia and secretory granules reappearing on day 5 in major bronchi, day 7 in terminal bronchioles, the Clara cells did not return to normal over the next seven days of intermittent high level exposure experiments (Kawakami et al., 1989). The responses of selenium-supplemented rats with regard to CC16 levels all confirm that selenium appears to have some protective effect on the Clara cells and their production of the antioxidant CC16 protein. As seen from the recovery period results at 48 h, they also appear to recover faster than Se- rats. However, these interesting protective effects, which warrant further investigations, do not prevent injury by very high doses NO_2 such as the 50 ppm burst exposure.

Indicators of cellular damage. The above results confirm alkaline phosphatase to be one of the most sensitive markers of pulmonary damage, as previously reported (Pauluhn, 2000). The decrease in acid phosphatase observed at 50 ppm had also been noted by Pauluhn who found it associated to increased phophatidylcholine in alveolar macrophages, linking it to protracted lysosomal catabolism due to increased phospholipid phagocytosis from oxidative damage to the airways epithelium (Pauluhn, 2000).

Antioxidants and oxidant formation. It is obvious from the results that selenium status influenced GPx levels in response to NO_2 inhalation. Concerning the 1ppm long-term exposure of Se- rats to NO_2, GPx levels in BALF are not significantly raised, yet selenium in plasma is significantly decreased. Although GPx in plasma has not moved either, GPx in red blood cells has increased significantly and one hypothesis could be that selenium is being used for its synthesis, as part of the long-term adaptative response one month after the start of the inhalation studies, corroborated by its noted increase at all exposure levels.

Ceruloplasmin results were most interesting, showing marked differences according to selenium levels, rising mostly in Se- rats at 1 ppm long-term NO_2 exposures. This could be interpreted, in the light of the unchanged total protein and albumin in BALF and the raised alkaline phosphatase levels, as an argument in favor of the theory that Cp does indeed stem from respiratory epithelial cells, as the increase noted does not correspond to an increase in plasma leakage across the alveolocapillary barrier at this level of exposure. Ceruloplasmin has indeed an even larger molecular weight than albumin (132kDa vs 68kDa) and its increase in BALF following NO_2 exposure and oxidative stress is therefore corroborating the hypothesis that it could indeed originate from within the airway and be synthesized and released by epithelial cells. These findings in rats cannot be extrapolated to humans, as there is to date no evidence to the synthesis and release of Ceruloplasmin by human respiratory epithelial cells (Baker et al., 2000). Results concerning SOD levels also appear to indicate it can be upregulated and adapted over longer periods of exposure, and that selenium may play a part in this regulatory mechanism. On the other hand, the GST post-exposure results at 48h indicate that this isoenzyme family takes a while to be induced in response to oxidative stress, and that its induction is also favored by a healthy selenium status, the

precise mechanism of which would also need further investigations as those described in the lung, the alpha isoenzymes, are supposed to have selenium-independent GPx activity (Morrison et al., 1998).

Unexpectedly, while a certain degree of protection from selenium status was evident in the chemiluminescence results at 1 and 10 ppm, the increases in TBARs in BALF were clearly more marked in Se+ than Se- animals at all exposure levels (acute 50 ppm, subacute 5 ppm and chronic 10 ppm NO_2), except for the 1 ppm exposure levels where TBARs were only raised in plasma. These findings do not seem to match others to be found in the literature, such as the reports by Ichinose and Sagai (Ichinose et al., 1988; Ichinose & Sagai, 1989; Sagai et al., 1987; Sagai & Ichinose, 1991), Sevanian's studies concerning Vitamin E (Sevanian et al., 1982) or Tom's report on vitamin A status (Tom et al., 1985) who all agree that when GPx levels in the lung are decreased there is a corresponding increase in lipid peroxidation, in particular in the presence of oxidative stress. However, measurements of TBARs in those reports were made on lung homogenates, so any comparison should only be made with caution. NO_2 is thought to act directly by inducing peroxidation on the lipid cell membranes through oxidative stress, and increased TBAR levels can therefore imply either a greater rate of lipid peroxidation, or a lesser catabolism of the molecules, the mechanism of which is complex and studied elsewhere (Giulivi et al., 1995). Why the changes are more marked in selenium-supplemented animals for the higher doses of exposure is unclear and requires further investigations. It could also simply be a reflection of a degree of aspecificity of TBAR measurements, which are well recognized (Moore & Roberts, 1998).

CONCLUSIONS

These findings confirm the important role of antioxidant status, and selenium status in particular, in the lung's own defense mechanisms such as the integrity of the alveolo-capillary barrier and the production of anti-oxidants, among others CC16, when challenged by air pollutants such as NO_2.

However, in view of the unexpected decreased CC16 levels in BALF observed in selenium-supplemented rats, further corroborated in the human studies of the BIOART project, one should exert caution when advocating selenium supplementation.

Further studies on the complex interactions of selenium and CC16 in their protective antioxidant properties in the lung are required as these would shed light on the important fields of lung disease, asthma and allergy.

REFERENCES

Baker, C.S., Evans, T.W., and Haslam, P.L. 2000. Measurement of ceruloplasmin in the lungs of patients with acute respiratory distress syndrome: is plasma or local production the major source? *Respiration* 67:533-538.

Barth, P.J., and Muller, B. 1999. Effects of nitrogen dioxide exposure on Clara cell proliferation and morphology. *Pathol. Res. Pract.* 195:487-493.

Beauchamp, C., and Fridovich, I. 1971. Superoxide dismutase: improved assays and an assay applicable to acrylamide gels. *Anal. Biochem.* 44:276-287.

Beilstein, M.A., and Whanger, P.D. 1988. Glutathione peroxidase activity and chemical forms of selenium in tissues of rats given selenite or selenomethionine. *J. Inorg. Biochem.* 33:31-46.

Bernard, A., Marchandise, F.X., Depelchin, S., Lauwerys, R., and Sibille, Y. 1992. Clara cell protein in serum and bronchoalveolar lavage. *Eur. Respir. J.* 5:1231-1238.

Blomberg, A., Krishna, M.T., Helleday, R., Soderberg, M., Ledin, M.C., Kelly, F.J., Frew, A.J., Holgate, S.T., and Sandstrom, T. 1999. Persistent airway inflammation but accommodated antioxidant and lung function responses after repeated daily exposure to nitrogen dioxide. *Am. J. Respir. Crit Care Med.* 159:536-543.

Broeckaert, F., Arsalane, K., Hermans, C., Bergamaschi, E., Brustolin, A., Mutti, A., and Bernard, A. 2000. Serum clara cell protein: a sensitive biomarker of increased lung epithelium permeability caused by ambient ozone. *Environ. Health Perspect.* 108:533-537.

Chang, L.Y, Graham, J.A., Miller, F.J., Ospital, J.J., and Crapo, J.D. 1986. Effects of subchronic inhalation of low concentrations of nitrogen dioxide. I. The proximal alveolar region of juvenile and adult rats. *Toxicol. Appl. Pharmacol.* 83:45-61.

Chapin, K. 1995. Clinical microscopy. In Manual of Clinical Microbiology 6th ed., eds. P.R. Murray, E.J. Baron, M.A. Pfaller, F.C. Tenover, and R.H. Yolken, pp. 33-51. Washington, DC: American Society for Microbiology.

Chen, H., and Tappel, A.L. 1995. Vitamin E, selenium, trolox C, ascorbic acid palmitate, acetylcysteine, coenzyme Q, beta-carotene, canthaxanthin, and (+)-catechin protect against oxidative damage to kidney, heart, lung and spleen. *Free Radical Res.* 22:177-186.

Dierynck, I., Bernard, A., Roels, H., and De Ley, M. 1995. Potent inhibition of both human interferon-gamma production and biologic activity by the Clara cell protein CC16. *Am. J. Respir. Cell Mol. Biol.* 12:205-210.

Dillard, C.J, Sagai, M., and Tappel, A.L. 1980. Respiratory pentane: a measure of in vivo lipid peroxidation applied to rats fed diets varying in polyunsaturated fats, vitamin E, and selenium and exposed to nitrogen dioxide. *Toxicol. Lett.* 6:251-256.

Dixon, W.J. 1950. Analysis of extreme values. *Ann. Math. Stat.* 21:488-506.

Dixon, W.J. 1951. Ratios involving extreme values. *Ann. Math. Stat.* 22:68-78.

Dogra, S.C., Khanduja, K.L., Gupta, M.P., and Sharma, R.R. 1983. Effect of vitamin A deficiency on pulmonary and hepatic protective enzymes in rat. *Acta Vitaminol. Enzym.* 5:47-52.

Giulivi, C., Lavagno, C.C., Lucesoli, F., Bermudez, M.J., and Boveris, A. 1995. Lung damage in paraquat poisoning and hyperbaric oxygen exposure: superoxide-mediated inhibition of phospholipase A2. *Free Radical Biol. Med.* 18:203-213.

Gray, E.L., Patton, F.M., Goldberg, S.B., and Kaplan, E. 1954. Toxicity of the oxides of nitrogen. II. Acute inhalation toxicity of nitrogen dioxide, red fuming nitric acid, and white fuming nitric acid. *AMA. Arch. Ind. Health.* 10:418-422.

Guo, L., Xue, A.N., Wang, S.Q., Chen, J.Y., Wu, Y.D., and Zhang, B. 2001. Induction of apoptosis by superoxide anion and the protective effects of selenium and Vitamin E. *Biomed. Environ. Sci.* 14:241-247.

Habig, W.H., Pabst, M.J., and Jakoby, W.B. 1974. Glutathione S-transferases. The first enzymatic step in mercapturic acid formation. *J. Biol. Chem.* 249:7130-7139.

Halatek, T., Hermans, C., Broeckaert, F., Wattiez, R., Wiedig, M., Toubeau, G., Falmagne, P., and Bernard, A. 1998. Quantification of Clara cell protein in rat and mouse biological fluids using a sensitive immunoassay. *Eur. Respir. J.* 11:726-733.

Hatch, G.E., Slade, R., Selgrade, M.K., and Stead, A.G. 1986. Nitrogen dioxide exposure and lung antioxidants in ascorbic acid-deficient guinea pigs. *Toxicol. Appl. Pharmacol.* 82:351-359.

Hermans, C., and Bernard, A. 1999. Lung epithelium-specific proteins: characteristics and potential applications as markers. *Am. J. Respir. Crit. Care Med.* 159:646-678.

Ichinose, T., Arakawa, K., Shimojo, N., and Sagai, M. 1988. Biochemical effects of combined gases of nitrogen dioxide and ozone. II. Species differences in lipid peroxides and antioxidative protective enzymes in the lungs. *Toxicol. Lett.* 42:167-176.

Ichinose, T., and Sagai, M. 1989. Biochemical effects of combined gases of nitrogen dioxide and ozone. III. Synergistic effects on lipid peroxidation and antioxidative protective systems in the lungs of rats and guinea pigs. *Toxicology* 59:259-270.

Jenkinson, S.G., Spence, T.H. Jr., Lawrence, R.A., Hill, K.E., Duncan, C.A., and Johnson, K.H. 1987. Rat lung glutathione release: response to oxidative stress and selenium deficiency. *J. Appl. Physiol.* 62:55-60.

Kakinoki, Y., Ohashi, Y., Tanaka, A., Washio, Y., Yamada, K., Nakai, Y., and Morimoto, K. 1998. Nitrogen dioxide compromises defence functions of the airway epithelium. *Acta Oto-Laryngol*. 538:221-226.

Kantorski, J., and Tchorzewski, H. 1992. The effect of serine and thiol protease inhibitors on the chemiluminescence of human neutrophils in investigations in vitro. *J. Biolumin. Chemilumin.* 7:37-45.

Kawakami, M., Yasui, S., Yamawaki, I., Katayama, M., Nagai, A., and Takizawa, T. 1989. Structural changes in airways of rats exposed to nitrogen dioxide intermittently for seven days. Comparison between major bronchi and terminal bronchioles. *Am. Rev. Respir. Dis.* 140:1754-1762.

Lesur, O., Bernard, A., Arsalane, K., Lauwerys, R., Begin, R., Cantin, A., and Lane, D. 1995. Clara cell protein (CC-16) induces a phospholipase A2-mediated inhibition of fibroblast migration in vitro. *Am. J. Resp. Crit. Care Med*. 152:290-297.

McElroy, M.C., Pittet, J.F., Allen, L., Wiener-Kronish, J.P., and Dobbs, L.G. 1997. Biochemical detection of type I cell damage after nitrogen dioxide-induced lung injury in rats. *Am. J. Physiol.* 273:L1228-L1234.

Moore, K., and Roberts, L.J. 1998. Measurement of lipid peroxidation. *Free Radical Res.* 28:659-671.

Morrison, R.J., Singhal, S.S., Bidani, A., Heming, T.A., and Awasthi, S. 1998. Glutathione S-transferases of rabbit lung macrophages. *Toxicol. Appl. Pharmacol.* 148:229-236.

Neve, J., Chamart, S., and Molle, L. 1987. Optimization of a direct procedure for the determination of selenium in plasma and erythrocytes using Zeeman effect atomic absorption spectroscopy. In Analytical Chemistry in Medicine and Biology, eds. P. Bratter, and P. Schramel, pp. 349-358. Berlin: Walter de Gruyter.

Paglia, D.E., and Valentine, W.N. 1967. Studies on the quantitative and qualitative characterization of erythrocyte glutathione peroxidase. *J. Lab. Clin. Med.* 70:158-169.

Papi, A., Amadesi, S., Chitano, P., Boschetto, P., Ciaccia, A., Geppetti, P., Fabbri, L.M., and Mapp, C.E. 1999. Bronchopulmonary inflammation and airway smooth muscle hyperresponsiveness induced by nitrogen dioxide in guinea pigs. *Eur. J. Pharmacol.* 374:241-247.

Pauluhn, J. 2000. Acute inhalation toxicity of polymeric diphenyl-methane 4,4'-diisocyanate in rats: time course of changes in bronchoalveolar lavage. *Arch. Toxicol.* 74:257-269.

Sagai, M., Arakawa, K., Ichinose, T., and Shimojo, N. 1987. Biochemical effects on combined gases of nitrogen dioxide and ozone. I. Species differences of lipid peroxides and phospholipids in lungs. *Toxicology* 46:251-265.

Sagai, M., and Ichinose, T. 1991. Biochemical effects of combined gases of nitrogen dioxide and ozone. IV. Changes of lipid peroxidation and antioxidative protective systems in rat lungs upon life span exposure. *Toxicology* 66:121-132.

Sall, J., Lehman, A., and Creighton, L. 2001. JMP start statistics: A guide to statistical and data analysis using JMP and JMP IN software. Duxburry, Pacific Grove, CA.

Sevanian, A., Hacker, A.D., and Elsayed, N. 1982. Influence of vitamin E and nitrogen dioxide on lipid peroxidation in rat lung and liver microsomes. *Lipids* 17:269-277.

Spallholz, J.E. 2001. Mechanisms for the carcinostatic activity of Se compounds. In Selenium and Prevention of Cancer, ed. Y Palmieri, pp. 1-12. Grimbergen, Belgium: Selenium-Tellurium Development Association.

Sun, Y., Ha, P.C., Butler, J.A., Ou, B.R., Yeh, J.Y., and Whanger, P. 1998. Effect of Dietary Selenium on Selenoprotein W and Glutathione Peroxidase in 28 Tissues of the Rat. *J. Nutr. Biochem.* 9:23-27.

Sunderman, F.W. Jr., and Nomoto, S. 1970. Measurement of human serum ceruloplasmin by its p-phenylenediamine oxidase activity. *Clin. Chem.* 16:903-910.

Tom, W.M., Prasongwatana, V., and Boyde, T.R. 1985. The effects of vitamin A nutritional status on glutathione levels and microsomal lipid peroxidation in rat lung. *Experientia* 41:1046-1047.

Wasowicz, W., Neve, J., and Peretz, A. 1993. Optimized steps in fluorometric determination of thiobarbituric acid-reactive substances in serum: importance of extraction pH and influence of sample preservation and storage. *Clin. Chem.* 39:2522-2526.

WHO. 2000. *Air quality guidelines for Europe*. Edited by WHO. 2nd ed. 91 vols., *European series*. Genève: WHO Regional Publications.

Yao, X.L., Levine, S.J., Cowan, M.J., Logun, C., and Shelhamer, J.H. 1998. Tumor necrosis factor-alpha stimulates human Clara cell secretory protein production by human airway epithelial cells. *Am. J. Respir. Cell Mol. Biol.* 19:629-635.

CHROMOSOMAL ABERRATIONS BY FLUORESCENCE IN SITU HYBRIDIZATION (FISH) – BIOMARKER OF EXPOSURE TO CARCINOGENIC PAHS

Olena Beskid, Blanka Binkova, Zdík Dusek, Pavel Rössner,[1] Ivan Kalina,[2] Todor A. Popov,[3] Peter B. Farmer,[4] Radim J.Srám[1]

ABSTRACT

The fluorescence in situ hybridisation (FISH) technique with whole chromosome painting for chromosomes #1 and #4 was used to study the impact of air pollution containing higher concentrations of carcinogenic polycyclic aromatic hydrocarbons (c-PAHs) in three European cities, Prague (Czech Republic), Kosice (Slovakia) and Sofia (Bulgaria). In each site an exposed group was followed consisting of police officers or bus drivers who work usually through busy streets for at least 8 h, and a control group, who spent more than 90% of their daily time indoors. In Prague, a significant increase was observed in all the studied endpoints in the police officers compared to the control population (P<0.05). This difference is most apparent between exposed and control nonsmokers (F_G/100=1.56±1.34 vs. 1.14±1.02, P<0.05). In Kosice, the exposed group differed from controls in the endpoint percentage of aberrant cells (% AB.C) (0.29±0.19 vs. 0.21±0.20, P<0.05) and t/1000 (3.91±3.14 vs. 2.84±3.10, P<0.05). In Sofia two exposed groups were followed: police officers and bus drivers. FISH endpoints were significantly higher in police officers compared to controls (F_G/100=1.60±0.99 vs. 0.82±0.79, P<0.01). In bus drivers compared to controls there was an increase in % AB.C. (0.25±0.18 vs.0.13±0.13, P<0.01) and t/1000 (3.24±2.28 vs. 2.13±2.05, P<0.05). This is the first study using the FISH method to analyse the impact of environmental air pollution. According to the original hypothesis it is expected that the most important group of chemicals responsible for the biological activity of air pollution represent c-PAHs.

[1] Academy of Sciences of the Czech Republic, Prague, Czech Republic.
[2] Safarik University, Kosice, Slovak Republic.
[3] National Center of Hygiene, Sofia, Bulgaria.
[4] University of Leicester, Leicester, U.K

INTRODUCTION

Prospective cohort studies have shown that prolonged exposure to particulate air pollution may be associated with an increased rate of morbidity and mortality from respiratory and cardiovascular diseases in the general population (Dockery et al., 1993; Pope et al., 1995; 2002). High exposure is associated with an increased risk of cancer (Katsouyanni and Pershagen, 1997), and the presence of a wide variety of genotoxic compounds in environmental air pollution has been demonstrated. Binkova et al. (1999) observed that extracts of particulate matter were able to preferentially produce PAH-DNA adducts in calf thymus DNA. Distinct aromatic DNA adducts derived from carcinogenic polycyclic aromatic hydrocarbons (c-PAHs, benz[a]anthracene, benzo[b]fluoranthene, benzo[k]fluoranthene, benzo[a]pyrene (B[a]P), benzo[g,h,i]perylene, chrysene, dibenzo[a,h]anthracene and indeno[1,2,3-cd]pyrene) accounted for approximately 50% of the total radioactivity detected. Dejmek et al. (2000) demonstrated that the effect of exposure to particulate matter on intrauterine growth retardation (IUGR) may be at least partly explained by the presence of c-PAHs. It was also shown that IUGR is positively related to PAH-DNA adducts in placentas (Sram et al., 1999). Perrera et al. (1999) showed in studies in Poland that ambient air pollution was significantly associated with PAH-DNA adducts in both maternal and cord white blood cells. Newborns with elevated PAH-DNA adducts had significantly decreased birth weight compared to newborns with lower DNA adducts. All these results indicate that in some regions c-PAHs may be a major source of the genotoxic activity of organic mixtures associated with air pollution.

The genotoxicity of air pollution has repeatedly been observed using biomarkers of exposure. DNA adducts measured by a ^{32}P-postlabeling method have become the most popular of these biomarkers of exposure. For biomarkers of effect, the conventional cytogenetic analysis of peripheral blood lymphocytes has been accepted as a technique suitable for the monitoring of genetic damage in somatic cells since the early 1970s. Today, chromosomal aberrations in human peripheral lymphocytes are recognized as internationally standardized and validated biomarkers of effect (Carrano and Natarajan, 1988; Albertini et al., 2000). This method was widely used to evaluate the impact of occupational exposure to carcinogens (Sram et al., 2004a) and now it is generally accepted that a high frequency of chromosomal aberrations in peripheral lymphocytes is predictive of an increased risk of cancer (Hagmar et al., 1998; Bonassi et al., 2000; Smerhovsky et al., 2001).

The fluorescence in situ hybridisation (FISH) technique became available for public health purposes substantially later, in the middle of 1990s. In comparison with conventional cytogenetic analyses, which detect particularly unstable types of aberrations, FISH using whole chromosome painting was developed as a rapid and sensitive method of detecting structural rearrangements, especially reciprocal translocations (Swiger and Tucker, 1996; Pressl and Stephan, 1998). The FISH technique detects translocations, which are long lasting injuries likely transferred through many cell cycles. This means, that the FISH technique measures the type of chromosomal changes related to cancer and that these changes may circulate in peripheral blood lymphocytes (PBL) for a long period of time.
The FISH painting methods have been often applied in the field of ionising radiation research (Natarajan et al., 1996; Lindholm et al., 1998; Matsumoto et al., 1998). As far as the exposures to chemical carcinogens, the available data are scant. Only few reports were published on the effect of occupational exposure to carcinogens (Sram et al., 2004b).

We used the FISH technique with whole chromosome painting for chromosomes #1 and #4 to study the impact of air pollution containing higher concentrations of c-PAHs in three European cities, Prague (Czech Republic), Kosice (Slovakia) and Sofia (Bulgaria). In each site an exposed group was followed consisting of policemen or bus drivers, who work usually through busy streets for at least 8 h, and controls, who spent more than 90% of their daily time indoors.

MATERIALS AND METHODS

Study subjects

The study was undertaken in three European cities, Prague (Czech Republic), Kosice (Slovakia) and Sofia (Bulgaria). The exposed group in each site were male police officers (and in Sofia also bus drivers), who spent > 8 h outdoors. The control group was matched by age, gender and length of employment, spending > 90% of daily time indoors. The populations were followed in winter, as the highest exposure to PAHs occurs during this season at the selected sites. The personal monitoring was supplemented with data from HiVol samples from stationary air pollution monitors in those cities.

Air sample collection, extraction of EOM and chemical analysis

Personal monitoring was provided using a personal sampler from the U.S. Environmental Protection Agency (U.S. EPA, 1999; Watts et al., 1994). Respirable particles smaller than 2.5 μm were collected on Teflon-impregnated glass fiber filters. The sampler was connected to a pump and operated continuously during a shift (8-12 h). Particles from the filter were extracted (EOM, extractable organic matter) for quantitative PAH analysis.

PM_{10} air particles (particles <10 μm) were collected for three months at each city in the period when the population studies took place (during the months of biological sample collections and two months before). The samples were collected daily for a 24 h period with the exception of determinations of high particle concentrations in winter, when the sampling period was shortened to 12 h. HiVol air samplers (Anderson) equipped with 20 x 20 cm Pallflex filters (T60A20) were used. The filters with samples collected at the same location were pooled together (Binkova et al., 2003).

Quantitative chemical analysis of PAHs from all three cities was performed by HPLC with fluorimetric detection in the laboratories of the certified company Ecochem, a.s., Prague (EN ISO CSN IEC 17025). Ecochem also carried out the extraction of the filters. All procedures were performed according to US EPA methods (U.S. EPA, 1999). The carcinogenic PAHs (c-PAHs) analysed were benz[a]anthracene, benzo[b]fluoranthene, benzo[k]fluoranthene, benzo[a]pyrene (B[a]P), benzo[g,h,i]perylene, chrysene, dibenzo[a,h]anthracene and indeno[1,2,3-cd]pyrene.

The following levels of air pollution were recorded during the study period : Prague – from HiVol sampling PM_{10} 62.6 μg/m^3, c-PAHs 24.7 ng/m^3, B[a]P 3.50 ng/m^3, from personal monitoring for exposed population at the days of sampling c-PAHs 12.04±11.10 ng/m^3, B[a]P 1.79±1.67 ng/m^3, for controls c-PAHs 6.17±3.48, B[a]P 0.84±0.60 ng/m^3; Kosice - from HiVol sampling PM_{10} 58.0 μg/m^3, c-PAHs 11.9 ng/m^3, B[a]P 1.37 ng/m^3, from personal monitoring for exposed population at the days of sampling c-PAHs 21.72±3.12 ng/m^3, B[a]P 2.94±1.44 ng/m^3, for controls c-PAHs 6.39±1.56, B[a]P 1.07±0.66 ng/m^3; Sofia - from HiVol sampling PM_{10} 89.9 μg/m^3, c-PAHs 36.4 ng/m^3, B[a]P 4.84 ng/m^3, from personal monitoring for exposed population at the days of sampling c-PAHs 45.19±25.41 ng/m^3, B[a]P 4.31±2.60 ng/m^3 (policemen) and c-PAHs 34.46±33.22 ng/m^3, B[a]P 5.40±3.18 ng/m^3 (bus drivers), for controls c-PAHs 24.14±16.82 ng/m^3, B[a]P 1.96±1.53 ng/m^3 (Farmer et al., 2003).

Cell cultivation

Whole venous blood cultures were established within 24 h after blood collection in sodium-heparinised tubes, according to the method described by Sorsa et al. (1994). Lymphocyte cultures were set up in the tissue culture flasks, each culture containing 0.6 ml of whole blood and 7.5 ml of medium (cultivation medium for one culture was composed of RPMI 1640 Sevac 1.06 ml, calf serum Opavac 1.80 ml, distilled water 4.24 ml, glutamine 0.10 ml, $NaHCO_3$ (7.5%) 0.16 ml, PHA HA-15 Murex, U.K. 0.10 ml).

The cultures for the FISH were cultivated at 37^{o}C and harvested after 72 h of incubation to obtain a sufficient number of mitoses. Colchicine (Fluka) was added to a final concentration of 0.5 μg/ml 2 h before the end of the incubation. The cells were collected by centrifugation, re-suspended in pre-worm (37^0 C) hypotonic solution (0.075 M KCl) for 10 min and fixed in acetic acid/methanol according to the standard protocol The cell suspensions were stored at -20^0 C. Slides were always prepared using the air-dry method on the day before painting by FISH, randomly numbered and after the painting scored “blind” in numerical order.

FISH

Fresh slides were prepared by dropping the fixed pellet of metaphase cells onto slides, which were stored in ethanol with 1 % ether at -4ºC. The protocol used to perform the FISH with whole chromosome probes for chromosomes #1 and #4 was adapted from the protocol provided by Cambio (Cambridge, UK). The counterstain, following the washes, was DAPI mixed with mountant Vectashield (Vector Laboratories, Burlingame, CA) in final concentration 0.24 μg/ml.

Analysis of FISH slides was performed using a Zeiss microscope equipped with a triple filter for simultaneous visualization of DAPI (blue), FITC (green) and Cy-3 (red) signals (Rubes et al. 1998). Color images were

collected using a computer-controlled Zeiss axioskop with monochrome CCD camera (JAI Corporation, Japan) and ISIS 4.4.16 software (MetaSystems GmbH, Germany). 1000 metaphases per subject were analyzed.

All aberrant cells were classified according to the Protocol for Aberration Identification and Nomenclature (PAINT) (Tucker et al., 1995). The Protocol defines translocations as follows: translocation (t) is a rearranged chromosome with a single centromere and is to be counted as an aberration; the translocated chromosome must exhibit at least two colors, reciprocal translocation (rcp) is the exchange of genetic material between the two chromosomes of a different color, dicentric chromosome (dic) contains two centromeres from the chromosomes of different colors, acentric fragment (ace) is a linear part of red or green painted chromosome without centromere, insertion (ins) is an acentric chromosomal material inside the chromosome of another color. Other analyzed parameters were percentage of aberrant cells (%AB.C.) and the number of color junctions (NCJ). All cells with color junction or painted acentric fragment were counted as aberrant cells.

Genomic frequencies (F_G) of stable chromosome exchanges were calculated according to Lucas et al. (1993) using the equation : $F_G = F_{rg} /2.05 [f_r (1-f_r) + f_g (1 - f_g) -f_r f_g]$. F_{rg} is the translocation frequency measured by FISH after two-color painting, f_r and f_g are the fractions of the genome painted red and green, respectively.

Subjects with suspected clonality were excluded from the final evaluation.

Statistical analysis

The Student t-test was used to analyze the differences between the groups. The influence of independent variables (like c-PAHs exposure, age and smoking) on FISH endpoints was estimated by multiple regression model.

RESULTS

According to personal monitoring, the air pollution during the 8 h shift before the blood sampling to PAHs and B[a]P seems to be in the order Sofia>Kosice>Prague. As air pollution may vary daily, the exposure during the last three months characterized from HiVol sampling seems to be more important for the relationship between c-PAHs and biomarker of effect. Then the exposure in three cities is in the order Sofia>Prague>Kosice.

In Table 1 are summarized all the cytogenetic endpoints for exposed and control groups divided further according to smokers and nonsmokers. Probably the most important endpoints are $F_G/100$, % AB.C. and t/1000 (translocations per 1000 cells).

In Prague was observed a significant increase in all studied endpoints in the police officers ($P<0.05$). This difference is the most apparent between exposed and control nonsmokers ($F_G/100=1.56\pm1.34$ vs. 1.14 ± 1.02, % AB.C.$=0.29\pm0.20$ vs. 0.22 ± 0.16, t/1000$=4.21\pm3.61$ vs. 3.05 ± 2.73, $P<0.05$, respectively) (Figure 1). A similar effect of air pollution was not observed in smokers.

In Kosice the exposed group differed from controls in the endpoints % AB.C. (0.29 ± 0.19 vs. 0.21 ± 0.20, $P<0.05$) and t/1000 (3.91 ± 3.14 vs. 2.84 ± 3.10, $P<0.05$). No differences were observed between exposed and control nonsmokers (Figure 2) or exposed and control smokers.

In Sofia two exposed groups were followed: police officers and bus drivers. FISH endpoints were significantly higher in police officers compared to controls ($F_G/100=1.60\pm0.99$ vs. 0.82 ± 0.79, % AB.C.$=0.25\pm0.14$ vs. 0.13 ± 0.13, t/1000$=4.19\pm2.65$ vs. 2.13 ± 2.05, $P<0.01$, respectively). All these parameters were also significantly higher in exposed nonsmokers (Figure 3, $P<0.05$) as well as in exposed smokers ($P<0.05$). In bus drivers compared to controls were increased % AB.C. (0.25 ± 0.18 vs.0.13 ± 0.13, $P<0.01$) and t/1000 (3.24 ± 2.28 vs. 2.13 ± 2.05, $P<0.05$). In bus drivers nonsmokers vs. control nonsmokers all endpoints were significantly increased (Figure 3, $P<0.05$ or $P<0.01$). No difference was observed between smokers from both groups.

Comparing three cities, there was no difference between Prague and Kosice, all groups, nonsmokers and smokers, respectively. Similarly Kosice and Sofia did not differ. Comparing Prague and Sofia, FISH % AB.C. was higher in Prague in exposed smokers (0.39 ± 0.32 vs. 0.26 ± 0.14), in control smokers (0.40 ± 0.23 vs. 0.17 ± 0.13) and control nonsmokers (0.22 ± 0.16 vs. 0.09 ± 0.13, $P<0.05$ respectively) and t/1000 in control smokers (5.50 ± 4.04 vs. 2.69 ± 1.93, $P<0.05$).

Table 1. Effect of air pollution on chromosomal aberrations in peripheral lymphocytes.

Site	Group	N	Age	F_G/100	% AB.C.	t	rcp	dic	ace	ins	NCJ
Prague	Exposed	50	31.8	1.72±1.57*	0.33±0.25*	4.62±4.21*	1.74±1.84*	0.16±0.42	0.72±1.20	0.12±0.39*	5.16±4.62**
	SM	17	33.3	2.02±1.95	0.39±0.32	5.41±5.21	1.88±2.06	0.18±0.53	1.12±1.69	0.18±0.53	6.29±5.72
	NS	33	31.0	1.56±1.34*	0.29±0.20*	4.21±3.61*	1.67±1.74*	0.15±0.36	0.52±0.80	0.09±0.29	4.58±3.91*
	Controls	48	29.4	1.25±1.11	0.24±0.18	3.35±2.99	1.21±1.32	0.08±0.35	0.44±0.68	0.00±0.00	3.42±2.96
	SM	6	35.5	2.05±1.51	0.40±0.23	5.50±4.04	2.17±1.72	0.00±0.00	0.83±0.75	0.00±0.00	5.50±4.04
	NS	42	28.5	1.14±1.02	0.22±0.16	3.05±2.73	1.07±1.22	0.10±0.37	0.38±0.66	0.00±0.00	3.12±2.71
Kosice	Exposed	47	32.0	1.52±1.17	0.29±0.19*	3.91±3.14*	1.26±1.21	0.30±0.59*	0.98±1.59	0.09±0.28	4.70±3.55*
	SM	23	32.7	1.69±1.17	0.33±0.21*	4.43±3.12	1.39±1.20	0.26±0.45	1.30±1.92	0.00±0.00	5.13±3.91
	NS	24	31.2	1.36±1.18	0.26±0.16	3.42±3.15	1.13±1.23	0.33±0.70	0.67±1.17	0.17±0.38	4.29±3.20
	Controls	45	35.3	1.12±1.30	0.21±0.20	2.84±3.10	0.93±1.16	0.13±0.34	0.56±0.94	0.11±0.32	3.33±3.41
	SM	21	37.1	1.22±1.48	0.21±0.18	2.95±3.22	0.90±1.09	0.14±0.36	0.67±1.06	0.10±0.30	3.48±3.60
	NS	24	33.8	1.03±1.14	0.21±0.22	2.75±3.05	0.96±1.23	0.13±0.34	0.46±0.83	0.13±0.34	3.21±3.31
Sofia	Police-men	26	31.3	1.60±0.99**	0.25±0.14**	4.19±2.65**	1.46±1.07**	0.08±0.27	0.23±0.65	0.08±0.27	4.42±2.86
	SM	17	31.5	1.69±1.06*	0.26±0.14*	4.47±2.83*	1.65±1.06*	0.12±0.33	0.24±0.75	0.00±0.00	4.59±2.94
	NS	9	31.0	1.42±0.88*	0.23±0.13*	3.67±2.35*	1.11±1.05	0.00±0.00	0.22±0.44	0.22±0.44	4.11±2.85
	Drivers	25	37.7	1.22±0.85	0.25±0.18**	3.24±2.28*	1.24±1.01*	0.28±0.68	0.64±0.99	0.04±0.20	3.64±2.83
	SM	15	38.3	1.11±0.79	0.25±0.20	2.93±2.12	1.07±0.88	0.40±0.83	0.80±1.08	0.00±0.00	3.47±2.92
	NS	10	36.8	1.38±0.95*	0.26±0.16**	3.70±2.54*	1.50±1.18*	0.10±0.32	0.40±0.84	0.10±0.32	3.90±2.81
	Controls	23	38.4	0.82±0.79	0.13±0.13	2.13±2.05	0.70±0.76	0.09±0.29	0.22±0.60	0.09±0.29	2.35±2.46
	SM	13	40.4	1.04±0.74	0.17±0.13	2.69±1.93	0.85±0.80	0.08±0.28	0.31±0.75	0.08±0.28	3.00±2.55
	NS	10	35.8	0.54±0.79	0.09±0.13	1.40±2.07	0.50±0.71	0.10±0.32	0.10±0.32	0.10±0.32	1.50±2.17

* P < 0.05, ** P < 0.01, N – number of subjects, SM – smokers, NS – nonsmokers, F_G/100 – genomic frequency of translocations/100 cells, % AB.C. – percentage of aberrant cells, t – number of translocations/1000 cells, rcp – number of reciprocal translocations/1000 cells, dic – number of dicentric chromosomes/1000 cells, ace – number of acentric fragments/1000 cells, ins – number of incertions/1000 cells, NCJ – number of color junctions/1000 cells

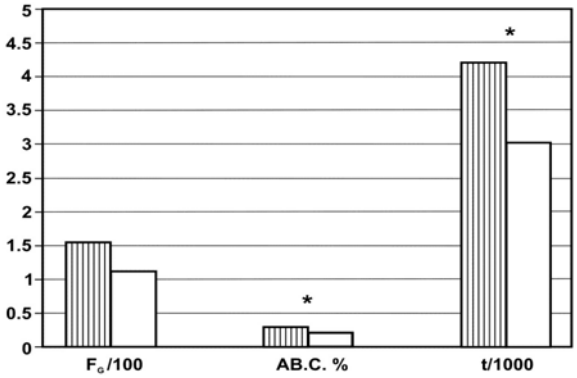


Figure 1: Effect of air pollution determined by FISH on nonsmokers in Prague, Czech Republic. (P < 0.05; shaded columns – police officers; empty columns – controls; F_G/100 – genomic frequency of translocations/100 cells; % AB.C. – percentage of aberrant cells; t – number of translocations/1000 cells).

According to the FISH results genetic injury of chromosomes seems to be higher in Prague than in Sofia; Kosice does not differ from these two cities.

Multiple regression analysis (Table 2) indicates the significant effect of exposure to c-PAHs at Kosice and Sofia, of age in Prague and Sofia, and of smoking in Prague, on FISH endpoints.

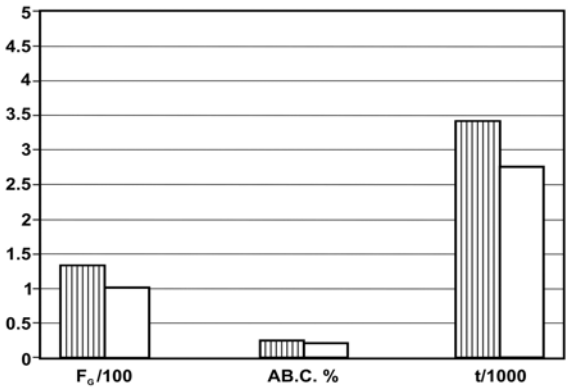


Figure 2: Effect of air pollution determined by FISH on nonsmokers in Kosice, Slovakia. (P < 0.05; shaded columns – police officers; empty columns – controls; F_G/100 – genomic frequency of translocations/100 cells; % AB.C. – percentage of aberrant cells; t – number of translocations/1000 cells).

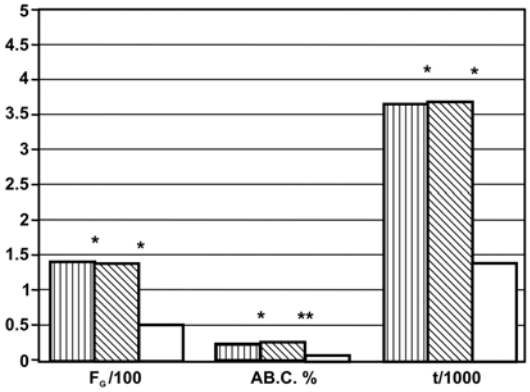


Figure 3: Effect of air pollution determined by FISH on nonsmokers in Sofia, Bulgaria. (P < 0.05; oblique shaded columns – police officers; shaded columns – bus drivers; empty columns – controls; F_G/100 – genomic frequency of translocations/100 cells; % AB.C. – percentage of aberrant cells; t – number of translocations/1000 cells).

Table 2. The results of multivariate analysis for FISH endpoints.

Site	FISH parameter	Intercept	Exposure	Age	Smoking status
Prague	F_G/100	1.459	0.215 (p=0.39)	**0.078 (p=0.0000)**	0.312 (p=0.30)
	% AB.C	0.259	0.047 (p=0.26)	**0.010 (p=0.0001)**	***0.089 (p=0.0743)***
	t	3.913	0.598 (p=0.37)	**0.210 (p=0.0000)**	0.813 (p=0.31)
Kosice	F_G/100	0.915	**0.572 (p=0.0245)**	**0.053 (p=0.0009)**	0.136 (p=0.58)
	% AB.C	0.187	**0.111 (p=0.0047)**	**0.009 (p=0.0002)**	0.004 (p=0.92)
	t	2.361	**1.502 (p=0.0198)**	**0.133 (p=0.0010)**	0.283 (p=0.65)
Sofia	F_G/100	0.741	**0.562 (p=0.0191)**	-0.000 (p=0.99)	0.194 (p=0.36)
	% AB.C	0.117	**0.113 (p=0.0055)**	-0.000 (p=0.94)	0.038 (p=0.28)
	t	1.952	**1.505 (p=0.0180)**	-0.008 (p=0.83)	0.515 (p=0.36)

F_G/100 – genomic frequency of translocations/100 cells, %Ab.C – percentage of aberrant cells, t- number of translocations/1000 cells.

When we analyzed the impact of exposure to B[a]P (as the representative of c-PAHs, groups of nonsmokers) to F_G/100 in all three cities, the outcome was similar for data/group by personal monitoring (Figure 4 A) or HiVol sampling (Figure 4 B). But using data from individual personal monitoring, the effect of B[a]P exposure to F_G/100 is not very significant (Figure 5).

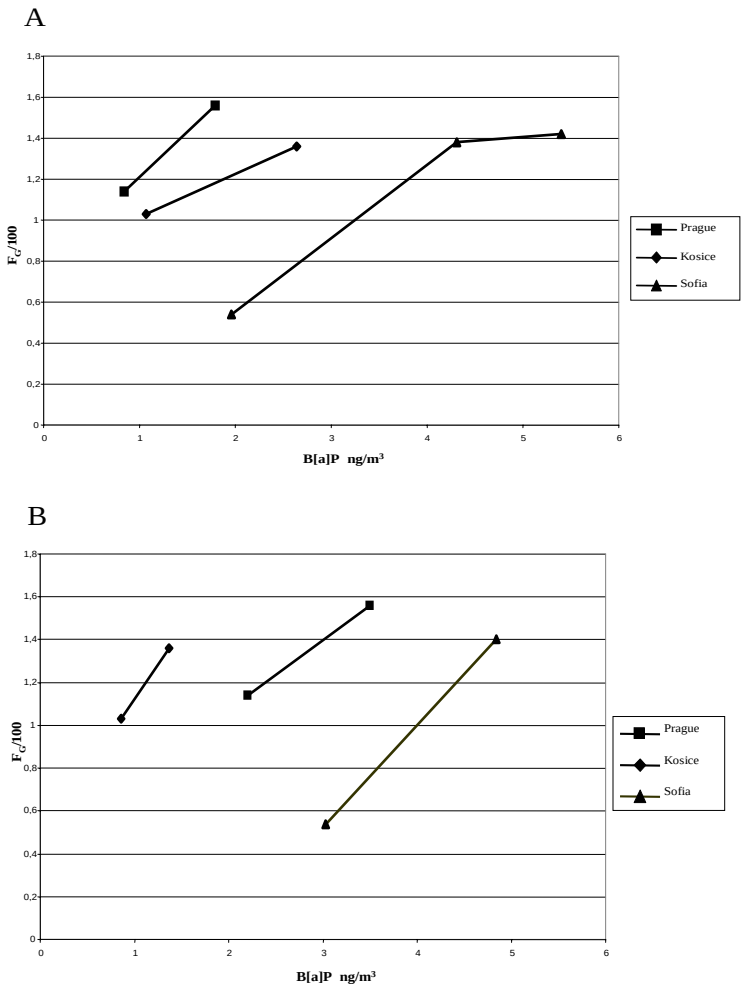


Figure 4: A – The relationship between B[a]P exposure by personal monitoring and F_G/100, exposed vs. control in Prague, Kosice and Sofia. B - The relationship between B[a]P exposure by HiVol sampling and F_G/100, exposed vs. control in Prague, Kosice and Sofia.

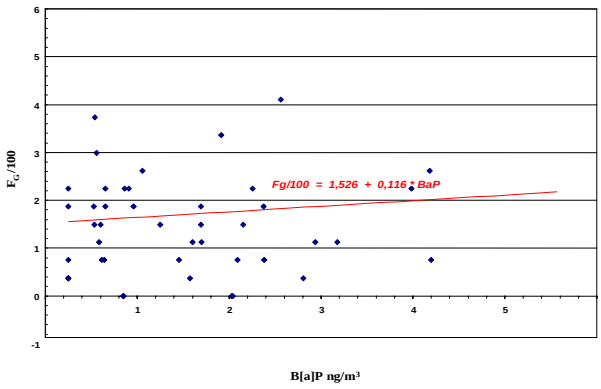


Figure 5: The relationship between B[a]P exposure by personal monitoring and F_G/100 – data for individual subjects in Prague.

DISCUSSION

The obtained results indicate a significant genotoxicity of organic compounds adsorbed onto ambient air particles, which induce stable translocations determined in peripheral lymphocytes. It corresponds to genotoxicity of extracted organic matter from air particles as was proved by Binkova et al. (2003) for the sample from Prague. We may conclude that FISH analysis indicates that police officers in Prague, Kosice and Sofia as well as bus drivers in Sofia represent a group with increased genotoxic risk.

In both the cities Prague and Sofia, traffic represents a significant source of pollution, and this may also be affected by the age of used cars as well as their technical quality (e.g. diesel emissions from trucks). Genetic damage observed in all three cities Prague, Kosice and Sofia seems to be important as a marker of possible health injury during the next decades, especially likely to affect future pregnancies, cardiovascular diseases and cancer. The differences in genetic effects observed between the occupationally exposed and control populations in this study reflect the longer exposure to polluted air experienced by the former population. It seems to be appropriate to study the air pollution in all three cities more thoroughly as other non-occupationally exposed city dwellers experience long-term environmental exposure to similarly polluted air.

The basic difference between the studied exposed groups and controls is that the police officers usually walk through busy streets in 8-12 h shifts, but controls spent usually more than 90% of their time indoors. Usually it is calculated that in a non-smoking house the concentration of $PM_{2.5}$ and c-PAHs is 50-60 % of outdoor pollution. It means that the police officers may be exposed to approximately twice the concentration of $PM_{2.5}$ and c-PAHs than the controls during their workshifts. In all three cities the level of air pollution to c-PAHs significantly increased the genomic frequency of translocations during the winter period.

This is the first study when the FISH method was used to analyze the impact of environmental air pollution. According to the original hypothesis it is expected that the most important group of chemicals responsible for the biological activity of air pollution represent c-PAHs. Using biomarkers of exposure, an increased level of DNA adducts in exposed groups compared to controls has usually been observed (Peluso et al., 1998). Using cytogenetic endpoints as biomarkers of effect, no increase of chromosomal aberrations by the conventional method, SCE or micronuclei by air pollutants was observed in Europe (Bolognesi et al., 1997a, b; Binkova et al., 1996). Zhao et al. (1998) observed an increase of SCE and micronuclei in traffic policemen compared to controls, but the PAH exposure was not determined. We may postulate that the FISH method seems to be more sensitive than other cytogenetic endpoints, and it may be concluded as well that the increase of genomic translocations represent a more significant health risk for their carriers, especially for the process of carcinogenesis.

It will be necessary to analyze different factors, which could affect genomic frequency of translocations such as for example life style, antioxidant vitamin levels or genetic polymorphism of metabolic genes. Preliminary results from Prague indicate that the frequency of translocations determined by FISH was associated with B(a)P-like DNA adducts (corresponding to the exposure of c-PAHs) (Sram et al., 2004b). This result supports the idea to use FISH whole chromosome painting as a new sensitive biomarker of effect to evaluate not only occupational exposure to carcinogens, but also environmental exposure to much lower concentrations as detected in air pollution, and to establish how to translate (use) these data for risk assessment.

Acknowledgements

The authors would like to thank Ing. Ivo Solansky for the statistical analysis.
EC (grant IC QLRT-2000-00091 project EXPAH) and the Czech Ministry of the Environment (grant VaV/740/5/03) supported this study.

REFERENCES

Albertini, R.J., Anderson, D., Douglas, G.R., Hagmar, L., Hemminki, K., Merlo, F., Natarajan, A.T., Norppa, H., Shuker, D.E., Tice, R., Waters, M.D., and Aition, A. 2000. IPCS guidelines for the monitoring of genotoxic effects of carcinogens in humans. *Mutat. Res.* 463:111-172.

Binkova, B.B., Lewtas, J., Miskova, I., Rossner, P., Cerna, M., Mrackova, G., Peterkova, K., Mumford, J., Meyer, S., and Sram, R. 1996. Biomaker studies in Northern Bohemia. *Environ. Health Perspect.* 104:591-597.

Binkova, B.B., Vesely, D., Vesela, D., Jelinek, R., and Sram, R.J. 1999. Genotoxicity and embryotoxicity of urban air particulate matter collected during winter and summer period in two different districts of the Czech Republic. *Mutat. Res.* 440:45-58.

Binkova, B., Cerna, M., Pastorkova, A., Jelinek, R., Benes, I., Novak, J., and Sram, R.J. 2003. Biological activities of organic compounds adsorbed onto ambient air particles: comparison between the cities of Teplice and Prague during the summer and winter seasons 2000-2001. *Mutat. Res.* 525:43-59.

Bolognesi, C., Gallerani, E., Bonatti, S., De Ferrari, M., Fontana, V., Valerio, F., Merlo, F., Abbondandolo, A. 1997a. Sister chromatid exchange induction in peripheral blood lymphocytes of traffic police workers. *Mutat. Res.* 394:37-44.

Bolognesi, C., Merlo, F., Rabboni, R., Valerio, F., and Abbondandolo, A. 1997b. Cytogenetic biomonitoring in traffic police workers: micronucleus test in peripheral blood lymphocytes. *Environ. Mol. Mutagen.* 30:396-402.

Bonassi, S., Hagmar, L., Stromberg, U., Huisi, A., Montagud, A.H., Tinnerberg, H., Forni, A., Heikkila, P., Wanders, S., Wilhardt, P., Hansteen, I.-L., Knudsen, L., and Norppa, H. 2000. Chromosomal aberrations in lymphocytes predict human cancer independently of exposure to carcinogens. *Cancer Res.* 60:1619-1625.

Carrano, A., and Natarajan, A.T. 1988. Considerations for population monitoring using cytogenetic techniques. *Mutat. Res.* 204:379-406.

Dejmek, J., Solansky, I., Benes, I., Lenicek, J., and Sram, R.J. 2000. The impact of polycyclic aromatic hydrocarbons and fine particles on pregnancy outcome. *Environ. Health Perspect.* 108:1159-1164.

Dockery, D.W., Pope, C.A., Xu, X., Spengler, J.D., Ware, J.H., Fay, M.E., Ferris, B.G., and Speizer, F.E. 1993. An association between air pollution and mortality in six US cities. *New Engl. J. Med.* 329:1753-1759.

Farmer, P.B., Singh, R., Kaur, B., Sram, R.J., Binkova, B., Kalina, I., Popov, T.A., Garte, S., Taioli, E., Gabelova, A., and Cebulska-Wasilewska, A. 2003. Molecular epidemiology studies of carcinogenic environmental pollutants. Effects of polycyclic aromatic hydrocarbons (PAHs) in environmental pollution on exogeneous and oxidative DNA damage. *Mutat. Res.* 544:397-402.

Hagmar, L., Bonassi, S., Stromberg, U., Mikoczy, Z., Lando, C., Hansteen, I.-L., Montagud, A.H., Knudsen, L., Norppa, H., Reuterwall, C., Tinnerberg, H., Brogger, A., Forni, A., Hogstedt, B., Lambert, B., Mitelman, F., Nordenson, I., Salomaa,, S., and Sherfving, S. 1998. Cancer predictive value of cytogenetic markers used in occupational health surveillance programs: a report from an ongoing study by the European Study Group on Cytogenetic Biomarkers and Health. *Mutat. Res.* 405:171-178.

Katsouyanni, K., and Pershagen, G. 1997. Ambient air pollution exposure and cancer. *Cancer Cause Control* 8:289-291.

Lindholm, C., Tekkel, M., Veidebaum, T., Ilus, T., and Salomaa, S. 1998. Persistence of translocations after accidental exposure to ionizing radiation. *Int. J. Radiat. Biol.* 74:565-571.

Lucas, J.N., and Sachs, R.K. 1993. Using three-color chromosome painting to test chromosome aberration models. *P. Natl. Acad. Sci. USA* 90:1484-1487.

Matsumoto, K., Ramsey, M.J., Nelson, D.O., and Tucker, J.D. 1998. Persistence of radiation-induced translocations in human peripheral blood determined by chromosome painting. *Radiat. Res.* 149:602-613.

Natarajan, A.T., Balajee, A.S., Boei, J.J., Darroudi, F., Dominguez, I., Hande, M.P., Meiers, M., Slijepcevic, P., Vermeulen, S., and Xiao, Y. 1996. Mechanisms of induction of chromosomal aberrations and their detection by fluorescence in situ hybridization. *Mutat. Res.* 372:247-258.

Peluso, M., Merlo, F., Munnia, A., Valerio, F., Perrotta, A., Puntoni, R., and Parodi, S. 1998. ^{32}P-postlabeling detection of aromatic adducts in the white blood cell DNA of nonsmoking police officers. *Cancer Epidem. Biomar.* 7:3-11.

Perrera, F.P., Jedrychowski, W., Rauh, V., and Whyatt, R.M. 1999. Molecular epidemiological research on the effects of environmental pollutants on the fetus. *Environ. Health Perspect.* 107:451-460.

Pope, C.A., Thun, M.J., Namboodiri, M.M., Dockery, D.W., Evans, J.S., Speizer, F.E., and Health, C.W. 1995. Particulate air pollution as a predictor of mortality in a prospective study of US adults. *Am. J. Respir. Crit. Care Med.* 151:669-674.

Pope, C.A., Burnett, R.T., Thun, M.J., Calle, E.E., Krewski, D.K., Ito, K., and Thurston, G.D. 2002. Lung cancer, cardiopulmonary mortality and long-term exposure to fine particulate air pollution. *J. Am. Med. Assoc.* 287:1132-1141.

Pressl, S., and Stephan, G. 1998. Chromosome translocations detected by fluorescence in situ hybridization (FISH) – a useful tool in population monitoring? *Toxicol. Lett.* 96-97:189-194.

Rubes, J., Kucharova, S., Vozdova, M., Musilova, P., and Zudova, Z. 1998. Cytogenetic analysis of peripheral lymphocytes in medical personnel by means of FISH. *Mutat. Res.* 412:293-298.

Sorsa, M., Autio, K., Demopoulos, N.A., Jarventaus, P., Rossner, P., Sram, R.J. Stephanou, G., and Vladimiropoulos, D. 1994. Human cytogenetic biomonitoring of occupational exposure to 1,3-butadiene. *Mutat. Res.* 309:321-326.

Smerhovsky, Z., Landa, K., Rossner, P., Brabec, M., Zudova, Z., Hola, N., Pokorna, Z., Mareckova, J., and Hurychova, D. 2001. Risk of cancer in an occupationally exposed cohort with increased level of chromosomal aberrations. *Environ. Health Perspect.* 109:41-45.

Sram, R.J., Binkova, B., Rossner, P., Rubes, J., Topinka, J., and Dejmek, J. 1999. Adverse reproductive outcomes from exposure to environmental mutagens. *Mutat. Res.* 428:203-215.

Sram, R.J., Rossner, P., and Smerhovsky, Z. 2004a. Cytogenetic analysis and occupational health in the Czech Republic. *Mutat. Res.* 566:21-48.

Sram, R.J., Beskid, O., Binkova, B., Rossner, P., and Smerhovsky, Z. 2004b. Cytogenetic analysis using fluorescence in siitu hybridization (FISH) to evaluate occupational exposure to carcinogens. *Toxicol. Lett.*149:335-344.

Swiger, R.R., and Tucker, J.D. 1996. Fluorescence in situ hybridization. *Environ. Mol. Mutagen.* 27:245-254.

Tucker, J.D., Morgan, W.F., Awa, A.A., Bauchinger, M., Blakey, D., Cornforth, M.N., Littlefield, L.G., Natarajan, A.T., and Shasserre, C. 1995. A proposed system for scoring structural aberrations detected by chromosome painting. *Cytogenet. Cell. Genet.* 68:211-221.

U.S. EPA 1999. EPA Report, Compendium of methods for toxic organic compounds in ambient air. Compendium method TO-13A. No. 625/R-96/010b, US EPA, OH.

Watts, R., Lewtas, J., Stevens, R., Hartlage, T., Pinto, J., Williams, R., Hattaway, K., Miskova, I., Benes, I., Kotesovec, F., and Sram, R.J. 1994. Czech-U.S. EPA health study: Assessment of personal and ambient air exposure to PAH and organic mutagens in the Teplice district of Northern Bohemia. *Int. J. Environ. Anal. Chem.* 56:271-287.

Zhao, X., Niu, J., Wang, Y., Yan, Ch., Wang, X., and Wang, J. 1998. Genotoxicity and chronic health effects of automobile exhaust: a study on the traffic policemen in the city of Lanzhou. *Mutat. Res.* 415:185-190.

PART IV - POLICY TOOLS AND APPROACHES

AIR QUALITY MODELLING FOR POLICY DEVELOPMENT

Neville Reid, P.K. Misra[1], Markus Amman[2], Jeremy Hales[3]

ABSTRACT

Atmospheric models constitute the best tools available for the setting of policy, and may, in some cases, be the only tools that are available. The best examples of their kind bring together all current knowledge of pollutant behaviour in the atmosphere, making it possible to unravel the often complex interactions between pollutants and atmospheric dynamics. They also allow the possibility of evaluating hypothetical changes in emissions and other conditions to evaluate potential abatement strategies, or to assess the impact of proposed new emission sources. This paper provides an overview of mathematical atmospheric models and their application to the development of air quality policy. The paper discusses the types of atmospheric models currently in use, categorized by spatial scale, and the requirements for credible modelling. Issues associated with model validity and accuracy are described and case studies are reviewed to illustrate atmospheric model use in policy development and the need for careful analysis in interpreting model predictions.

[1] Environmental Monitoring & Reporting Branch, Ontario Ministry of the Environment, Etobicoke, ON M9P 3V6 Canada.
[2] International Institute for Applied Systems Analysis, Laxenburg, Austria.
[3] Enviar, Pasco, WA 99301 USA

INTRODUCTION

A model may be defined as a representation of reality. The particular representation used in any given case can take a number of forms. Examples include scale models constructed of cardboard or wood to show the appearance of a building or a ship, or the conceptual models of human interaction that we all carry around and invoke many times per day, albeit often unconsciously. However, in the context of this paper the models considered are mathematical representations of the behaviour of pollutants in the atmosphere. The objective of this paper is to provide a description of mathematical atmospheric models, and their application to policy.

Policy is another term which has a number of possible definitions. The particular context required here includes measures designed to control or eliminate atmospheric pollutants to protect human health. In general, "control or eliminate" means "reduce or eliminate the emission of these pollutants to the atmosphere". Other policy issues are, of course, also of current importance, e.g., visibility degradation especially in pristine areas, and impacts on ecosystem health.

The value of models in policy development lies in their ability to provide a quantitative link between pollutant emissions at one or many locations, and the resulting concentrations or doses of pollutants experienced by the human or other receptors, whose health is to be protected. This allows changes in impact resulting from hypothetical changes in emissions to be readily evaluated. Such evaluation is almost impossible to carry out experimentally, either because it is far too costly to shut down a source or sources, and to attempt to measure the concentration changes at the desired receptor locations, or because the assessment is required for a source that is yet to be built. In addition, the complexity of the interrelated chemical and physical processes is such that measurements alone are extremely difficult to interpret without the conceptual framework provided by a model.

To a large extent, the acceptance of models as valuable tools has followed the realisation that an airshed can cover several jurisdictions; emissions from sources in one jurisdiction can and do have impacts on the residents of other jurisdictions. Policy negotiations between jurisdictions are very difficult without a quantitative understanding of the link between the emissions and the impacts. Models used to support such negotiations must be credible. This has driven continued improvements in the science of models, and in their evaluation against measurements.

This paper will summarise the types of atmospheric models currently in use, and the resources required to run them. The very important subject of model accuracy will be discussed and case studies will be presented. In preparing this paper a wide range of current scientific literature has been reviewed. However, this paper is not intended as a comprehensive review of the current state of modelling. Such reviews are available elsewhere, e.g., the two assessment reports produced by NARSTO (2000, 2003) and the paper by Peters et al. (1995).

POLICY APPLICATIONS OF MODELS

A distinction can be made between the use of models in the development of policies, and their use as research platforms. As a research platform a model serves as a framework for current knowledge of the chemical and physical behaviour of the atmosphere. Comparisons of model predictions with measurements then constitute tests of the current knowledge and the way in which it is implemented in the models. In this kind of evaluation model shortcomings can be of greater interest and importance than successes, since they indicate shortcomings in the underlying scientific knowledge and spur advances.

There are at least two major ways in which models can be applied in policy development:

- An appropriate model is applied to the emissions from a source, or group of sources, to predict concentrations at selected receptor locations. These predicted concentrations are then used with a risk assessment/health impact model to predict the health outcomes associated with the emissions.
- A determination is made of what the maximum allowable concentration of a given pollutant should be to be protective of human or ecosystem health. An appropriate atmospheric model is then used to assess what the emissions must be to ensure that concentrations remain at, or below, the desired concentration under all conditions.

Note that application of modelling to one problem pollutant will often provide results which are applicable to others. For example, modelling for fine particulate matter may also generate useful results for ozone, and probably also for visibility.

TYPES OF MODELS

The mathematical models discussed in this paper use data (emissions, meteorology, topography, land use, etc.) in the prediction of pollutant concentrations. Other types of mathematical models exist, particularly receptor models, which use ambient measurements and knowledge of the relative composition of emissions from the relevant sources or source categories to calculate the relative contributions of the sources at the measurement point. Receptor models are valuable in determining such contributions if the required data are available, but can not be used with confidence to predict the effect of reducing emissions from any of the sources, because they inherently assume that there is a linear relationship between emissions and concentrations. They will not be included in the discussion presented here. Further details may be found in NARSTO (2003) and the references cited therein.

Note that the terminology of modelling varies, and may depend on the particular application. The models discussed here may also be described as “chemical-transport models”, “diagnostic” (applied to the current situation) or “prognostic” (applied to a future situation).

The predictive models discussed in this paper may be categorised in many ways. The approach adopted here is to categorise by applicable spatial scale, with further consideration of mathematical formulation, and type of pollutant considered. Examples of current use models will be given in each category.

It is convenient to consider three spatial scales: local, meso- to regional, and global.

Local scale

Local scale modelling is typically used to assess the impact of single sources, or small groups of sources, over distances ranging up to tens of kilometres. Typically the pollutants are emitted from a stack, with an initial velocity, and at a temperature which is generally above that of the ambient air. Under the combined effect of the exit velocity and the buoyancy due to its elevated temperature, the plume rises above the stack top, before being bent over, and transported or advected by the wind. Turbulent eddies in the atmosphere spread the plume out in the horizontal and vertical directions. This dispersion leads to dilution of the pollutant as the plume travels downwind. The process is typically modelled by assuming a Gaussian concentration distribution in the horizontal and vertical directions (Figure 1). The resulting concentration is then given by:

$$C(x,y,z) = \frac{Q}{2\pi V \sigma_z \sigma_y} \exp\left[\frac{-y^2}{\sigma_y^{\,2}}\right] \exp\left[\frac{-z^2}{\sigma_z^{\,2}}\right] \qquad (1)$$

where
- C = concentration at point x, y, z
- x = distance along plume centre line
- y = horizontal distance from plume centre line
- z = vertical distance from plume centre line
- Q = emission rate
- V = wind speed
- σ_y, σ_z = dispersion coefficients in the y and z directions

Note that σ_y and σ_z specify how much the plume spreads, and are derived from theoretical arguments or empirical fits to observed data. They are increasing functions of x, and their numerical values are dependent on the stability of the atmosphere. Because the distance and time scales considered are short, chemical or physical transformation of the pollutants modelled is almost never included in these models.

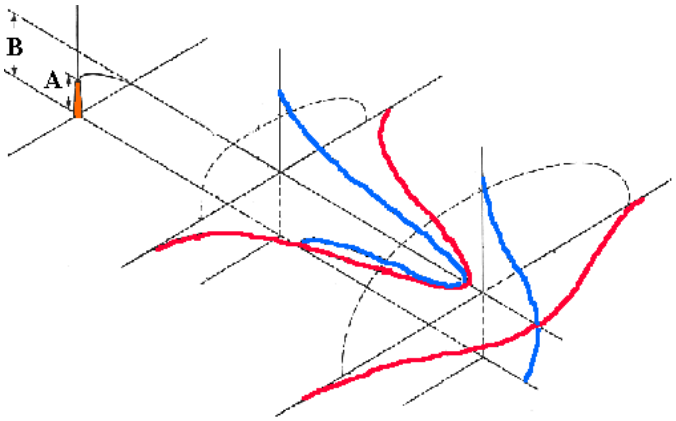


Figure 1: Depiction of Gaussian plume.

Equation 1 represents the simplest case of Gaussian dispersion. Many variations are employed to treat more complex situations, including:

- multiple sources
- impaction of the plume on the ground, or its confinement by topography
- special behaviour under certain stability or boundary layer conditions, e.g., fanning, looping, fumigation, etc.
- the effect of buildings on plume behaviour

Examples of Gaussian type models in current use include ISC and AERMOD in North America, and AUSPLUME used in Australia. The major application for this type of model is in certifying or licensing the emissions from industrial facilities. The appropriate form of the model is run for the facility seeking certification (noting that it may well still be in the design stages), to ensure that the resulting pollutant concentrations are below the relevant air quality standards or criteria, and are thus protective of human and ecosystem health.

A special case of the short term models is in their application in emergency situations, where they are applied in the case of a pollutant leak or discharge of any type. The objective in this case will normally be to define areas in which concentrations are at dangerous levels, necessitating evacuation or other emergency measures. The major difficulty in emergency response modelling usually lies in adequately defining the source emission characteristics. The effective release point may be at ground level, if there is a steady leak from a tank, or if the release material is denser than air. In the case of a fire, on the other hand, buoyancy may lift the pollutants a considerable distance vertically before transport and dispersion begin. It is also usually difficult to determine the rate of pollutant release, and in the case of a fire, even to know what pollutant is involved.

Mesoscale to Regional Scale

Mesoscale to regional scale models consider spatial scales ranging from a few hundred to a few thousand kilometres. These are the spatial scales over which many of the most pressing air pollution concerns are important, and are also the scales which often cross jurisdictional boundaries. Taken together, these facts mean that models on these scales are generally the most important for policy makers. Mesoscale is itself subdivided as meso-gamma (0 to 20 km), meso-beta (0 to 200 km) and meso-alpha (0 to 2000 km). Meso-alpha overlaps with what is usually considered regional scale (up to three or four thousand kilometres). Up to ten or fifteen years ago it was usual to consider mesoscale separately from regional scale, a separation imposed by limitations in the science of modelling and in computer capability. Advances in both make it now possible and convenient to consider both scales together. References to regional scale will be taken to include mesoscale in the remainder of this paper.

There are two major types of regional scale models, depending on the mathematical framework used.

Lagrangian models consider air parcels which travel with the wind (i.e., they are advected). Lagrangian models are

often also referred to as trajectory models, since the air parcel under consideration follows a trajectory defined by the winds, as illustrated in Figure 2. Typical trajectory times for regional scale applications are 3 to 5 days, though longer or shorter times are also used. The trajectories are normally calculated as linear segments, each one covering 3 to 6 hours, with the segment length and direction determined by the average wind speed and direction over the appropriate time step.

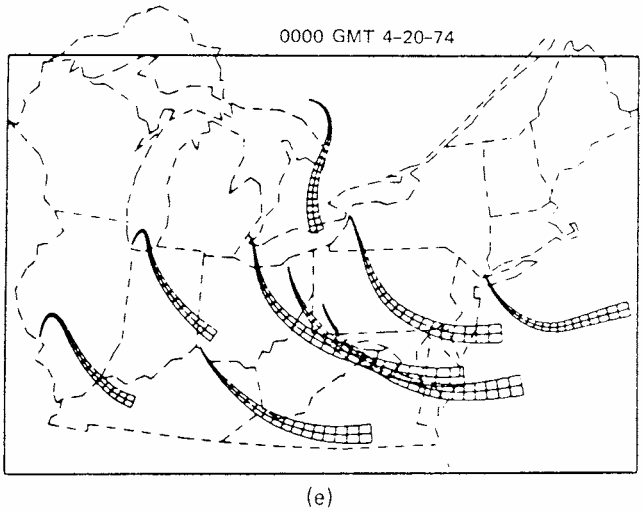


Figure 2: Depiction of a Lagrangian framework for several emission sources in North America.

There are a number of possible variants within the general category of Lagrangian models:

- The trajectory may be run forward from a given source, to evaluate its impact on receptor areas. A new trajectory will be initiated at intervals, e.g., every 3 to 6 hours. Atmospheric variability is such that these trajectories will all be different, so that all affected receptors will be impacted if a long enough time period is followed.
- The trajectory may be run backwards. This actually means that the trajectory is defined in backward steps from a given receptor, and the air parcel is then released from the start point of the trajectory, finishing at the receptor. This approach allows assessment of the effect of all possible emission sources on the given receptor.
- A number of variants deal with the treatment of the air parcel under consideration. For example, it may be considered as a “puff”, which expands during travel, to simulate atmospheric dispersion, or it may be treated as a “wall of cells”, with pollutant mass transferring from inner cells towards outer cells, again simulating atmospheric dispersion. These two methods (and others related to them) allow emissions to enter the air parcel, and chemical and physical transformations to take place within the parcel. A third method treats the emission of “particles”, with a number of releases occurring for each trajectory, but with a random displacement added normal to the trajectory direction. This displacement simulates the effect of dispersion, and the ensemble of particle positions along the trajectory gives the concentration of pollutant at points on or near the trajectory. This last method does not readily allow incorporation of emissions or chemical and physical processes.

Eulerian Models consider a mathematical framework anchored to the surface of the earth, as shown in Figure 3.

Eulerian models are often also referred to as grid models, since the framework is a three dimensional grid, with pollutants being emitted into the grid at the appropriate points. Pollutants travel through the grid, under the influence of the local winds, undergoing chemical and physical transformations as they go. Although a wide range of degrees of complexity is possible in the treatment of individual processes in an Eulerian model, models of the current generation tend to be quite similar overall.

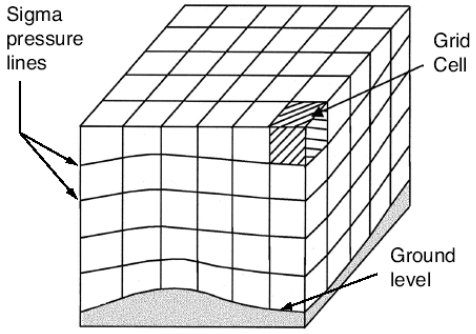


Figure 3: Eulerian modelling framework.

Both types of regional scale models have advantages and disadvantages, as summarised in Table 1.

Table 1. Advantages and disadvantages of Lagrangian and Eulerian models.

	Lagrangian	**Eulerian**
Advantages	• Computationally relatively simple • Especially suitable for small number of sources or receptors • Easy to determine transboundary fluxes	• Able to include treatment of all processes currently considered important. • Non linear phenomena, especially those associated with chemical processes, can be more easily incorporated in a direct fashion
Disadvantages	• Can not readily treat a number of processes, e.g., vertical structure of the atmosphere, including changes in wind speed and direction with altitude, and pollutant transport in layers aloft • If many sources or receptors are considered, computational effort approaches that required for Eulerian case	• Computationally more demanding

Examples of current use Lagrangian models include Calpuff and Hysplit, while Eulerian models would include Models-3/CMAQ, UAM and TAPM.

Global Models

As the name implies, global models consider the transport of pollutants throughout the atmosphere, with no artificial restriction of the domain. Many or most current use global models are Eulerian in formulation; GRANTOUR (Liousse et al., 1996) is a Lagrangian exception. The large spatial extent of these models dictates that the spatial resolution (grid spacing) must be relatively coarse to keep the computational demands within reasonable bounds.

To date, most global modelling has been confined to carbon dioxide and the climate change issue, which also means that chemical transformation is not treated, further streamlining the computation. However, expansion to other pollutants has recently taken place. An example is Environment Canada's global scale mercury modelling (Dastoor and Larocque, 2004).

Processes Treated in Models

Modern atmospheric models, particularly those on the regional scale, treat a number of processes. These are summarised briefly below:

- Emission. Emissions of the pollutants treated by the model must be presented at a resolution (temporal, spatial, chemical, etc.) appropriate to the model. Emissions from both anthropogenic and biogenic sources will be required. Much of the emission information will itself be generated by models, especially for mobile and biogenic sources.
- Transport. Transport involves the movement of pollutants by air motions in the atmosphere. Wind speeds at higher altitudes are generally higher than those at the surface, and the directions are usually different too. Pollutants which have been transferred to higher layers can thus travel considerable distances, in directions which may not be predictable from observations made at the surface of the earth. Vertical movement of pollutants, e.g., by convective processes, can thus be very important. Wind fields may be derived directly from observations, but it is now much more common to obtain them from a meteorological model, which may also provide information on clouds and precipitation, as required for other modelled processes.
- Chemistry. The underlying chemistry of the atmosphere is that of reactive species in the gas phase, driven by sunlight. However, reactions in the aqueous phase (e.g., in cloud droplets) and on the surface of particles must also be considered.
- Physical transformation. The products of some gas phase reactions are condensed solid and liquid phases, taking the form of very small particles. The processes by which these particles are formed, and by which they grow must be included. Particle size is an important parameter, from the point of view of human health impacts, as well as their dynamics in the atmosphere. For example, particles can play a role in the formation of clouds.
- Deposition. Atmospheric pollutants can be deposited to the surface of the earth by dry and wet processes. Dry deposition occurs when the pollutant impacts and is retained on the surface. Wet deposition is the process by which pollutants are incorporated into water droplets or ice crystals and subsequently removed in precipitation.

It may be seen from the above description of processes that modern atmospheric models rely on information provided to them by other models. It is therefore more correct to speak of a system of models, rather than a model, though both terms will be used here.

REQUIREMENTS TO RUN MODELS

A number of requirements must be met before a model or models can be used by an institution or agency. Ironically, acquiring the model itself may be the least difficult of these. This follows from the ready availability of codes for a number of models, e.g., through the World-wide Web. The US EPA in particular has a free distribution policy for many models including Models-3/CMAQ, HYSPLIT, AERMOD, etc.

A number of the important requirements for credible modelling are summarised below.

Choice of model

Even though a model may be freely available, it does not follow that it is necessarily the most appropriate for the situation under investigation. In particular, the science of the model must match the pollutant(s) of concern. For example, if the pollutant of concern is fine particulate matter, the model chemistry must be able to handle reactions of NO_x, SO_2, volatile organic compounds (VOC), ammonia, etc. Reactions in both the gas and aqueous phases must be included, and preferably also heterogeneous reactions taking place on the surfaces of particles. Apart from correct treatment of transport and diffusion, the formation and growth of particles must be included, and the model must be able to track the evolution of particle mass as a function of size. The ability to treat deposition of pollutants to the surface of the earth by both wet and dry processes is also required.

Input data

Modern models require a considerable volume of data. The specific needs reflect the science incorporated in the model, but will typically include the following:

- Emissions. For all sources treated by the model (for each grid square of an Eulerian model) the rate of emission is required for each of the chemical species followed by the model, specifically including each of the VOC species or categories used in the model chemistry, i.e., both anthropogenic and biogenic. These emissions

should relate to the specific time period being studied. A very large effort is devoted to emission inventory construction world wide, but it remains true that there is still considerable room for improved accuracy. In general, emissions of sulphur dioxide are specified with the best accuracy, followed by oxides of nitrogen, then volatile organic compounds. With the need to model fine particulate matter formation and transport, emissions data are also required for primary particle emissions as a function of particle size, as well as emissions of ammonia. Neither of these is particularly well categorised as yet. Specifically, primary particles in small size ranges (e.g., $PM_{2.5}$ or smaller) may be emitted in large amounts from unpaved roads or agricultural operations, but it appears that only a fraction of these emissions are "effective," i.e., are subject to transport. This fraction is not well known. Ammonia emissions are also subject to large uncertainty and it is possible that not all sources are known.

- Geophysical data. Information is required on a range of surface parameters, including topography, land use category and vegetation type. These are generally more readily available than emissions data, and some of them (like topography) do not usually change with time.
- Meteorology. Meteorological information is typically required to drive the transport in the model. This information is needed at several levels in the atmosphere, and must also be for the period to be modelled.
- Initial and boundary conditions. It is usual to specify the initial chemical concentrations for a number of the major species in the model. These will be taken from typical or average values measured, or previously modelled, for the region of interest. The exact values chosen may not be completely critical, as the model will usually be run for some initial, warm up period (typically about two days), so that concentrations in the modelling domain are determined by model processes within the domain. However, poor choice of initial conditions will lead to long warm up times. It is also necessary to specify concentrations at the boundaries of the model, except for global models. It is usual to allow a buffer of two to three grid squares on the edges of the domain for an Eulerian model, within which the concentrations will be determined more by the boundary conditions than by the processes within the model. Again, the exact values chosen for the boundary conditions may not be important, except that poor choices will require large buffer zones, and boundary conditions will be very important for inert substances which are not emitted within the model domain. Note that while it is relatively simple to estimate initial and boundary conditions at the surface based on measurements, these values are also required at higher levels in the atmosphere, where measurements are very much sparser. Current practice, which addresses the specification of initial and boundary conditions is to nest the model. In nesting an initial run is carried out for a large domain at relatively coarse resolution. This is followed by a run at finer resolution for a smaller subdomain, using predictions from the first run to provide initial and boundary conditions.

Computer resources

Modern Eulerian models are very computer intensive. However, modern computers are much more powerful than were their predecessors. The result is that computational limitations, although they still exist, are much less restrictive than they were a decade ago. Many modern models are run on large computer facilities, but others are now run on computer workstations, or clusters of desktop computers, which are within the budget of many institutions.

Practitioners

The need for skilled practitioners to run models should not be underestimated. Many models are now freely available, as noted above. However, the danger exists that these models can produce misleading results, unless their use is overseen by knowledgeable persons. Training of new generations of modellers is therefore seen as a very important component of modelling.

To summarise, many models are now available at little or no cost. However, the establishment of an atmospheric modelling capability carries a non-trivial cost, because of the need to provide appropriate supporting data, computer resources and scientists. Examples of model applications are discussed later in this paper, including two cases illustrating how misinterpretation can occur.

MODEL VALIDITY AND ACCURACY

A fundamental requirement for the use of models in policy is that their predictions be credible. This means that the model must not only get the right answer, but that it must get the right answer for the right reasons. The second point is important, because a model which predicts the correct current concentration of a pollutant because of a cancellation of errors can not be relied upon to provide correct predictions for altered conditions, for example for scenarios of reduced emissions as would typically be required for policy evaluation.

Validation

Full model evaluation and validation should include:

- thorough peer review of the science of the model
- evaluation of the model's ability to predict concentrations of the pollutant of interest, by comparing predictions against measurements, preferably over a wide range of meteorological conditions (this operational evaluation tests the model's ability to get the right answer)
- comparison of the performance of two or more models
- more detailed evaluation of the ability of the model to predict correctly the concentrations of other chemical species involved in the chemical scheme, for example an ozone prediction model would be tested also for its ability to predict the concentrations of NO_x, selected volatile organic compounds and other product species such as PAN (peroxyacetyl nitrate). This procedure, commonly referred to as diagnostic evaluation, tests the ability of the model to get the right result for the right reasons. Diagnostic evaluation also includes, where possible, tests of the individual, component modules of the model.

Model predictions and measurements may be compared at several levels of rigour. The most demanding test occurs when predictions and measurements are paired in time and space. In other words the model is evaluated on its ability to simulate exactly what occurs at a given point at a given time. Less rigorous tests would allow relaxation of the pairing in either time or space.

Relaxation of the time pairing allows a model to score well if it, for example, predicts a similar pattern of ozone concentration through the day, but one hour later than was actually observed. Relaxation of the spatial pairing would allow the model to score well if its predictions for a given point matched well at all times with the measurements made at a different point. Good performance in these relaxed tests might indicate that the chemistry/transport component of the model is performing well, but that there is a deficiency in the meteorological simulation. Thus if the model's wind speeds are too low, high concentrations would reach a given point too late, whereas if the wind direction is off, the prediction will be given for the incorrect place.

Other, more relaxed, tests are also often performed, such as not requiring pairing in either space or time; comparing model predictions averaged over 4 (or more) adjacent grid cells, with point measurements; or comparing various averages. These could be, for example modelled and measured averages over selected areas, or over selected times. Good performance in such tests might suggest that the model will be useful for broad scale, long term applications, but not for the simulation of episodes.

It is important to note that there is a fundamental limitation to how well models can be expected to replicate measurements. This is a consequence of incommensurability, which arises because a model predicts concentrations averaged over a certain volume, whereas measurements are made at a point. For an Eulerian model the volume is the size of the grid cell, which will generally be a minimum of four kilometres on a side, by fifty metres deep, and may well be substantially larger. It is clear that if the concentration of the pollutant in question varies in space, as is almost always the case, the model prediction is likely to diverge from the measurement.

Another concern is that most model validation to date has focused on summer conditions. However, with the increased attention now given to fine particulate matter, which can attain high concentrations at other times of year, more effort will have to be devoted to validation for seasons other than summer.

Model Accuracy

Two main questions must be faced in assessing or discussing model accuracy:

- How accurate is the model?
- How accurate does the model need to be?

These apparently simple questions are, as yet, only partially answerable. In fact, it may never be possible to answer them completely. Further discussion is in order as is considerable further investigation.

How accurate is the model?

For policy purposes it would be desirable to be able to state that the model prediction is uncertain to ± X%. Such a definitive statement can not be made, because model uncertainty depends on many factors, some of them specific to the particular application. Thus, model uncertainty includes contributions from uncertainties in the input data (meteorology, emissions, etc.) and in the model itself. Model uncertainties include uncertainty in parameters like chemical rates, uncertainties in the science on which the model is based, and uncertainties in implementation of the science into numerical form). In addition, the process of model evaluation itself is somewhat uncertain, because of measurement uncertainties, and also because of the problem of incommensurability.

Examples of the results obtained in model evaluation are presented in Table 2. The statistical measures defined in the table are typical of what is used in evaluation, but other measures may also be used.

Table 2. Performance evaluations of models for $PM_{2.5}$ and components with the SCAQA data base in the Los Angeles Basin. Two episodes were used: 24 - 25 June 1987 and 27 - 28 August 1987. (NARSTO, 2003).

MODEL		**UAM-AERO**	**GATOR**	**CIT**	**UAM-AIM**	**SAQM-AERO**
Period		**June 25**	**Aug 27-28**	**Aug 28**	**June 24-25**	**Aug 28**
Statistics		**Normalised statistics[a,b] (%)**	**Normalised statistics[a,c] (%)**	**Normalised statistics[a,d] (%)**	**Normalised statistics[e,f] (%)**	**Normalised statistics[e,g] (%)**
$PM_{2.5}$ mass	error	32	44	46	NA[h]	NA
	bias	+24	-3	+46	NA	10
Sulphate	error	48	28	34	NA	NA
	bias	-10	+4	-30	-21	-33
Nitrate	error	18	68	61	NA	NA
	bias	+11	-21	+47	+52	-14
EC[i]	error	15	57	50	NA	NA
	bias	-10	+30	+35	NA	NA
OC[j]	error	38	49	40	NA	NA
	bias	-38	-44	+14	NA	+38

[a]Normalised error = $\frac{1}{N}\sum_{i=1}^{N}\left|\frac{P_i - O_i}{O_i}\right|$; normalised bias = $\frac{1}{N}\sum_{i=1}^{N}\left(\frac{P_i - O_i}{O_i}\right)$; where P_i=prediction, O_i=observation, N=number of samples

[b]Mean over all sites and sampling periods of the normalised errors of sampling-period averaged concentrations
[c]Mean over all sites and hours of the normalised errors of 1-hour averaged concentrations (note that sampling periods exceeded 1 hour)
[d]Mean over all sites of the normalised errors of the 24-hour average concentrations
[e]Normalised bias of means = $\frac{\left(\sum P_i - O_i\right)}{\sum O_i}$

[f]Normalised bias of the means over all sampling periods and sites of the sampling-period average concentrations
[g]Normalised bias of the means over all sites of the 24-hour average concentrations
[h]Not available
[i]Elemental carbon
[j]Organic carbon

The range of statistical measures used, and of the values reported in Table 2 indicate that further development in this area is warranted. Table 3 (from NARSTO, 2003) presents a different view of model reliability, containing a

semiquantitative judgement of the certainty which can be associated with the components of and inputs to a modelling system.

Table 3. Levels of confidence in aspects of model simulations (NARSTO, 2003).

Model Aspect	Confidence Level[a]	Model Aspect	Confidence Level[a]
PM Mass Components		**Gases**	
PM ultrafine	VL	SO_2	H
PM fine	M	NO_x	H
PM coarse	M	NH_3	M
PM Composition		VOC	M
Sulphate	M – H	HNO_3	M
Nitrate	M	O_3	M
Ammonium	M	**Spatial Scale**	
OC[b] primary	L	Continental	L
OC secondary	VL	Regional	M
BC[c]	L	Urban	L – M
Crustal material	L	**Temporal Scale**	
Water	L	Annual	L
Metals, biologicals, peroxides	VL	Seasonal	L
		Episodic	M

[a] H: High, M: Medium, L: Low, VL: Very Low
[b] OC: Organic Carbon
[c] BC: Black Carbon

Even if a statement in the desired form could be made about the model, further uncertainty arises because policy applications require the prediction of some future or unknown state. This future state will involve emission changes, as new facilities are built, or as emissions of existing facilities are controlled, and will also correspond to unknown meteorological conditions, and possibly also to changed surface conditions (e.g., changes in land use).

How accurate does the model need to be?

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. At least two situations are possible.

- If an atmospheric model is used to provide concentrations or doses that are then incorporated into dose response models, followed by cost/benefit analyses, it is important to bear in mind the uncertainties involved in each step of the calculation. In particular, since errors usually add in quadrature, a model error which is, say, one third of those of the other two steps will not contribute significantly to the overall error. In this type of approach model predictions that are accurate within a factor of two or three may be perfectly acceptable.

- In the contrasting case, "acceptable" pollutant concentrations (or standards) are set by following a weight of evidence approach. Atmospheric models are then used to determine the extent of control required by emission sources contributing to the pollutant burden. The control decisions usually carry significant cost implications, so that model validity is subjected to intense scrutiny. The diagnostic evaluation of the model is important in this application, because, as discussed above, the model prediction is obtained for conditions outside the range included in the validation. The ability of the model to predict correctly the relative change in concentration for a given change in emissions may actually be more important than its ability to predict the absolute concentration. To illustrate, consider the following simplified example:

The ambient concentration of a specific pollutant is 25 μg m^{-3}, whereas the standard for this pollutant is 20 μg m^{-3}. The required reduction is therefore 20%. When the model is run for current conditions, the predicted concentration is 30 μg m^{-3}. To determine the appropriate emission reductions, the model is run to find what level of emissions will give a predicted concentration of 24 μg m^{-3} (i.e., 30 μg m^{-3} reduced by 20%, thus assuming that the model correctly calculates the per cent reduction for a given emission change). This approach is rigorously accurate if the

atmospheric concentrations are directly proportional to emissions, and there is no background contribution from sources which can not be controlled.

Further work on defining model accuracy is clearly needed. It is relevant to note recent work in Germany under the TFS Model evaluation (Tilmes et al., 1999) which looked at policy questions associated with different regulatory options and attempted to specify model performance features necessary to decide cleanly between the regulatory alternatives. The results of this procedure were termed Model Quality Objectives (MQO), as a parallel to Data Quality Objectives, which are normally specified in advance of measurement campaigns. The MQO developed in the initial work were quite stringent, and none of the models evaluated was able to meet them.

EXAMPLES OF ATMOSPHERIC MODEL USE IN POLICY DEVELOPMENT

Presented below are examples of atmospheric model applications which were involved in, or could be of relevance to policy discussions. Also presented are cautionary examples, which illustrates the need for careful analysis in the application of a model.

Assessment of ozone abatement strategies for the Greater Madrid area

Ozone concentrations in the Greater Madrid area can exceed 100 ppb (200 μg m^{-3}). Public notification is required if the concentration exceeds 90 ppb (180 μg m^{-3}), which is the EU guideline. A modelling analysis was carried out to determine the effectiveness of emission reduction scenarios in reducing peak concentrations (Palacios et al., 2002).

The meteorological model used in this work was the Topographic Vorticity Model (TVM), which provided input for the chemical-transport model, which was based on the CIT model developed at California Institute of Technology. The meteorological model covered an area of 300 by 300 kilometres, with a maximum horizontal resolution of 5 x 5 km. Twenty four vertical layers were considered extending to over 15 km. The chemistry and transport were calculated for an inner domain covering 270 x 200 km, also at 5 km resolution, and having 8 levels, up to 4400 metres.

Two episodes were considered in the analysis. They occurred on 14 July 1992 and 15 July 1995. Wind directions were different for these two episodes, which were considered representative of the conditions which most often result in high ozone conditions in the region. The statement was made that high ozone concentrations in the Greater Madrid area are associated with local emissions, as the area is removed from significant regional sources.

Emission scenarios considered are summarised in Table 4.

Table 4. Emission scenarios for Greater Madrid modelling.

Scenario	Description
1	100% increase in road traffic emissions
2	25% decrease in road traffic emissions
3	50% decrease in road traffic emissions
4	70% decrease in road traffic emissions
5	100% decrease in road traffic emissions
6	25% decrease in total anthropogenic emissions
7	50% decrease in total anthropogenic emissions
8	70% decrease in total anthropogenic emissions
9	25% decrease in total anthropogenic VOC emissions
10	25% decrease in total anthropogenic NO_x emissions
11	No anthropogenic emissions
12	Emissions from 0000 to 0800
13	Emissions from 0900 to 2300

Response to these scenarios was analysed in several ways:

1. *Effect on maximum ozone concentration domain-wide:* Reduction in total anthropogenic emissions was more effective than reduction in vehicular emissions alone (remembering that total anthropogenic emissions include motor vehicles). Indeed, total elimination of vehicular emissions, without reducing other anthropogenic emissions, would not reduce peak ozone concentrations for 14 June 1992 below the 90 ppb guideline value, and close to 100% elimination of vehicle emissions would be required for 15 July 1995.
2. *Effect on total amount of ozone above 60 ppb:* This quantity was used to measure total production of ozone in the domain, since 60 ppb appeared to be a background concentration, produced even when anthropogenic emissions within the domain were totally eliminated. Reduction of total anthropogenic emissions by 70% and total elimination of vehicular emissions produced similar reductions in ozone production, i.e., 32 to 51%. This is consistent with what might be expected from the emission inventory, which indicates that 82% of total NO_x emissions and 59% of total VOC emissions are associated with motor vehicles.
3. *Effect of emission reductions for certain time periods:* Scenarios 12 and 13 address the effect on ozone production of eliminating early morning (prior to 9 a.m.) and daytime (9 a.m. to midnight) emissions. It was found that both make a contribution to ozone formation.
4. *Effect of NO_x versus VOC control:* The effect of separate 25% decreases in NO_x and VOC emissions were investigated. At certain locations and times, reduced NO_x emissions resulted in higher ozone concentrations, which indicates that NO_x scavenging is currently limiting the formation of ozone at these locations. Overall, however, the NO_x reduction led to reduced production of ozone for both episodes. On the other hand, reducing VOC emissions produced reduced ozone for only the 15 July 1995 episode, but had essentially no effect during the 14 July 1992 episode.

This study illustrates the approach which is usually adopted in developing policies for pollutant abatement, i.e., develop a set of emission reduction scenarios, and evaluate their effectiveness using a model or models. In practice, the scenarios will have to take into account practical factors, the most important of which will usually be the cost and availability of emission control equipment. In addition, the study brings out some interesting points which also bear on the use of models in policy development:

- What is the most appropriate measure for the pollutant in question? It is now well established that many or all pollutants of current concern have no threshold concentration below which they may be considered harmless. This would suggest that the most appropriate measure should be total exposure for the affected population. However, standards are usually expressed as short term averages (e.g., one hour) not to be exceeded at any specific location. This tends to focus attention on reducing emissions so as to avoid exceedence of the standard, which may not minimise population exposure.
- What time period should be studied? Modern atmospheric models are costly in terms of set up, actual running, and analysing the results. It is therefore very common, as was done in this study, to investigate one or a very few episodes. As was illustrated here, two similar episodes, as judged on the basis of the underlying dispersion characteristics, gave somewhat different results. Appropriate control is required for all conditions, but conclusions based on a limited number of episodes can not be generalised with any degree of confidence. This will be especially true if overall human exposure is of concern, rather than control of peak pollutant concentration.
- The authors of the study actually modelled each scenario twice, using two different chemical mechanisms. Somewhat different results were obtained under certain circumstances, underlining the importance of model validation.

The impact of urban development on air quality and energy use.

Melbourne, Australia was used in a case study, investigating the effects of alternative urban forms on air quality (Manins, et al., 1998). The population of Melbourne was 3,168,300 in 1991, which was taken as the base year for the study. This population is expected to increase by 500,000 by 2011. The purpose of the study was to set up a number of scenarios for the growth of the city, and evaluate their impact on ambient air quality, greenhouse gas emissions and transport energy use.

The growth scenarios were (Figure 4):

- *Business as Usual* – extrapolation of current patterns into the future, with the additional population assigned to zones within the city in proportion to the 1991 base population of each zone.
- *Compact City* – increased population and density of an inner group of eight suburbs. The population density of these suburbs becomes 300 persons per hectare, well above current densities.
- *Edge City* – increased population, housing densities and employment at selected nodes within the middle ring of the city; increased investment in orbital freeways linking these nodes.
- *Corridor City* – focus of growth along three linear corridors emanating from the Central Business District, supported by upgraded public transit infrastructure along the corridors.
- *Fringe City* – additional growth predominantly on the fringe of the city, with 30% of the new population added to currently open areas on the urban fringe, 60% within the outer ring of the city, and 10% to the middle ring. New manufacturing and service industries were also added to the same zones, in the same proportions.
- *Ultra City* – additional growth primarily in provincial cities within 100 km of the principal city, and linked by high speed rail transport. Seventy percent of the new population is added to the four provincial centres, with the remaining 30% added proportionately to the current city as in the Business as Usual scenario.

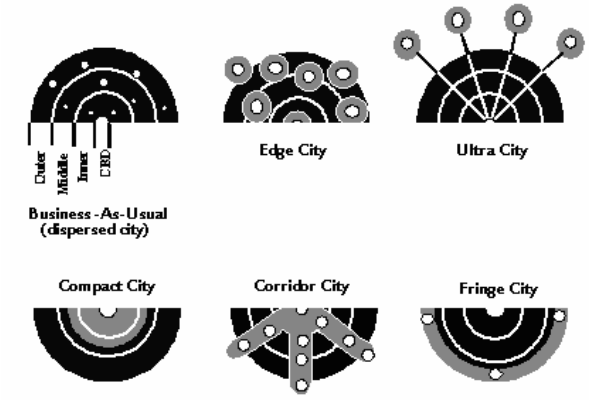


Figure 4: City configurations.

The assessment involved initial application of a land use-transport model (LUTE) which provided emissions for a gridded atmospheric model used to calculate the pollutant concentrations for each scenario.

For the LUTE analysis the region was divided into 26 zones. In each zone emissions were calculated, and gridded, based on the land use of the zone. A transportation gravity model was used to calculate trips between pairs of zones. The trips were assigned to the links between zones, and the emissions calculated, incorporating the effect of congestion on traffic emissions.

The atmospheric model comprised a gridded meteorological model (LADM) driving a chemistry and transport model based on the CalTech model. Computed calculations were combined with population to determine population exposure. To minimise computational and resource requirements the analysis was based on two episodes, one during the winter and the other during the summer. It was clearly understood that a more comprehensive analysis would cover further meteorological conditions.

The results of the analysis may be summarised as follows:

- The Business as Usual scenario is the worst option (with one exception) for all measures (smog and particulate matter exposure, greenhouse gas emissions and transportation energy use).
- The one exception is that the Compact City is worst when exposure to fine particulate matter is considered. This is because considerable use is made of wood burning for space heating in Melbourne in winter time. The corresponding particulate matter emissions are per capita based, which means that the Compact City format concentrates both the emissions and the exposed population. (This conclusion follows even though allowance was made for anticipated tighter control of wood burning emissions by 2011).

- Energy consumption and carbon dioxide emission are closely related, and for both of these the Compact City provided the lowest impact scenario.

Overall the conclusion may be drawn that urban form does affect ambient air quality, and that any type of planning is better than none.

Sensitivity of particulate matter concentration in Ontario to emissions

Although not intended for the development of policy, a recent modelling study carried out in Ontario illustrates the relationship between precursor emissions and ambient concentrations of particulate matter (PM). Atmospheric fine particulate matter is a complex phenomenon, because it may be emitted directly (primary PM), and is also formed by the reaction in the atmosphere of several precursor compounds, including SO_2, NO_x, VOC and ammonia (secondary PM). As is the case for ozone, the possibility of disbenefits also exists, i.e., increased PM concentration when emissions are decreased. A study was therefore carried out to provide a preliminary assessment of the sensitivity of PM concentrations to changes in emissions.

The Models-3/CMAQ system was used with a domain that extended from northern Florida to Hudson Bay, and from west of Chicago to the Canadian Maritime Provinces. Two episodes were considered in this study: one in July 1995 and the other in February 1998. Unlike the case of North American ozone, PM episodes can occur at any time of year. As discussed previously, the conclusions of this study apply rigorously only to these two episodes, but an effort was made to assess how representative the episodes were of high PM episodes in general.

The emission scenarios considered are summarised in Table 5.

Table 5. Emission scenarios for Ontario modelling.

Number	**Reduction**	**Area over which reduction was applied (D - entire domain, C - Canada only)**
1	50% SO_2	D
2	50% SO_2	C
3	45% NO_x	D
4	45% NO_x	C
5	50% primary PM_{10} and $PM_{2.5}$	D
6	50% primary PM_{10} and $PM_{2.5}$	C
7	45% VOC	D
8	45% VOC	C

The chemical composition of the particulate matter is different between the winter and summer episodes, being high in nitrate in the former and high in sulphate in the latter. This means that the precursor dependence is also different, so the two episodes will be discussed separately (Figure 5).

The impacts of the emission changes will differ in different regions of the modelling domain, depending on the spatial relationship between the area considered, the area in which the emissions are affected by the scenario, and the meteorological conditions, especially wind direction. The discussion which follows therefore considers several sub-domains, as defined in Figure 5.

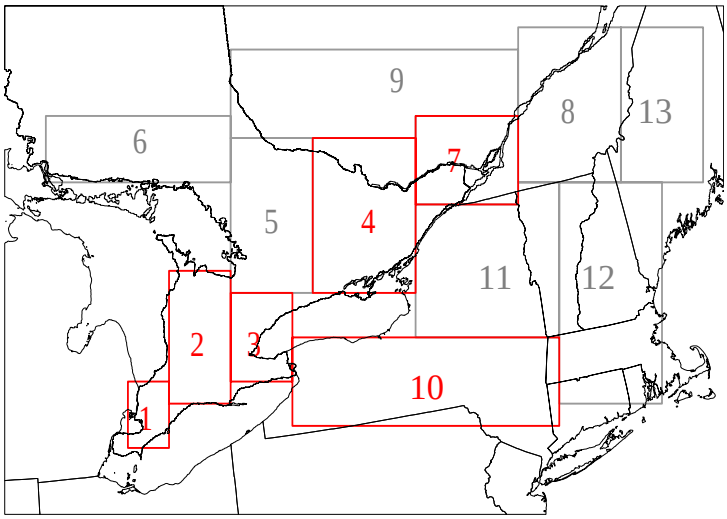


Figure 5: Subregions discussed in scenario modelling.

Summertime Episode

SO_2 reductions:
For this summer episode where the days with high modelled $PM_{2.5}$ were usually dominated by sulphate concentrations, reducing SO_2 by 50% across the domain resulted in 36 to 45% reductions in sulphate, accompanied by small reductions in ammonium and increases in nitrate concentrations. $PM_{2.5}$ reductions were 10 to 32%. Reducing SO_2 by 50% in Canada only resulted in 1 to 15% reductions in sulphate for the sub regions analyzed. The larger improvements were in sub regions 4 and 7, which are downwind of the major urban and industrial centres of Ontario. Reductions in ammonium and increases in nitrate concentrations again occurred. The net changes in $PM_{2.5}$ were reductions of up to 5%.

NO_x reductions:
Reducing NO_x by 45% across the domain resulted in small reductions in sulphate for the sub-regions analyzed. The reductions in nitrate were 26 to 46% with net reductions in $PM_{2.5}$ of 4 to 10%. The response to a 45% reduction in Canadian NO_x emissions was largest in sub regions 3, 4 and 7. For these three regions reductions in nitrate concentrations were in the 20 to 40% range, with corresponding reductions in $PM_{2.5}$ in the 1 to 6% range.

VOC reductions:
Reducing VOC emissions by 45% domain wide resulted in modelled reductions of non-biogenic secondary organic aerosols (SOA) of about 40 to 45%. The net changes in modelled $PM_{2.5}$ were reductions of 1 to 3%. Reducing Canadian VOC emissions by 45% resulted in modelled reductions of non-biogenic SOA ranging from about 2% in southwestern Ontario to 18% in sub region 7. The net changes in modelled $PM_{2.5}$ were reductions of 0 to 3%.

Primary PM reductions:
Reducing primary PM emissions by 50% domain wide resulted in modelled reductions in primary PM concentrations for all size ranges of 50% in all sub regions. The net changes in modelled total $PM_{2.5}$, i.e., primary plus secondary, were reductions of 7 to 19%. Reducing Canadian primary PM emissions by 50% resulted in modelled reductions in primary $PM_{2.5}$ concentrations of about 10% in southwestern Ontario and 25 to 40% in sub regions 3, 4 and 7.

Wintertime Episode

SO_2 reductions:
For the winter episode where the days with high modelled $PM_{2.5}$ were usually dominated by ammonium nitrate concentrations, reducing SO_2 by 50% across the domain resulted in 15 to 33% reductions in sulphate for the sub

regions analyzed. There were small increases in nitrate concentrations and decreases in ammonium concentrations, with net reductions in $PM_{2.5}$ of 1 to 2%.

Reducing SO_2 by 50% for Canada only resulted in 5 to 15% reductions in sulphate for all sub regions except region 10 (New York) where the reductions were smaller. Accompanying changes in nitrate and $PM_{2.5}$ were 1% or less.

NO_x reductions:
Reducing NO_x by 45% across the domain resulted in 6 to 32% reductions in nitrate. There were significant percentage increases in sulphate concentration, but the absolute changes were small. The net changes in $PM_{2.5}$ were reductions of 4 to 17% (but the reduction in sub region 7 was much smaller at 0.4%). Reducing NO_x by 45% in Canada resulted in 4 to 29% reductions in nitrate, again with increases in sulphate concentrations, of about 5 to 20%. The net changes in $PM_{2.5}$ were reductions of 2 to 14%.

VOC reductions:
Reducing VOC emissions by 45% domain wide resulted in modelled reductions of non-biogenic SOA of about 50 to 55%. Because of the small contribution of SOA to total modelled $PM_{2.5}$, reductions were limited to 2 to 8%. Reducing Canadian VOC emissions by 45% resulted in modelled reductions of non-biogenic SOA ranging from 30 to 44%. The net changes in modelled $PM_{2.5}$ were again small, with reductions of 1 to 6%.

Overall, the largest reductions in $PM_{2.5}$ in summertime were achieved by domain-wide reductions in sulphur dioxide emissions. Wintertime $PM_{2.5}$ is dominated by nitrate, and thus responds better to reductions in NO_x emissions.

The sensitivity of predicted PM concentrations to variation in ammonia emissions was also studied. This was necessary, not because control of ammonia emissions is anticipated, but because there are substantial uncertainties associated with the current ammonia inventory. Although the absolute PM mass predicted by the model is affected by the reduction in ammonia imposed in this sensitivity test, it is important to note that the percentage response to the emission reduction scenarios is very similar for the summer time episode. This lack of sensitivity of the response to absolute ammonia emissions gives enhanced confidence in the sensitivities to precursor emissions for the summer. There is somewhat greater variation with ammonia emissions in the winter time, with lower ammonia emissions being associated with lower reductions in PM concentrations when the precursor emissions are reduced. Improvement in the quality of the ammonia emission inventory is clearly required.

Cautionary Tales

To underline the need for care and experience in interpreting model predictions, two cautionary tales are presented below. These were taken from NARSTO (2003).

Kern County California 1985
Modelling results can be quite sensitive to the inputs provided and, in turn, to decisions made to construct those inputs based on available data, however rich or sparse. The experiences associated with modelling efforts conducted in 1985 in preparation for hearings to adopt NO_x emission control requirements for Kern County, California, provide an illustration.

Modelling using UAM-IV was conducted by two groups – state agency staff and a consultant retained by the private sector. Both groups used the same model and the same data base. Each was able to discuss matters with the other if they wished. One group determined that NO_x emissions were limiting in the area of highest O_3 concentrations, the other group that VOCs were limiting, with NO_x reductions having adverse effects on O_3 in some sub-areas. Each recommended that the pollutant found to be limiting in its analysis should be controlled. How could such a difference arise when so much of the two analyses shared the same information and approaches?

Subsequent investigation has uncovered two primary reasons. First, a sub-area in the eastern side of the county was lacking surface meteorological data. It was believed that flow in this sub-area formed a portion of an eddy of considerable size. In order to “give guidance” to the wind field interpolation program, one group inserted a “phantom” meteorological station with prescribed “data”. The other group did not; they instead allowed the

interpolation program to determine the shape of the flow in the area, including those portions in complex terrain. The net result was that each modelling effort produced somewhat different directions for the wind flow passing through the general area where the data were lacking. Since the emission fields were highly variable spatially, this led to mixtures of differing proportions of precursors being transported to the areas of high O_3 concentration.

Second, one VOC surface measurement was available, made in the morning, in a direction generally upwind. To establish initial conditions, one group extrapolated the measurement aloft, in essence setting the concentrations of VOC aloft at levels as high as at the surface. The second group assumed that, since VOC is emitted at the surface, and vertical mixing had yet to occur, VOC concentrations taper off sharply with elevation. Thus the first group specified a high VOC-to-NO_x ratio aloft, the second a much lower ratio. Moreover the group specifying the high VOC-to-NO_x ratio modelled only a relatively short time period; thus, initial conditions displayed an exaggerated impact on O_3 concentrations. The net result was that from two slightly different and reasonable sets of inputs, two very different mixtures of precursors in the relevant downwind area were produced. This, in turn, generated control strategies that were different in kind and not just in degree.

The Pitfalls of Model Simulations with Compensating Errors

Any operational evaluation that is based solely on a simple comparison of observed and model-calculated O_3 concentrations is unlikely to detect existing, compensating model errors. This situation is particularly problematic in the policy context because models having such errors may accurately reproduce a particular observed O_3 field, but for the wrong reasons. This, in turn, can instill a misplaced confidence in a model's predictive capability, possibly leading to inappropriate policy decisions based on its use.

The schematic shown in Figure 6 gives a simplified example of such a situation. It shows an EKMA (Empirical Kinetic Modelling Approach) plot, which is a depiction of the relationship between ozone concentration, shown as contours, and NO_x and VOC concentration. Ozone concentration is a minimum towards the lower left corner, increasing outwards. Here Point A represents the model's predicted O_3 concentration on an EKMA diagram using the actual (or true) NO_x and VOC emission inventories. Point B, on the other hand, represents the model's result obtained with a NO_x inventory that is biased low, combined with a high-biased VOC inventory. By pure coincidence these emission inventories compensate one another to produce the same model-predicted O_3 concentration for simulation B as for A.

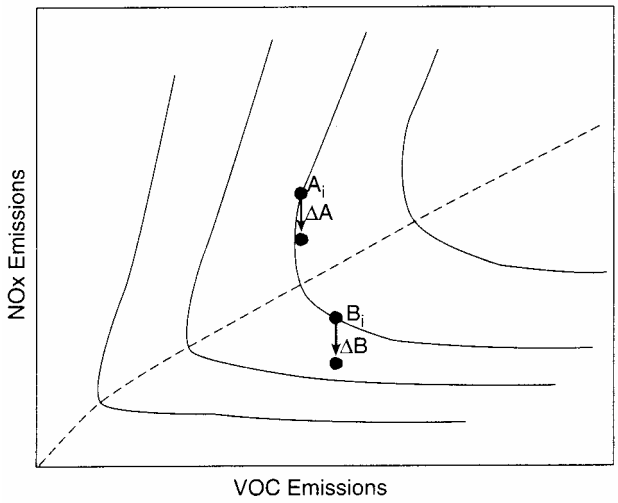


Figure 6: EKMA plot showing contours of ozone concentration as a function of NOx and VOC.

Now consider what happens if these simulations are used to estimate the effect of a hypothetical reduction in NO_x emissions. In the case of the correct emissions the solution moves to the new point on the EKMA diagram indicated by ΔA. As can be seen from the figure, this results in a slight increase in O_3. Conversely, the same NO_x-emission

adjustment to the simulation carried out using the incorrect inventories moves the solution a distance ΔB on the diagram. This corresponds to an O_3 decrease, a directionally incorrect result.

Thus we see that the presence of compensating errors in air quality modelling can pose a major pitfall for policymakers, and could, in principle, lead to the adoption of directionally incorrect control strategies. For these reasons, model applications in the policymaking arena should be carefully screened for compensating errors and related biases. This can be accomplished through rigorous operational evaluations using precursor species as well as O_3 diagnostic testing of modules, independent testing by two or more groups, and comparisons of model results with observations.

CONCLUSION

Atmospheric models constitute the best tools available for the setting of policy, and may, in some cases, be the only tools that are available. This is because the best examples of their kind bring together all current knowledge of pollutant behaviour in the atmosphere, making it possible to unravel the often complex interactions between pollutants and atmospheric dynamics. They also allow the possibility of evaluating hypothetical changes in emissions and other conditions to evaluate potential abatement strategies, or to assess the impact of proposed new emission sources.

However, successful model applications require careful planning and execution. First and foremost is the necessity for understanding the problem, both in terms of the health impacts and the atmospheric science. This includes knowing which pollutant or pollutants are important, and what measure of the pollutants is appropriate. In other words, is there concern about peak concentrations or about longer term average concentrations, or is population exposure a more relevant measure? With the realisation that many pollutants do not have a threshold exposure below which no adverse effects occur, very often comes the need to consider long term average concentrations or exposures. Readers are referred to the Health Effects background paper (Samet and Krewski, this issue) for further discussion of these issues.

Modelling long term effects, as opposed to studying short term episodes, presents difficulties. Apart from the additional input data requirements, running a model for a year takes 30 times as long (in computer time) as it does to run a 12 day episode. When this is compounded over the study of a number of scenarios, the cost quickly becomes prohibitive. Advances have been made in aggregation schemes, which assemble a long-term average from a suitable combination of representative, short-term episodes, but further development is required. It is also relevant to note that while concerns about ozone are mainly confined to the summer months, episodes of high particulate matter concentration can occur at any time of year, necessitating long term simulations.

Clear problem definition amounts to generation of a conceptual model of the situation. A more formal discussion of conceptual models is provided by NARSTO (2003). Based on this conceptual model, the most appropriate mathematical modelling tool can be chosen. Considerations which enter into the choice of model or modelling system include matching the model chemistry to the pollutants of concern, and ensuring that the spatial scale of the model is appropriate. If only one source, or a small group of sources, is important, a Lagrangian formulation may be most appropriate. This is also true if a single receptor, or small group of receptors, is of concern, otherwise an Eulerian model may be a better choice. It is, of course, necessary that the model be properly evaluated and validated, to ensure that the results, and the policies based thereon, are credible.

In fact model evaluation should be a continuous process, since different applications, with different mixes of emissions, meteorological conditions, etc., can lead to differences in performance. Peters et al. (1995) express this viewpoint very strongly, “A model developed or utilized without continual comparison against actual data is less than worthless: it is dangerous.”

Along with the model, an appropriate infrastructure is also essential. This will include all of the appropriate input data, a suitable computing platform, and skilled modellers to carry out the runs, and provide quality assurance and interpretation. These runs will involve scenarios, usually based on altered emissions, but possibly including other changes, such as different treatments of chemistry in the model, or altered meteorological conditions.

Model results will be scrutinised, analysed and interpreted, before being presented to the policymakers. If at all possible the analysis should include incorporation of measurements. Above all, we make a plea for the involvement of all parties – people who make measurements, people who run models, and people who make policy – at all stages of the process. If all of these steps are carried out, the guidance provided by atmospheric models to the policy development process can be used with confidence.

REFERENCES

Dastoor, A.P., and Larocque, M. 2004. Global circulation of atmospheric mercury: a modelling study. *Atmos. Environ.* 38:147-161.

Liousse, C., Penner, J.E., Chuang, C., Walton, J.J., Eddleman, H., and Cachier, H. 1996. A global three-dimensional model study of carbonaceous aerosols. *J. Geophys. Res.* 101: 19411-19432.

Manins, P.C., Cope, M.E., Hurley, P.J., Newton, P.W., Smith, N.C., and Marquez, L.O. 1998. The impact of urban development on air quality and energy use. Proc. 14th International Clean Air & Environment Conference, Melbourne, Australia.

NARSTO 2000. An Assessment of Tropospheric Ozone Pollution – A North American Perspective, NARSTO Management Office (Envair), Pasco, Washington.

NARSTO 2003. Particulate Matter Science for Policy Makers – A North American Perspective, NARSTO Management Office (Envair), Pasco, Washington.

Palacios, M., Kirchner, F., Martilli, A., Clappier, A., Martín, F., and Rodríguez, M.E. 2002. Summer ozone episodes in the Greater Madrid area. Analyzing the ozone response to abatement strategies by modelling, *Atmos. Environ.* 36:5323-5333.

Peters, L.K., Berkowitz, C.M., Carmichael, G.R., Easter, R.C., Fairweather, G., Ghan, S.J., Hales, J.M., Leung, L.R., Pennell, W.R., Potra, F.A., Saylor, R.D., and Tsang, T.T. 1995. The current state and future direction of eulerian models in simulating the tropospheric chemistry and transport of trace species: A review. *Atmos. Environ.* 29:189-222.

Tilmes, S., Mohnen, V., and Schaller, E. 1999. TFS Model Evaluation, NRW-94. Final Report.

TOOLS AND STRATEGIES FOR IMPROVING POLICY RESPONSES TO THE RISK OF AIR POLLUTION

Ari Rabl[1], Jatin Nathwani[2], Mahesh Pandey[3], Fintan Hurley[4]

ABSTRACT

This paper offers a brief review of the need for cost-benefit analysis (CBA) and the available policy instruments for air pollution. To prioritize different possible actions, one needs to know which source of pollution causes how much damage. This requires an impact pathway analysis, i.e. an analysis of the chain emission $\rightarrow$ dispersion $\rightarrow$ dose-response function $\rightarrow$ monetary valuation. The methodology for this is described and illustrated with the results of the ExternE (External Costs of Energy) project series of the European Commission. Two examples of an application to CBA are shown: one where a proposed reduction of emission limits is justified, one where it is not. It is advisable to subject any proposed regulation to a CBA, including an analysis of the uncertainties. Even if the uncertainties are large and a policy decision may have to take other considerations into account, a well documented CBA clarifies the issues and provides a basis for rational discussion. One of the main sources of uncertainty lies in the monetary valuation of premature mortality, the dominant contribution to the damage cost of air pollution. As an alternative an innovative policy tool is described, the Life Quality Index (LQI), a compound indicator comprising societal wealth and life expectancy. It is applied to the Canada-wide standards for particulate matter and ozone. Regardless of monetary valuation, a 50% reduction of PM_{10} concentrations in Europe and North America has been shown to yield a population-average life expectancy increase in the order of 4 to 5 months.

[1] Ecole des Mines, Paris, France.
[2] Management Science & Institute for Risk Research, University of Waterloo, Waterloo, Ontario, Canada.
[3] Civil Engineering, University of Waterloo, Waterloo, Ontario, Canada.
[4] Institute of Occupational Medicine, Edinburgh, Scotland UK.

INTRODUCTION

Air pollution has a variety of undesirable effects on human health, on buildings and materials, on agricultural crops and on ecosystems. There is now a general scientific consensus that air pollution causes health damage. Furthermore, the projects undertaken to quantify the costs of environmental damage have all come to the essential conclusion that the cost of health impacts by far outweighs damage from all other categories (ExternE, 1998; ORNL/RFF, 1994; Rowe et al., 1995; Abt, 2000) for the classical air pollutants (PM, NO_x, SO_2, O_3, VOCs, CO).

To reduce air pollution, several government regulations have been enacted over the past decades and these regulations have played a crucial role in curbing the emission of pollutants. Even though the regulations have become far more stringent than they were in the past, there is some debate about whether the current standards provide sufficient protection for human health. If stricter regulations are to be enacted, there is a policy imperative to determine whether the benefits exceed the costs. The rational foundation for introducing new and stringent regulation to control air pollution could be enhanced considerably if the benefits are quantified.

Past decisions about environmental policy have generally been made without quantifying the benefits. Initially, increasing demands for cleaner air were met by technical developments (such as flue gas de-sulfurization) without prohibitive costs. A simple criterion seemed adequate for making decisions. This criterion was based on the idea that a toxic substance has no effect below a certain threshold dose. If that is the case, it is sufficient to reduce the emission of a pollutant below the level where the highest dose anywhere is below the threshold. Standards for ambient air quality were developed, for example by the World Health Organization, and industry was required to reduce the emissions to reach these standards.

However, the situation is changing. Epidemiologists have not been able to find “no-effect thresholds” for air pollutants. The most recent guidelines of the World Health Organization indicate that there seems to be no such threshold for particulate matter (PM). The available evidence suggests that the dose-response function may be linear at low dose for PM, and quite possibly for other air pollutants as well. At the same time the incremental cost of reducing the emission of pollutants increases sharply as lower emission levels are reached. Thus the question "how much to spend?" acquires growing urgency. General principles such as sustainable development or the precautionary principle provide no answer (except in their most extreme and totally impractical interpretation of demanding zero pollution) because the difficulties lie in the specifics of each situation. One needs cost-benefit analysis (CBA). Of course, a cost-benefit analysis of air pollution should include all benefits, not just those due to health impacts.

Environmental CBA is often controversial. The objection to environmental CBA is often based on the view that one should not assign monetary values to goods such as a beautiful landscape, the existence of a rare animal or human life. However, this objection is less persuasive if we consider monetary valuation not in terms of the intrinsic value of the item in question but the collective willingness to pay (WTP) to avoid losing the item. For instance our WTP (including ability to pay) to avoid the risk of a premature death is limited, even if we feel that the value of life is infinite.

Above all, a thorough and well-documented CBA can provide a systematic assessment of the consequences of a decision before it is too late, and by clearly exposing the assumptions, it facilitates informed discussion of disagreements; for these reasons a CBA should always be attempted for important decisions even if the uncertainties are large. A CBA should however be used with care, firstly because of the large uncertainties, secondly because it may be desirable to take non-monetary considerations into account, for example the perception of risks and the distribution of costs and benefits among the population. Having estimated monetary values for all the benefits that can be quantified, any remaining non-monetary considerations can be evaluated by means of multicriteria analysis. Thus, the benefits of a CBA lie in what is made transparent in the process of doing it, as well as in the answers it provides.

The extra cost of a cleaner environment must be paid, ultimately by tax payer or consumer. Even if immediate trade-offs do not cross budget categories, ultimately the money we spend on reducing the emission of pollutants is not available for other causes such as the education of our children.

Links can be subtle and unexpected. When evaluating a decision, it is necessary to consider the consequences of alternatives and unintended effects. For example, lowering the limit for the allowable emission of dioxins from waste incinerators will avoid some cancer deaths, but people will have to pay more for waste disposal. Such costs induce effects elsewhere in the economy. For example, in the USA Keeney (1994) has shown that for each $ 5 to 10 million of cost imposed by a regulation there will be on average one additional premature death due to this cost.

The value of achieving clean air objectives must be commensurate with the benefits and it is important to aim for a level that is optimal for society.

POLICY INSTRUMENTS

In recent years the term "external cost" has been widely used to designate the costs of environmental damage. The term damage cost is more appropriate since it avoids ambiguities that arise with the use of at least two definitions of external cost:

1) costs imposed on non-participants, that are not taken into account by the participants in a transaction;
2) costs imposed on non-participants, that are not paid by the participants in a transaction.

According to the first definition a damage cost is internalized if the polluter reduces the emissions to the socially optimal level, for example as a result of a regulation that imposes an emission limit. The second definition requires, in addition, that the polluter compensate the victims for any damage, for example by paying a pollution tax. In either case, the level of emissions is equal to the social optimum. But the corresponding damage cost is external only according to the second, not the first definition. Some economists have used the term "relevant externality" to designate the portion (if any) of the damage cost that is greater than the social optimum.

Between these two levels of internalization there is a substantial difference in the costs borne by the polluter, as shown by Desaigues and Rabl (2001). In practice the argument for full internalization according to the second definition loses its justification, since it is almost impossible to compensate the victims correctly (identifying who suffers how much damage is too difficult and uncertain).

Of course, government intervention is necessary to reduce the emission of air pollutants. There are various possible policy instruments and many different ways of implementing them. For brevity we list in Table 1 the principal types of regulations or policy instruments that have been used for this purpose. Some of the options when adopted have a direct impact on the emissions whereas others such as eco-labels and portfolio standards can affect emissions indirectly by reducing the consumption of materials or energy. At one end of the spectrum, characterized as "command-control", are regulations that impose rigid and specific constraints (e.g. a limit on the concentration of SO_2 in the flue gas of power plants). At the other end are market mechanisms: the government can propose certain general goals or targets, for instance a national emission ceiling or a tax per ton of SO_2, and let the markets respond by providing an appropriate solution. The middle column of the table indicates whether the regulations are based on command-control (C) or on market mechanisms (M).

A command-control approach (provided, of course, that compliance is enforced) yields predictable results (e.g. the specified reduction of SO_2), but often the costs are high because all polluters must take the same action. The costs of pollution abatement depend on specific local circumstances and vary greatly from one polluter to another. For instance, they are much higher for an industry that must install an expensive retrofit than for one that can include the pollution control equipment in the early design phase of a new factory. Under a pollution tax, there is some flexibility and the owner can choose and optimize how much of the pollutant to remove by abatement equipment, paying the tax on whatever remains. Whereas a pollution tax achieves reductions at the lowest possible overall cost for society per avoided unit of pollutant (highest economic efficiency), the magnitude of the realized reduction is difficult to predict.

Table 1. Policy instruments for reducing air pollution.

Type	C or M	Examples
Limits on emission of pollutants	C	max. mg SO_2 per m^3 of flue gas; max. g CO per km driven by cars
Choice of technologies	C	usually by demanding "Best Available Technology" (BAT), e.g. flue gas desulfurization for coal or oil fired boilers
Broad initiatives to reduce emissions of an entire sector or country	depends on implementation	The National Emission Ceilings of the EU
Subsidies for clean technologies	M	tax credit for wind and solar in California during 80s
Eco-labels	M	"printed on recycled paper"; "no chlorine used"; "energy star" label for computers
Pollution taxes	M	€/ton of a pollutant
Tradable permits	M	government sets cap on number of permits (e.g. ton of SO_2), polluters can trade these permits
Portfolio standards	M	government sets minimum % for the market share of a clean technology, e.g. "zero emission" vehicles in California, or “green kWh” from solar energy, and industry adjusts the prices to achieve these goals.

Tradable permits are a policy instrument that combine the highest economic efficiency with predictable results. Under this system the government issues permits for a specified quantity of a pollutant that may be emitted in a region, and industry can freely buy or sell these permits. There are several variants, the two main distinctions being whether the government sells the permits at an auction or gives them away (for instance to each polluter according to last year’s emission). Obviously, industry prefers the latter. In the USA tradable permits have been used for SO_2 for the last decade already, with great success: the cost per avoided kg turned out to be much lower than under the previous regime of command-control.

There has been some opposition to the idea, especially from people who misunderstand “permit” to mean unlimited license to pollute and do not recognize that the most widely used regulation, namely emission limits, is de facto a permit (to emit up to the specified limit) that is given away freely but cannot be traded. And tradable permits are not an unlimited license to pollute: the polluter incurs a cost for each kg of emitted pollutant.

At this point we can make several general recommendations:

- Market instruments yield a better allocation of societal resources than command-control.
- Tradable permits that are given away free are preferable to permits that are auctioned or to pollution taxes because they imply much smaller changes in the costs incurred by industry (for the same reduction of emissions) and thus less perturbation of the economy when they are introduced or modified (Desaigues and Rabl, 2001).
- Subsidies are treacherous because experience has shown how difficult they are to remove when they are no longer justified; they should be used only if automatic termination can be guaranteed.
- Specific policy proposals should be carefully evaluated before application to ensure that they are cost-effective and that they will not entail unexpected harmful side effects. In the past most regulations had not been subjected to a cost-benefit analysis before passage, and some have not been well chosen (see the examples later in this paper).
- Compliance with government regulations must be verified and enforced. It is desirable for the process to be clear and transparent. In this context one could add that major polluters should be required to post their emissions data on the internet rather than treating them like an industrial secret as has so often been the case, for example with utility companies in many countries of Europe.

THE NEED FOR IMPACT PATHWAY ANALYSIS

Policy decisions must act on the sources of pollution. To provide adequate guidance to the formulation of policies it is not sufficient to calculate the damage per exposure; one needs to know which source of a pollutant causes how much damage. This requires an impact pathway analysis (IPA), tracing the passage of the pollutant from where it is emitted to the affected receptors (population, crops, forests, buildings, etc.), as shown in Figure 1. The impacts and costs are summed over all receptors of concern. Since the dispersion of air pollutants is significant over distances of hundreds to thousands of km, the analysis must account for all impacts at the scale of an entire continent.

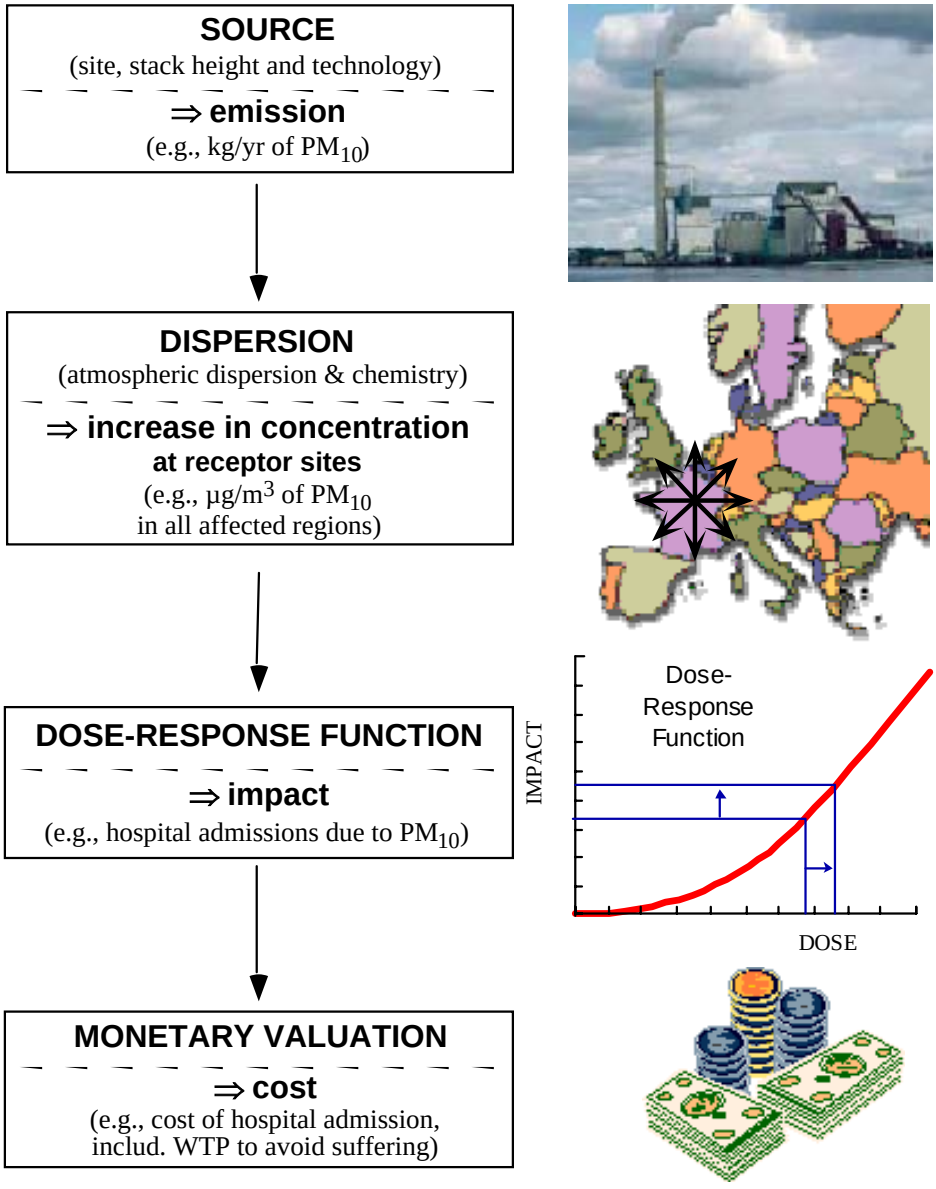


Figure1. Impact pathway analysis for the example of an air pollutant.

This requires a multidisciplinary system analysis with inputs from engineers, dispersion modelers, epidemiologists, and economists. The largest and most up-to-date effort of this kind is the ExternE (External Costs of Energy) project series of the European Commission (EC) which began in 1991 and is still continuing. Initially ExternE was carried out jointly with the USA (ORNL/RFF, 1994), but the US participation stopped in 1995. Some work analogous to ExternE has continued (USDOE, 2003), and other projects that parallel efforts of ExternE include the work of IIASA on the RAINS model (see e.g. Mechler et al., 2002), of Kuenzli and colleagues (Kuenzli et al., 2000), and of Ostro, Rowe and co-workers. All of these studies find that health impacts account for well over 90% of the quantifiable damage cost of air pollution other than global warming, and that among health cost the cost of mortality is by far the largest. Therefore the monetary valuation of air pollution mortality is a central issue. First we present some results of ExternE followed by a discussion of monetary valuation. A very brief summary of the methodology is given in Appendix A; for more detail the reader is referred to the ExternE reports or to review papers (e.g. Rabl and Spadaro, 1999).

RESULTS OF EXTERN E

Damage Costs per kg of Pollutant

The output of an impact pathway analysis is the damage cost per kg of an emitted pollutant, given the site and conditions of the source. Some results for typical French conditions are presented in Figure 2. For primary pollutants there is a strong dependence on site of source and height (h) of source above ground; for secondary pollutants the variation with site is much smaller (about +-30% for sources in France) and the variation with h is negligible. Note that the results cited in different reports of ExternE can be somewhat different because the methodology has been evolving. The key assumptions for the numbers shown here are listed in Table 2.

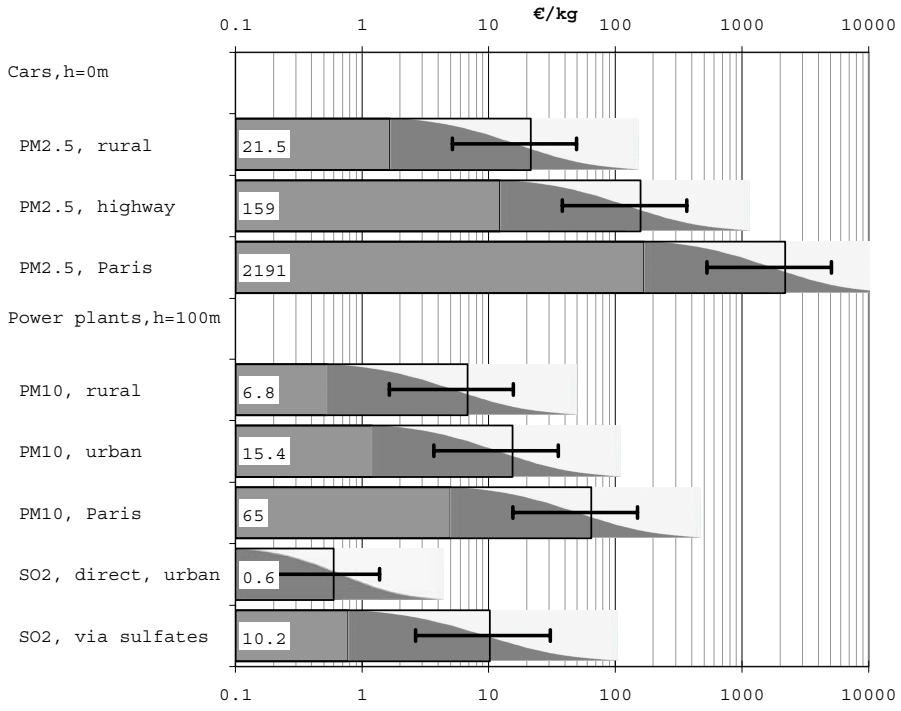


Figure 2. Typical damage costs for PM and SO_2 (values of ExternE 1998 for France), together with the uncertainty range. The gray curve indicates the probability that the true cost is below a specified value. The mean damage costs are shown in the labels. On a log plot the distribution is symmetric about the median which is lower than the mean. The error bars indicate 1 geometric standard deviation (68% confidence interval).

Table 2. Key assumptions for the damage costs cited in this paper.

Atmospheric dispersion models	
Local range:	gaussian plume models ISC and ROADPOL.
Regional range (Europe):	Harwell Trajectory Model as implemented in ECOSENSE software of ExternE. Ozone impacts based on EMEP model
Impacts on health	
Form of dose-response functions	Linearity of incremental impact due an incremental dose (e.g. $\mu g/m^3$ ambient pollutant concentration) for all health impacts.
Chronic mortality	Dose-response function slope = 4.1E-4 YOLL (years of life lost) per person per year per $\mu g/m^3$ derived from increase in age-specific mortality due to $PM_{2.5}$ (Pope et al. 1995), by integrating over age distribution.
Acute mortality	For SO_2 and ozone, with 0.75 YOLL per death.
Nitrate and sulfate aerosols	Dose response functions for nitrates same as for PM_{10}. Dose response functions for sulfates same as for $PM_{2.5}$ (slope = 1.7 times slope of PM_{10} functions).
Micropollutants	Only cancers have been quantified (As, Cd, Cr, Ni, dioxins, benzene, butadiene).
Monetary valuation	
Valuation of premature death	Proportional to reduction of life expectancy, with value of a YOLL (years of life lost) derived from VSL = 3.4 M€: 96.5 K€/YOLL for mortality.
Valuation of cancers	0.45 M€ nonfatal cancers, 1.5 to 2.5 M€ (depending on YOLL) fatal cancers, 1.5 M€ average for cancers from chemical carcinogens.
Discount rate	3% unless otherwise stated; results for nuclear are shown for 0% "effective discount rate" (=discount rate – escalation rate of cost).

YOLL = years of life lost, VSL = value of statistical life

Uncertainties

The uncertainties in this domain are very large. They should be evaluated, to give decision makers an idea of the reliability of the results. Uncertainties can be grouped into different categories, even though there may be some overlap:

- data uncertainty (e.g. slope of a dose-response function, cost of a day of restricted activity, and deposition velocity of a pollutant);
- model uncertainty (e.g. assumptions about causal links between a pollutant and a health impact, assumptions about form of a dose-response function (e.g. with or without threshold), and choice of models for atmospheric dispersion and chemistry);
- uncertainty about policy and ethical choices (e.g. discount rate for intergenerational costs, and "value of statistical life");
- uncertainty about the future (e.g. the potential for reducing crop losses by the development of more resistant species);
- idiosyncrasies of the analysts (e.g. interpretation of ambiguous or incomplete information).

The first two categories (data and model uncertainties) are of a scientific nature. They are amenable to analysis by statistical methods, combining the component uncertainties over the steps of the impact pathway, to obtain formal confidence intervals around a central estimate. For this, ExternE followed an approach based on lognormal distributions and multiplicative confidence intervals. For quantifying the sources of uncertainty, a survey was carried

out of experts and relevant information available in the literature. The results of this analysis are shown Figure 2; the error bars are one-geometric standard deviation intervals around the median estimate. The largest sources of uncertainty lie in the dose-response functions for health impacts and in the value of a life year. Details can be found in Rabl and Spadaro (1999).

One of the sources of uncertainty lies in the difficulty of identifying exactly which air pollutant causes how much damage, since epidemiological studies are carried out under real conditions of exposure to a mix of pollutants. Thus the total health damage attributable to all air pollutants is probably more certain than the individual damage costs for each pollutant. This becomes important when evaluating the effects of sources that generate mixtures which differ markedly from the general urban mixture as a whole. For example, the estimated damage costs of gas-fired power stations are much more dependent on judgments made about the health effects of NO_2 as a gas, and of nitrates as secondary particles, than are the estimated costs of coal-fired stations, where judgments about SO_2 and sulfates are more influential.

One should note that the full uncertainty is larger than the data and model uncertainties that have been quantified explicitly and shown in Figure 2.

Use of ExternE Results

Gradually the results of ExternE are diffusing into the world of decision makers. For example, ExternE is recognized as the reference for comparative risk assessment by agencies such as the International Atomic Energy Agency. In the EU, ExternE is increasingly used as input to environmental decisions, for example via cost-benefit analyses (Holland, 2001). Of course, a cost-benefit analysis of air pollution should include all benefits, not just those due to health impacts.

Figure 3 is a comparison of social costs and benefits for a proposed reduction of the emission limit for particulate matter (PM) emitted by cement kilns that use waste as fuel, one of the issues under discussion in formulating the new EC Directive on the incineration of waste. Even the upper bound of the benefit is lower than the lower estimate of the abatement cost. Clearly this proposed emission limit cannot be justified by a cost-benefit criterion (see Rabl (2000), a paper which contributed to the decision to require only 30 mg/m^3, not the lower limit of 15 mg/m^3 which had been proposed).

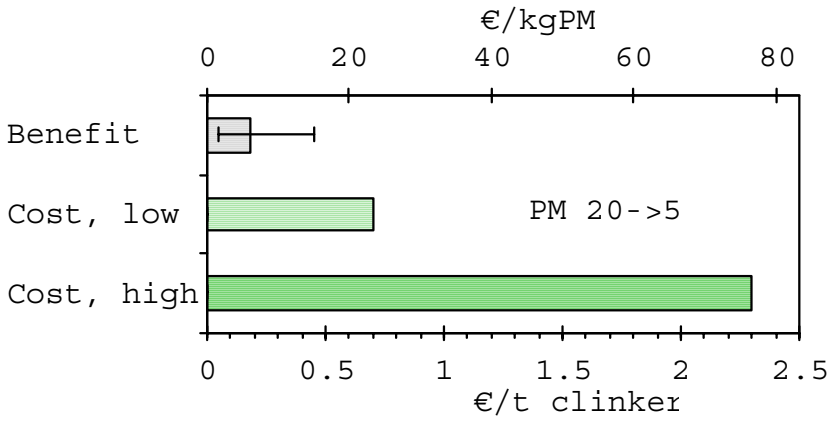


Figure 3. Comparison benefits for a reduction of average emission from 20 to 5 mg/m^3 (of emission limit from 50 to 15 mg/m^3). Costs and benefits are shown on two scales: per $t_{clinker}$ (bottom) and per kg_{PM} (top). Error bar indicates uncertainty of benefit.

By contrast Figure 4 shows an example where the proposed reduction of emission limits is justified (Rabl et al., 1998a). This figure compares cost and benefit for a reduction of emission limits for PM and SO_2 from municipal solid waste incinerators: the limit for PM being reduced from 30 to 10 mg/m^3 and the one for SO_2 from 300 to 50 mg/m^3. Whereas in Figure 3 the benefit is shown for a single site, cement kilns being typically located in rural sites, in Figure 4 it is appropriate to show at least three sites for incinerators: Paris (population of about 10 million, including suburbs), a typical urban site and a rural site. For all of these sites the benefit outweighs the cost. These reduced emission limits have been incorporated in the above mentioned EC Directive on the incineration of waste.

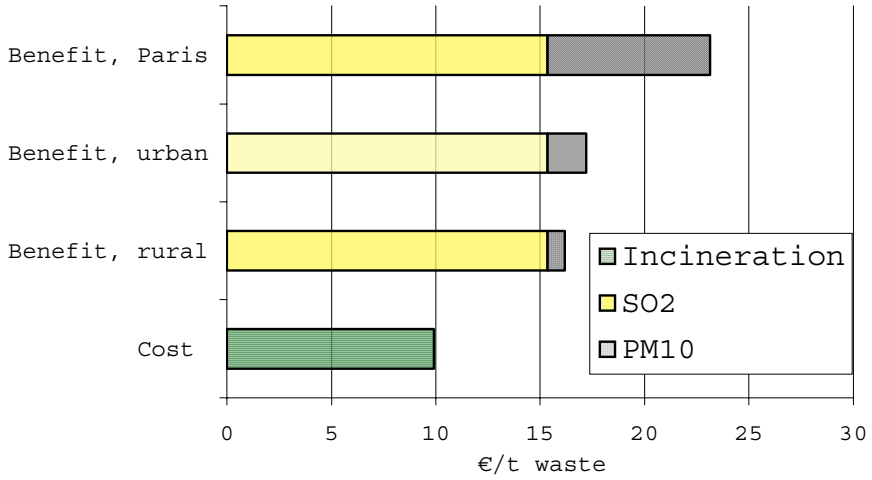


Figure 4. Cost and benefit for a reduction of emission limits for PM and SO_2 from municipal solid waste incinerators.

The difference between Figures 3 and 4 arises from the differences in technology. For waste incinerators the abatement technology under consideration reduces both PM and SO_2. The technology for cement kilns is different and reduces only PM (apart from rare exceptions cement kilns emit no SO_2).

As another application of ExternE we mention a CBA of the particle filter for diesel buses (Rabl et al., 1998b), where we found that this technology is justified by a wide margin in Paris, but not for rural bus routes. For a more recent CBA of particle filters, also for trucks and passenger cars, see Massé (2003); it finds that this technology is now mature and with current costs can be justified even for passenger cars.

The lesson to be drawn from these examples is that in some cases stricter limits for the emission of pollutants are clearly justified, in others they are not. A careful analysis should be carried out before implementing new policies, to avoid wasteful allocation of resources.

As emphasized in the preceding section, the uncertainties of the benefits are very large. But the abatement costs are also uncertain. For mature and widely used control technologies, the abatement costs are reasonably well known, but for the more important case of new technologies or new applications their estimation is often very difficult and uncertain. Often the industries concerned by a proposed reduction of pollution may not have all the required information, and if they do they may prefer to keep it confidential.

MONETARY VALUATION OF AIR POLLUTION MORTALITY

Ground rule

The goal of the monetary valuation of damages is to account for all costs, market and non-market. For example, the valuation of an asthma attack should include not only the cost of the medical treatment but also the willingness-to-pay (WTP) to avoid the residual suffering. If the WTP for a non-market good has been determined correctly, it is like a price, consistent with prices paid for market goods. Economists have developed several tools for determining non-market costs; of these tools contingent valuation (CV) has enjoyed increasing popularity in recent years (Mitchell & Carson, 1989). The results of well conducted studies are considered sufficiently reliable to be used in policy applications – see previous Section.

"Value of statistical life" VSL and Value of a Life Year

The "value of statistical life" (VSL) is such an important parameter for public policy that economists have carried out numerous studies to determine it (far more than a hundred, the vast majority in anglo-saxon countries). There are basically three approaches that have been used to determine how much an individual is willing to pay to reduce a risk of premature death:

i) comparisons of the relation between wage and risk in different professions with different levels of risk;
ii) the purchase of goods such as smoke detectors that reduce risks;
iii) contingent valuation (direct interrogation by means of questionnaire or interview).

The estimation of VSL has been a challenging and controversial topic in risk analysis. Empirical estimates of VSL typically range from $1 million to $10 million as reported in a review of literature (Tengs et al., 1995). For example, in the cost-benefit analysis of a Canadian Air Quality Standard, $4.1 million is an age-adjusted central estimate of VSL (CWS, 1999).

There has been no single official value used and recognized by all government agencies. ExternE takes VSL as the average of all the VSL studies that had been carried out in Europe when the ExternE project started in 1991. After adjustments for inflation, this amounts to 3.1 M€ in 1995. It is interesting to note that in the USA EPA uses values that are about twice as high. In France, by contrast, the report of Boiteux (2001) on external costs of transport recommends 1.0 M€.

A crucial question for air pollution mortality is whether one should simply multiply the number of premature deaths by VSL, or whether one should take into account the years of life lost (YOLL) per death. All studies before 1996, and in the USA until now, have done the former. The valuation issue is linked to the epidemiological one, of whether the estimates of mortality impacts are derived from studies of very recent pollution only ('acute mortality', from time series studies) or from studies of longer-term exposure ('chronic mortality', from cohort studies). Studies before 1996 quantified acute mortality only. ExternE (1995) estimated chronic mortality also, but only as a sensitivity analysis. By 1998 ExternE had incorporated chronic mortality estimates into its core analyses. They proved to be the dominant health effect. This is now the generally accepted approach.

There are several reasons why number of deaths × VSL is not appropriate for air pollution mortality.

a. First of all, premature deaths from air pollution tend to involve far fewer YOLL per death than accidents (on which VSL is based). This is certainly the case for acute mortality, where the earlier ('extra') deaths occur principally among people who are older and, almost certainly, particularly vulnerable for their age. The time series studies do not provide any direct information about YOLL, but it is widely understood that this is short, typically several months rather than many years.
b. Secondly, air pollution is a contributing, not the only, cause of the mortality of individuals, and the sum of deaths due to all contributing causes would be far greater than the total number of deaths in a population; by contrast, YOLL from different causes can be added to yield a total that is meaningful.
c. Furthermore, estimates of how long-term exposure to pollution affects mortality are best carried out using life table or life expectancy methods. These lead naturally to impacts in terms of total YOLL in a population, rather

than to 'extra' deaths (ExternE 1998; Hurley et al., 2000; Leksell and Rabl, 2001; COMEAP, 2001). Indeed, the total number of deaths due to air pollution cannot be determined whereas the total number of YOLL can (see, e.g., (COMEAP, 2001; Rabl, 2003) – the available methods cannot distinguish whether only some individuals are affected, with a substantial loss of LE, or whether everybody is affected; in the latter case the number of deaths due to air pollution would be equal to the total number of deaths.

Until now there have been almost no published studies that determine the value of a YOLL directly, by contrast to the numerous VSL studies, most of them based on accidents. Therefore ExternE has derived the value of a YOLL from VSL, by assuming that VSL is the present value of a discounted series of annual YOLL values. The ratio of the resulting value of a YOLL and VSL depends on the discount rate; it is typically in the range of 20 to 30. ExternE (1998) uses 84000 $€_{1995}$ per YOLL. One of the tasks of the current phase of ExternE is a contingent valuation (CV) to determine the value of a YOLL directly. Results will be published soon. Another study with similar aims, for the UK Government (DEFRA), by Mike Jones-Lee and colleagues, is also nearing completion and results will be available shortly.

ExternE also assumes 0.45 M€ for nonfatal cancers, and 1.5 to 2.5 M€ for fatal cancers (depending on the YOLL for each cancer type).

It is reasonable to think that the value of a YOLL should contain an adjustment for age to account for the fact that reduction in air pollution lowers death rates primarily among older persons. However, the empirical basis for age adjustment is weak and the practice is controversial: most older people do not like to hear that their life years are valued less. Furthermore, a discount factor should be applied because of latency, i.e. the time between exposure and premature death. Such factor should account both for the time value of money and for the perceived utility of an extra life year in the future. At the present time the empirical evidence on which to base judgments about the latter is weak; the underlying cohort studies are not informative about this aspect.

Current estimates of the monetary value of a life year lost due to air pollution are extremely uncertain. Furthermore, even when or if reliable CV studies for this purpose become available, they will be limited to developed countries. There is an increasing demand for the valuation of mortality in the rest of the world where CV will not be practical in the foreseeable future. Therefore it is advisable to develop alternative methods.

One option is to use values implicit in public decisions, for example in the health sector; this is described in Appendix B. However, one finds that the implied value of "cost per life year" can vary by about six orders of magnitude. Such estimates tend to reflect subjective decisions (preferences) of program administrators, often under the influence of unrepresentative or ill-informed pressure groups, rather than being representative of tradeoffs or peoples' willingness-to-pay for risk reduction. More importantly, the nature of risks in many life saving interventions is different than that associated with environmental interventions.

Another, very promising alternative is the life quality index (LQI), described briefly in the next section, with more detail about its derivation in Appendix C.

MONETARY VALUATION USING THE LIFE QUALITY INDEX (LQI)

Key Principles of the Life-Quality Index

The Life Quality Index (LQI) is a compound social indicator comprising societal wealth and longevity. It can also be interpreted as a utility function that is consistent with several principles of decision analysis. It has recently been applied to the cost-benefit analysis of pollution control programs (Pandey and Nathwani, 2003).

The proposed framework is intended to satisfy some basic reasoning and principles of risk management in public interest, namely, accountability, maximum net benefit, compensation and life measure, which have been discussed in detail elsewhere (Nathwani et al., 1997; Nathwani and Narveson, 1995). It incorporates the following principles:

(i) A unified rationale for application to all risks, if we are to have a working basis for practical professional action in society's interest when risks to life, health or property are important.
(ii) Maximizing the total expected net benefit to society. This principle has been accepted as fundamental to cost-benefit analysis. It satisfies the utilitarian concept of welfare, i.e., the greatest good for the greatest number. A simple and meaningful test of the effectiveness of allocation of scarce resources is: how much life saving does risk reduction buy, and could the same resource, if directed elsewhere, bring a better gain for society as a whole?
(iii) Compensation to ensure implementation of a policy is socially beneficial where there is a need to compensate the losers.
(iv) Enhancement of a relevant measure of life by maximizing the net benefit in terms of quality of life in good health for all members at all ages.

Definition of the Life-Quality Index

The LQI for a society is derived (see Appendix C) as

$$L = G^q E \tag{1}$$

where G is the real gross domestic product per person/year, E is the age-adjusted life expectancy in the country, and q is the elasticity of utility of consumption. q is related to a measure of labor productivity; for industrialized nations a typical value is 0.15.

The LQI consists of two major indicators: the real gross domestic product per person as a measure of resources and the quality of life (UN, 1990), and life expectancy which is a validated universal indicator of social development, environmental quality and public health (Gulis, 2000). Both indicators have been in use for half a century to express the wealth and health of a nation in numbers, and they are reliably measured.

Judging Risk with the Life Quality Index

Any project, program or regulation that materially affects the public by modifying risk through expenditure will have an impact on the Life-Quality Index. The net benefit criterion requires that a small change in the LQI due to a project or regulation should be positive or,

$$\frac{dL}{L} = q\frac{dG}{G} + \frac{dE}{E} \geq 0 \tag{2}$$

Here *dG* may represent the monetary cost of implementing a regulation (*dG* negative) or the monetary benefits that arise from a project (*dG* positive). The term *dE* is the change in life expectancy due to a change in the level of risk to the population associated with a project or, regulation.

The concept of societal WTP originates from the definition of compensating variation by Hicks (1939). It is the sum received by or from the individuals which, following a welfare change, leaves them at their original level of welfare. It can obtained from Eq.(4) by setting $dL/L = 0$ and rearranging the terms as

$$(-dG) = \frac{G}{q}\frac{dE}{E} \text{ (\$/person/year)} \tag{3}$$

Suppose benefits of a safety regulation are received by a population of size *N*, the aggregated value of societal WTP, i.e., the amount that will not alter the population life-quality (*C*) is equivalent to

$$C = (-dG) \times N = \frac{NG}{q}\frac{dE}{E} \text{ (\$/year)} \tag{4}$$

We propose the LQI-based measure of societal WTP for the valuation of mortality reduction in the cost benefit analysis of air-quality standards.

Application to the Canadian Air Quality Standard

Pandey and Nathwani (2003) have applied the LQI model to calculate monetary equivalent of benefit of reduction in mortality resulting from scenarios of improving air quality, which were studied during the development of a Canada Wide Standard (CWS) (CWS, 1999). The results of a cost-benefit analysis based on a simple multiplication of VSL and number of deaths are summarized in Table 3. Starting from the numbers in this study, with LQI the results in Table 4 were obtained for three rates of time preference r_{tp} (discounting of life years), 0%, 2% and 4%. The benefits associated with options to reduce particulate matter always outweigh the pollution control costs, as evident from benefit/cost ratios ranging from about 2 to 26. On the other hand, for all ozone options, these ratios are less than one, and so they do not satisfy the LQI criterion. Assuming that a rate of time preference of 2% is representative, the overall benefit/cost ratio of the Standard turns out as 1.9. Consideration of the effect of time preference rate is important, as benefit estimates can vary quite strongly when this rate changes by as little as one percent.

Table 3. Valuation of benefits of the Canada Wide Standard based on a simple multiplication of VSL and number of deaths, in Canadian $\$_{1996}$ (CWS, 1999).

Target Pollutant Level	**Avoided Mortality**	**Benefit of avoided mortality** [a]	**Cost**	**Benefit/Cost Ratio**
	(death/yr)	(million C$/yr)	(million C$/yr)	
$PM_{10}/PM_{2.5}$ ($\mu g/m^3$)				
70/35	1,021	4,186	170	24.6
60/30	1,639	6,720	620	10.8
50/25	2,790	11,439	1,600	7.1
Ozone (ppb)				
70	167	685	790	0.9
65	203	832	1,871	0.4
60	239	980	6,502	0.2
CWS				
$PM_{10}/PM_{2.5}/O_3$				
60/30/65	1,842	7,552	2,491	3.0

[a] Using central estimate of VSL = C$ 4.1 million/person; base year 1996, and discount rate 5%

Table 4. Cost-benefit analysis of Canada Wide Standard for Air Quality (CWS, 1999) using LQI approach.

Target Pollutant Level	**Cost (million C$/yr)**	**Benefit/Cost Ratio**		
		r_{tp} = 0%	r_{tp} = 2%	r_{tp} = 4%
$PM_{10}/PM_{2.5}$ ($\mu g/m^3$)				
70/35	170	26.6	15.6	9.0
60/30	620	11.7	6.9	4.0
50/25	1,600	7.7	4.5	2.6
Ozone (ppb)				
70	790	0.9	0.5	0.3
65	1,871	0.5	0.3	0.2
60	6,502	0.2	0.1	0.1
CWS[3]				
$PM_{10}/PM_{2.5}/O_3$				
60/30/65	2,491	3.3	1.9	1.1

CONCLUSIONS

1. In response to increasing evidence that particulate air pollution and ground level ozone have adverse impacts on public health and environment, stringent air-quality standards are under development worldwide.
2. The economic justification for pollution control programs largely rests on two aspects: using cohort studies to quantify the effects of long-term exposure on mortality, an approach which is now widely accepted; and the monetary valuation of reduction in mortality, which is a critical and controversial element of cost-benefit analysis.
3. The most relevant damages caused by air pollution (other than global warming) can be quantified and monetized using the impact pathway analysis of ExternE, albeit the uncertainties are large.
4. In many cases the accuracy of ExternE estimates is sufficient to provide guidance to a decision-maker, despite the uncertainties. For often, the policy issue or a problem related to the resolution of an environmental concern involves a "yes or no" type of question: "is the benefit greater than the cost?"
5. One of the main sources of uncertainty lies in the monetary valuation of air pollution mortality. The widely used "value of statistical life" is based on accidents and not appropriate, and the available estimates of the value of a life year lost due to air pollution are still very uncertain.
6. To circumvent the uncertainties of the valuation of air pollution mortality, the Life Quality Index LQI is proposed as an innovative policy tool because it allows integration of the key issues (discounting of life years, competing mortality risks, inter-temporal tradeoffs, age-dependent risks, and willingness to pay) in a consistent and transparent manner to support a credible analysis.
7. Assessment of potential health benefits of environmental policies, based on ExternE results, has clearly shown the gain in life expectancy that could be achieved in Europe and North America if the concentration of PM_{10} is reduced by 50%: the gain would be roughly 4 to 5 months (averaged over the entire population). This is a finding of enormous significance when compared to other measures for improving public health.
8. It is highly desirable to subject proposed environmental strategies to a cost-benefit analysis to help avoid costly mistakes and ensure that our scare resources are spent wisely. The necessary tools are now available. Of course, cost-benefit analysis in this domain is fraught with risks due to the large uncertainties and the subjective nature of some of the assumptions that may be made by the analyst. Whereas cost-benefit analysis must be used with caution, and not as the only criterion for a decision, it does provide a valuable framework to help clarify the issues.

Acknowledgments

The work of Ari Rabl has been supported in part by the ExternE project series of the European Commission, DG Research; Fintan Hurley has been supported by ExternE and by the UK Department of Health. We thank numerous colleagues who have been involved in discussing and developing the ideas presented here.

REFERENCES

Abbey, D.E., Lebowitz, M.D., Mills, P.K., Petersen, F.F., Lawrence Beeson, W., and Burchette, R.J. 1995. Long-term ambient concentrations of particulates and oxidants and development of chronic disease in a cohort of nonsmoking California residents. *Inhal. Toxicol.* 7:19-34.

Abt, 2000. *The Particulate-Related Health Benefits of Reducing Power Plant Emissions*. October 2000. Prepared for EPA by Abt Associates Inc., Bethesda, MD.

Boiteux, M., 2001. Transports: pour un meilleur choix des investissements. Rapport du Groupe présidé par M. Boiteux. La Documentation Française, Paris.

Brode, R.W., and Wang, J.F. 1992. User's Guide for the Industrial Source Complex (ISC2) Dispersion Models Volumes I-III. EPA-450/4-92-008a, EPA-450/4-92-008b, EPA-450/4-92-008c, U.S. Environmental Protection Agency, Research Triangle Park, NC.

COMEAP 2001. Committee on the Medical Effects of Air Pollutants. The quantification of the effects of air pollution on health in the United Kingdom. London: HMSO http://www.advisorybodies.doh.gov.uk/comeap/.

CWS, 1999. Canada-Wide Standards Development Committee for Particulate Matter (PM) and Ozone (1999). Discussion paper on PM and ozone. Canada Wide Standards Scenarios for Consultation. www.ccme.ca/pdfs/pm_ozone/pm_discussion.pdf and Compendium of Benefits Information. 99-08-17.

Dab, W., Medina, S., Quénel, P., Le Moullec, Y., Le Tertre, Y., Thelot, B., Monteil, C., Lameloise, P., Pirard, P., Momas, I., Ferry, R., and Festy, B. 1996. Short term respiratory health effects of ambient air pollution: results of the APHEA project in Paris. *J. Epidemiol. Commun. Health* 50:42-46.

Derwent, R.G., and Nodop, K. 1986. Long-range transport and deposition of acidic nitrogen species in north-west Europe. *Nature* 324:356-358.

Desaigues, B., and Rabl, A. 2001. Pollution tax and other policy instruments: who pays what? *Pollut. Atmos.* Special Issue Dec. 2001, pp. 27-40.

Dockery, D.W., Speizer, F.E., Stram, D.O., Ware, J.H., Spengler, J.D., and Ferries, B.G. 1989. Effects of inhalable particles on respiratory health of children. *Am. Rev. Respir. Dis.* 139:587-594.

Dusseldorp, A., Kruize, H., Brunekreef, B., Hofschreuder, P., de Meer, G., and van Oudvorst, A.B. 1995. Associations of PM_{10} and airborne iron with respiratory health of adults near a steel factory. *Am. J. Respir. Crit. Care Med*. 152:1932-1939.

ExternE 1995. *ExternE: Externalities of Energy*. ISBN 92-827-5210-0. Vol.1: Summary (EUR 16520); Vol.2: Methodology (EUR 16521); Vol.3: Coal and Lignite (EUR 16522); Vol.4: Oil and Gas (EUR 16523); Vol.5: Nuclear (EUR 16524); Vol.6: Wind and Hydro Fuel Cycles (EUR 16525). Published by European Commission, Directorate-General XII, Science Research and Development. Office for Official Publications of the European Communities, Luxembourg.

ExternE 1998. *ExternE: Externalities of Energy*. Vol.7: Methodology 1998 Update (EUR 19083); Vol.8: Global Warming (EUR 18836); Vol.9: Fuel Cycles for Emerging and End-Use Technologies, Transport and Waste (EUR 18887); Vol.10: National Implementation (EUR 18528). Published by European Commission, Directorate-General XII, Science Research and Development. Office for Official Publications of the European Communities, L-2920 Luxembourg. Results are also available at http://ExternE.jrc.es/publica.html.

ExternE 2000. External Costs of Energy Conversion – Improvement of the ExternE Methodology and Assessment of Energy-Related Transport Externalities. Final Report for Contract JOS3-CT97-0015, EC Directorate General for Research. P. Bickel et al., IER, Universität Stuttgart. Published as *Environmental external costs of Transport*, Friedrich, R. & P. Bickel (eds). Springer Verlag, Berlin, Heidelberg 2001.

Gulis, G. 2000. Life expectancy as an indicator of environmental health. *Eur. J. Epidemiol.* 16:161-165.

Hicks, J.R. 1939. The foundation of welfare economics. *Economic J.* 49:696-712.

Holland, M. 2001. Applications of the ExternE methodology. *Poll. Atmos.* Special Issue Dec. 2001, pp. 69-75.

Hurley, J.F. 2000. DH report.

Keeney, R.L. 1994. Mortality risks induced by the costs of regulations. *J. Risk Uncert.* 8:95-110.

Krewitt, W., Trukenmueller, A., Mayerhofer, P., and Friedrich, R. 1995. *ECOSENSE - an Integrated Tool for Environmental Impact Analysis*. In H. Kremers, and W. Pillmann (Eds.) Space and Time in Environmental Information Systems. Umwelt-Informatik aktuell, Band 7. Metropolis-Verlag: Marburg.

Kuenzli, N., Kaiser, R., Medina, S., Studnicka, M., Chanel, O., Herry, M., Horak, F., Puybonnieux-Texier, V., Quénel, P., Schneider, J., Seetaler, R., Vergnaud, J.-C., and Sommer, H. 2000. Public health impact of outdoor and traffic-related air pollution: a European assessment. *Lancet* 356:795-801.

Leksell, I., and Rabl, A. 2001. Air pollution and mortality: quantification and valuation of years of life lost. *Risk Anal.* 21:843-857.

Massé, E. 2003. Analyse économique de la rentabilite des filtres à particules sur les véhicules diesels neufs. Série Etudes N° 03-E08. Ministère de l'Ecologie et du Développement Durable, Paris.

Mechler, R., Amann, M., and Schöpp, W. 2002. A methodology to estimate changes in statistical life expectancy due to the control of particulate matter air pollution. Interim Report IR-02-035, May 2002.

Mitchell, R.C., and Carson, R.T. 1989. *Using Surveys to Value Public Goods: the Contingent Valuation Method.* Resources for the Future: Washington, DC.

Nathwani, J.S., and Narveson, J. 1995. Three principles for managing risk in the public interest. *Risk Anal.* 15:615-626.

Nathwani, J.S., Lind, N.C., and Pandey, M.D. 1997. Affordable Safety by Choice: The Life Quality Method. Institute for Risk Research, University of Waterloo, Waterloo, Canada, pp. 230.

ORNL/RFF 1994. *External Costs and Benefits of Fuel Cycles.* Prepared by Oak Ridge National Laboratory and Resources for the Future. Edited by Russell Lee, Oak Ridge National Laboratory, Oak Ridge, TN.

Ostro, B.D. 1987. Air pollution and morbidity revisited: a specification test. *J. Environ. Econ. Manage*. 14:87-98.

Pandey, M.D., and Nathwani, J.S. 2003. Canada wide standard for particulate matter and ozone: cost-benefit analysis using a life-quality index. *Risk Anal.* 23:55-67. See also Pandey, M.D., and Nathwani, J.S. 2003. A conceptual approach to the estimation of societal willingness-to-pay for nuclear safety programs. *Int. J. Nucl. Eng. Design* 224:65-77.

Pope, C.A., and Dockery, D.W. 1992. Acute health effects of PM_{10} pollution on symptomatic and asymptomatic children. *Am. Rev. Respir. Dis.* 1451123-1126.

Pope, C.A., Thun, M.J., Namboodri, M.M., Dockery, D.W., Evans, J.S., Speizer, F.E., and Heath. C.W. 1995. Particulate air pollution as a predictor of mortality in a prospective study of US adults. *Am. J. Resp. Crit. Care Med*. 151:669-674.

Rabl, A., and Spadaro, J.V. 1999. Environmental damages and costs: an analysis of uncertainties. *Environ. Intern*. 25:29-46.

Rabl, A. 2000. Criteria for limits on the emission of dust from cement kilns that burn waste as fuel. ARMINES/Ecole des Mines de Paris, Paris. March 2000. 10 pp.

Rabl, A. 2003. Interpretation of air pollution mortality: number of deaths or years of life lost? *J. Air Waste Manage. Assoc.* 53:41-50.

Rabl, A., Spadaro, J.V., and Desaigues, B. 1998a. Nouvelles réglementations pour les incinérateurs de déchets: Une Analyse Coût-Bénéfice (New regulations for waste incinerators: a cost-benefit analysis). *Environ. Tech. Info-Déchets* 175:17-21.

Rabl, A., Spadaro, J.V., and Desaigues, B. 1998b. Analyse coût-bénéfice des politiques de la lutte contre la pollution: validation d'outils à partir du cas de la pollution particulaire de l'air en IdF. Report for INSERM, Centre d'Energétique, Ecole des Mines.

Rabl, A., Spadaro, J.V. and McGavran, P.D. 1998c. Health risks of air pollution from incinerators: a perspective. *Waste Manage. Res.* 16:365-388.

Ramsberg, J.A.L., and Sjöberg, L. 1997. The cost-effectiveness of life saving measures in Sweden. *Risk Anal.* 17:467-487.

Roemer, W., Hoek, G., and Brunekreef, B. 1993. Effect of ambient winter air pollution on respiratory health of children with chronic respiratory symptoms. *Am. Rev. Respir. Dis*. 147:118-124.

Rowe, R.D., Lang, C.M., Chestnut, L.G., Latimer, D., Rae, D., Bernow, S.M., and White, D. 1995. *The New York Electricity Externality Study.* Oceana Publications, Dobbs Ferry: New York.

Schwartz, J., and Morris, R. 1995. Air pollution and hospital admissions for cardiovascular disease in Detroit, MI. *Am. J. Epidemiol*. 142:23-35.

Simpson, D. 1992. Long period modelling of photochemical oxidants in Europe. Calculations for July 1985, *Atmos. Environ*. 26A:1609-1634.

Simpson, D., and Eliassen, A. 1997. Control strategies for ozone and acid deposition – an iterative approach, Norwegian Meteorological Institute, EMEP MSC-W Note 5/97.

Tengs, T.O., Adams, M.E., Pliskin, J.S., Safran, D.G., Siegel, J.E., Weinstein, M.C., and Graham, J.D. 1995. Five hundred life-saving interventions and their cost-effectiveness. *Risk Anal.* 15:369-390.

Trukenmüller, A., and Friedrich, R. 1995. Die Abbildung der großräumigen Verteilung, chemischen Umwandlung und Deposition von Luftschadstoffen mit dem Trajektorienmodell WTM, in Jahresbericht ALS 1995, Stuttgart, p. 93-108.

United Nations Development Program 1990. *The Human Development Report.* Oxford University Press: England.

USDOE 2003. Valuing Externalities Workshop, February 20-21 in McLean, VA. Proceedings at www.netl.doe.gov.

Usher, D. 1973. An imputation to the measure of economic growth for changes in life expectancy. In National Accounting and Economic Theory: The Collected Papers of Dan Usher – Vol. 1, pp.105-144. Edward Elgar Publishing Ltd.: U.K. 1994.

Vossiniotis, G., Arabatzis, G., and Assimacopoulos, D. 1996. Description of ROADPOL: A Gaussian Dispersion Model for Line Sources, program manual, National Technical University of Athens, Greece.

Wordley, J., Walters, S., and Ayres, J.G. 1997. Short term variations in hospital admissions and mortality and particulate air pollution. *Occup. Environ. Med*. 54:108-116.

Appendix A. The Methodology of ExternE

Impact Pathway Analysis (IPA)

For the analysis of fuel chains, ExternE has coupled IPA with life cycle assessment (LCA), a tool that has been widely used for environmental analysis. The key idea of LCA is to take into account all the stages in the life cycle of a process or product. This is illustrated in Figure A1 for the example of electricity production. Whether an IPA of a single source or an LCA of an entire cycle is required, depends on the policy decision in question. For finding the optimal limit for the emission of SO_2 from a coal fired power plant, an IPA is sufficient, but the choice between coal and nuclear involves an LCA.

→ real impacts for each stage (site specific)

Goal: evaluate the entire matrix

Steps of impact pathway analysis → **Stage of fuel chain** ↓	Emission	Dispersion	Exposure-response function	Economic valuation
Fuel extraction				
Fuel transport				
Power plant				
Transmission of electricity				
Management of wastes				

Life cycle assessment: first sum over emissions ↓ Σ then → × multiplication by "**potential impact**" indices

Figure A1. Relation between impact pathway analysis and current practice of most LCA, illustrated for the example of electricity production From Rabl and Spadaro (1999).

In principle, the damages and costs for each pollution source in the life cycle should be evaluated by a site-specific IPA. But in practice almost all LCA has taken the shortcut of first summing the emissions over all stages and then multiplying the result by site-independent impact indices. Also, most practitioners of LCA reject the concept of monetary valuation, preferring instead to use non-monetary indicators of "potential impact" that are based on expert judgment.

Dispersion modeling

Since for most air pollutants other than the globally dispersing greenhouse gases, atmospheric dispersion is significant over hundreds to thousands of km both local and regional effects must be taken into account. ExternE has therefore used a combination of local and regional dispersion models. For dispersion over the local range (< 50 km from the source) two gaussian plume models have been used: ISC (Brode and Wang, 1992) for point sources such as power plants, and ROADPOL for lines sources (emissions from transport) (Vossiniotis et al., 1996).

At the regional scale one needs to take into account the chemical reactions that lead to the transformation of primary pollutants (i.e. the pollutants as they are emitted) to secondary pollutants, for example the creation of sulfates from SO_2. Here the Windrose Trajectory Model (WTM) (Trukenmüller and Friedrich, 1995) has been used to estimate the concentration and deposition of acid species. WTM is a user-configurable Lagrangian trajectory model, derived from the Harwell Trajectory model (Derwent and

Nodop, 1986); it differentiates between 24 sectors of the wind rose, such that from each sector a straight-line trajectory arrives at the receptor point. Concentrations at the receptor point are obtained by averaging over the results from these trajectories, suitably weighted by the winds in each 15° sector.

The creation of ozone has been modeled with the Source-Receptor Ozone Model (SROM) which is based on source-receptor (S-R) relationships from the EMEP MSC-W oxidant model for five years of meteorology (Simpson et al., 1992). Input to SROM are national annual NO_x and anthropogenic NMVOC emissions data from 37 European countries, while output is calculated for individual EMEP 150x150 km2 grid squares by employing country-to-grid square matrices. To account for the non-linear nature of ozone creation, SROM utilises an interpolation procedure allowing S-R relationships to vary depending upon the emission level of the country concerned (Simpson and Eliassen, 1997).

The ECOSENSE model (Krewitt et al., 1995), an integrated impact assessment model developed within ExternE, combines the results from the atmospheric dispersion models and the databases covering receptor data (population, land use, agricultural production, buildings and materials, etc.), meteorological data and emission data for the whole of Europe. Together with dose-response functions and monetary values stored in EcoSense, physical impacts and resulting (marginal) damage costs have been calculated within a consistent modeling framework, taking into account the information on receptor distribution. Impacts due to a point or line emission source are taken into account on a European scale, i.e. the dispersion of pollutants and related impacts are followed up throughout Europe.

Several validation tests have also been carried out to confirm the accuracy of the results. For example, the consistency between ISC and ROADPOL has been checked, and the concentrations predicted by WTM have been compared with measured data and with calculations of the EMEP program, the official program for the modeling of acid rain in Europe.

Health Impacts

The concentration-response (CR) functions for health used by ExternE are assumed to be linear (without threshold). Note that for the calculation of incremental damage costs there is no difference between the linear and the hockey stick function (with the same slope), if the background concentration is everywhere above this threshold; only the slope matters. For the classical air pollutants (particles, NO_x, SO_2, O_3, CO) there is some evidence of linearity down to levels as low as the background levels in most industrialized countries; the precise form of the CR function at extremely low doses is irrelevant for these pollutants.

In ExternE the working hypothesis has been to use the CR functions for particles and for O_3 as basis. Effects of NO_x and SO_2 are assumed to be subsidiary. The principal effects of NO_x and SO_2 arise indirectly from the particulate nature of nitrate and sulfate aerosols (NO_x is also a precursor for ozone) and they are calculated by applying the particle CR functions to these secondary aerosol concentrations. With this assumption the impacts of NO_x and SO_2 per kg of pollutant are roughly comparable to PM_{10}. But the uncertainties are large because there is insufficient evidence for the health impacts of the individual components or characteristics (acidity, solubility, …) of particulate air pollution. In particular there is a lack of epidemiological studies of nitrate aerosols because until recently this pollutant has not been monitored by air pollution monitoring stations. A summary of the most important CR functions for PM is shown in Table A1, together with the monetary values.

The exact functions being used are now old and in some respects out-of-date; we have plans to revise them in the next couple of years. However, the impact pathways being quantified remain generally valid – there are some new pathways, such as particles and infant mortality – and for existing pathways, newer evidence will improve but not markedly change the results. More evidence and improved understanding will, however, have reduced somewhat the uncertainties associated with the ExternE quantifications.

Table A1. C-R functions and costs for PM_{10}, as adapted and recommended by ExternE (1998). The exposure response slope, f_{CR}, has been expressed in units of cases/(person·yr·µg/m^3), relative to average population (thus it includes the fraction of the population that is affected).

End point for PM_{10} and reference	**f_{CR}** cases per (pers·yr·µg/m^3)	**Cost per case** €/case	**Cost per person** € per (pers·yr·µg/m^3)	**%** of PM_{10} cost
Chronic Mortality YOLL (Pope et al., 1995)	4.10E-04	84330	3.46E+01	85.0%
CB, Adults (Abbey et al., 1995)	3.92E-05	105000	4.12E+00	10.1%
RAD, Adults (Ostro, 1987)	2.00E-02	75	1.50E+00	3.7%
Bronchodilator usage, Asthmatic adults (Dusseldorp et al., 1995)	4.56E-03	37	1.69E-01	0.4%
Chronic cough, children (Dockery et al., 1989)	4.14E-04	225	9.32E-02	0.2%
CB, children (Dockery et al., 1989)	3.22E-04	225	7.25E-02	0.2%
HA, Cerebrovascular (Wordley et al., 1997)	5.04E-06	7870	3.97E-02	0.1%
Cough, Asthmatic adults (Dusseldorp et al., 1995)	4.69E-03	7	3.28E-02	0.1%
Congestive heart failure, Asthmatic 65+ (Schwartz and Morris, 1995)	2.59E-06	7870	2.04E-02	0.1%
Bronchodilator usage, Asthmatic children (Roemer et al. 1993)	5.43E-04	37	2.01E-02	0.0%
HA, Respiratory (Dab et al., 1996)	2.07E-06	7870	1.63E-02	0.0%
LRS, Asthmatic adults (Dusseldorp et al., 1995)	1.70E-03	7.5	1.28E-02	0.0%
Cough, Asthmatic children (Pope and Dockery, 1992)	9.34E-04	7	6.54E-03	0.0%
LRS, Asthmatic children (Roemer et al., 1993)	7.20E-04	7.5	5.40E-03	0.0%
Total PM_{10}			**4.07E+01**	100.0%

HA = hospital admission; CB = chronic bronchitis; LRS = lower respiratory symptoms;
RAD = restricted activity day; YOLL = years of life lost.
To derive f_{CR} from the data in the references (given e.g. as % increase per receptor), we have assumed:
3.5% of population is asthmatic, children are 20% of population, 14% of population is over 65.
For chronic mortality f_{CR} has been obtained by integration over life tables (ExternE 1998) or the Gompertz function for age-specific mortality (Leksell and Rabl, 2001), assuming that it applies only to the population over 30 years (= cohorts in Pope et al. (1995)).

Appendix B. Values of a Life Year Implicit in Public Decisions

Data on the costs of risk-reducing measures in the USA have been collected in an interesting study by the Harvard Center for Risk Analysis (Tengs et al., 1995). More than 500 life saving interventions were identified and the implied value of a YOLL was determined. The results show that there is an enormous range of cost/YOLL values, spanning over 11 orders of magnitude; such variations between the costs of different interventions were found in almost every category. The cost-effectiveness varies between different sectors of society, as can be seen from Table B1 which summarizes the median of the cost/YOLL estimates. The median cost is especially high in the environmental domain, $4,200,000, far higher than the median of $19,000 in the health care sector.

Table B1. Median of cost/YOLL estimates as a function of sector of society and type of intervention in the USA. Adapted from Tengs et al. (1995).

Sector of society	Type of intervention			
	Medicine	Fatal injury reduction	Toxin control	All
Health care	$19,000	na	na	$19,000
Residential	na	$36,000	na	$36,000
Transportation	na	$56,000	na	$56,000
Occupational	na	$68,000	$1,400,000	$350,000
Environmental	na	na	$4,200,000	$4,200,000
All	$19,000	$48,000	$2,800,000	$42,000

na = not applicable

An analogous study, following the same methodology and analyzing over 150 interventions, was carried out in Sweden by Ramsberg and Sjöberg (1997). Results are summarized in Table B2. Most of these interventions are implemented, and practically all have other objectives in addition to saving lives. The authors compare their results with those of Tengs et al., see Table B3. For medicine and fatal injury reduction the costs are comparable (approximately $20,000 in Sweden and $40,000 in the USA), but for toxin control the median cost is two orders of magnitude higher in the USA than in Sweden.

Table B2. Results for mean and median of cost/YOLL estimates in Sweden, in 1993$. Adapted from Ramsberg and Sjöberg (1997).

Category	n	Mean	Median
Medicine	101	$1,240,000	$14,000
Radiation	13	$30,000	$1,400
Road safety	32	$242,000	$66,500
Life style risks	3	$470	$340
Fire protection	7	$211,000	$15,000
Electrical safety	2	$1,245,000	$1,245,000
Accidents	1	$280,000	$280,000
Environmental pollutants	5	$235,000	$235,000
Crime	1	$15,000	$15,000

Table B3. Comparison of median cost/YOLL between Sweden and USA. Adapted from Ramsberg and Sjöberg (1997).

	Median cost/YOLS	
	Sweden	USA (Tengs et al. 1995)
Medicine	$13,800	$19,000
Toxin control	$19,600	$2,800,000
Fatal injury reduction	$69,000	$48,000
All	$19,500	$42,000

For the purpose of determining a value that could be recommended as guideline for environmental policy, it is not the median in these tables that is relevant but rather the upper range of values for interventions that are actually implemented. Data for the USA are difficult to interpret in this sense because the range is very large, covering many orders of magnitude. For Sweden Ramsberg and Sjöberg say that most of the interventions they consider are implemented, but even here the range is so wide that it is difficult to extract a recommendation. In any case it seems that the value of 84,000 €/YOLL chosen by ExternE (1998) is compatible with the numbers in Tables B1 to B3.

Appendix C. Derivation of Life Quality Index

The general idea is that a person's enjoyment of life, or utility in an economic sense arises from a continuous stream of resources available for consumption over the entire life. Therefore income required to support consumption and the time to enjoy are two determinants of the life quality. For a person at age *a*, the lifetime utility can therefore be interpreted as total consumption incurred over the remaining lifetime, which is a random variable.

The mathematical derivation is briefly described here and details can be found elsewhere (Pandey and Nathwani, 2003). Denote the consumption rate at some age τ as $c(\tau)$ ($/year), and assume that a valid function, $u[c(\tau)]$, exists that can quantify the utility derived from consumption. The probability of survival in the period *a* to *t* is denoted by $S(a, t)$. The present value of life-time utility for a person is equivalent to integration of $u[c(\tau)]$ from the present age *a* till a terminal age *T* with a suitable discount rate to reflect the fact that individuals tend to undervalue a prospect of future consumption in comparison to that of present. Thus,

$$L(a) = \frac{1}{S(0,a)} \int_a^T S(a,t)u[c(t)]e^{-r(t-a)}dt \qquad \text{(A1)}$$

where *r* denotes the rate of time preference for consumption. Assuming a power utility function and constant consumption rate, i.e., $c(t) = c$, and $u(c) = c^q$, eqn.A1 can be written in a compact form as

$$L(a) = u(c)e(a) = c^q\, e(a) \qquad \text{(A2)}$$

The life-time utility, $L(a)$, is a surrogate measure of quality-of-life of a person of age *a*. This type of reasoning primarily originates from the fundamental work of Usher (1973) on the impact of historical improvement of LE on economic growth.

The life-quality at the societal level is an aggregate of the values for all individuals in the society. To achieve this, *L(a)*, should now be integrated over the distributions of population age and consumption rate. As a matter of simplification, we assume that the consumption rate is equivalent to the real gross domestic product per person per year (*G*), a valid measure of average consumption in society. Integrating $L(a)$ over the population age-distribution, $f(a)$, leads to

$$LQI = \int_0^T L(a)\, f(a)da = c^q \int_0^T e(a)\, f(a)da = G^q E \qquad \text{(A3)}$$

where *E* denotes the discounted life expectancy averaged over the age-distribution of the population. The exponent *q* can be shown to equal the ratio $q = w/(1-w)$ where *w* is the average fraction of time spent on work in a country for producing *G* (11). For industrialized nations a typical value of *q* is 0.15.

The societal life-quality function, LQI, is a utility function as well as a composite social indicator, since it consists of two important indicators of development, namely GDP per capita and life expectancy. By setting E equal to LE at birth and ignoring the discounting, LQI was used to rank the level of national development (Nathwani et al., 1997) similar to Human Development Index proposed by the United Nations Development Program (UN, 1990).

ANALYSIS OF $PM_{2.5}$ USING THE ENVIRONMENTAL BENEFITS MAPPING AND ANALYSIS PROGRAM (BENMAP)

Kenneth Davidson,[1] Aaron Hallberg, Donald McCubbin,[2] Bryan Hubbell[1]

ABSTRACT

As epidemiological work from around the world continues to tie $PM_{2.5}$ to serious adverse health effects, including premature mortality. The U.S. Environmental Protection Agency (U.S. EPA) has developed a number of policies to reduce air pollution, including $PM_{2.5}$. To assist in the benefit-cost analyses of these air pollution control policies, the U.S. EPA has developed the Environmental Benefits Mapping and Analysis Program (BenMAP). BenMAP is meant to 1) provide a flexible tool for systematically analyzing impacts of changes in environmental quality in a timely fashion, 2) ensure that stakeholders can understand the assumptions underlying the analysis, and 3) adequately address uncertainty and variability. BenMAP uses a "damage-function" approach to estimate the health benefits of a change in air quality. The major components of the damage-function approach are population estimates, population exposure, adverse health effects, and economic costs. To demonstrate BenMAP's ability to analyze $PM_{2.5}$ pollution control policy scenarios, we assess 2 sample applications: 1) benefits of a national-level air quality control program, and 2) benefits of attaining 2 annual $PM_{2.5}$ standards in California (annual average standards of 15 $\mu g/m^3$ and 12 $\mu g/m^3$). In the former, we estimate a scenario where control of $PM_{2.5}$ emissions result in $100 billion of benefits annually. In the analysis of alternative standards, we estimate that attaining the more stringent standard (12 $\mu g/m^3$) would result in approximately 2000 fewer premature deaths each year than the 15 $\mu g/m^3$ achieves. BenMAP has a number of features to help clarify the analysis process. It allows the user to record in a configuration all of the choices made during an analysis. Configurations are especially useful for recreating already existing policy analyses. Also, BenMAP has a number of reporting options, including a set of mapping tools that allows users to visually inspect their inputs and results.

[1] U.S. EPA, Washington, DC 20460
[2] Abt Associates Inc., Bethesda, MD 20814
Disclaimer: The opinions expressed in this article are the authors' and do not necessarily represent those of the United States Environmental Protection Agency.

INTRODUCTION

A wide range of analyses have demonstrated that air pollution is related to many problems, including adverse health effects, agricultural crop losses, visibility reductions, and damage to buildings. In recent years, the health problems associated with particulate matter less than 2.5 microns in aerodynamic diameter ($PM_{2.5}$) have received increasing attention, as epidemiological work from around the world has tied $PM_{2.5}$ to serious adverse health effects, including premature mortality, chronic bronchitis, and hospital admissions. Impact assessments in a variety of settings have consistently identified $PM_{2.5}$ as a major cause of premature mortality and morbidity around the world (e.g., Kunzli et al., 2000). To reduce air pollution, including $PM_{2.5}$, in the United States, the U.S. Environmental Protection Agency (U.S. EPA) has developed a number of approaches, including tighter emission standards on motor vehicles and at industrial sources, reduced sulfur content in diesel fuels, an emissions cap and trading program for sulfur dioxide, and a recent proposal to expand trading to include nitrogen oxides.

Benefit-cost analysis has become an important tool in analyzing air policies. The U.S. Office of Management and Budget (OMB, 2003) recently released a report finding that the benefits of clean-air regulations during the past decade outweighed the costs to industry by at least a factor of five. As air quality standards become more stringent, however, the costs of achieving incremental improvements rise, making it more desirable to weigh the benefits and costs of new policies. Currently, the process of analyzing benefits and costs has a number of weaknesses. One is the time that it takes between proposing a new policy and analyzing its benefits and costs. Another is the black box nature of the analysis, and the difficulty for those outside the analysis process to understand the assumptions underlying it. Yet another is the inherent uncertainty and variability in the approaches used in an analysis. In response, the U.S. EPA is developing the Environmental Benefits Mapping and Analysis Program (BenMAP). The goals are 1) to provide a flexible tool for systematically analyzing the impacts of changes in environmental quality in a timely fashion, 2) to make sure that stakeholders can understand the assumptions underlying the analysis, and 3) to adequately address uncertainty and variability.

A new Windows-based program developed jointly by the U.S. EPA and Abt Associates Inc., BenMAP allows users to estimate the benefits associated with changes in environmental quality. Ambient air quality surfaces can be created from monitored air quality, modeled air quality, or a combination of the two.[3] Potentially exposed populations can be calculated for any year after 1990, using 1990 and 2000 census data along with county-level projections out to 2025 (and simple linear interpolation thereafter). Populations can be further broken down by race, gender, and age. Additionally, BenMAP includes large databases of concentration-response functions and economic valuations of health impacts, and allows users to add new functions and valuations.

BenMAP also allows users to save the details of a particular analysis so that it can be replicated using alternative environmental quality scenarios. This allows users to analyze multiple policies in a systematic fashion, ensuring that the results of each analysis can be compared directly with each other. This ability to create a consistent framework for multiple analyses means that the assumptions underlying the analyses can remain the same over time, making the benefits estimation process more transparent. Additionally, BenMAP stores each decision that a user makes at every step of the analysis so that an audit trail can be generated from all BenMAP output files. This provides a further degree of transparency to analyses.

Finally, BenMAP characterizes uncertainty arising from two sources: the health impact coefficient from the epidemiological study (usually represented by the standard error from the epidemiological study), and the dollar value assigned to each health effect. For each health endpoint, BenMAP generates distributions of incidence estimates, represented by percentiles, and then uses Monte Carlo techniques to sample from the distributions of incidence and unit values to derive distributions of economic values for each health endpoint. The distribution of total economic benefits can then be estimated by Monte Carlo sampling from each of the individual distributions of

[3] BenMAP can use output from a variety of models including: Regulatory Model System for Aerosols and Deposition (REMSAD), the Comprehensive Air Quality Model with Extensions (CAMx), the Urban Airshed Monitoring - Variable grid model (UAM-V), and the Community Multi-Scale Air Quality model (CMAQ). The following links provide more information on each model: REMSAD: http://remsad.saintl.com/; CAMx: http://www.camx.com; UAM-V: http://uamv.saintl.com/; and CMAQ: http://www.epa.gov/asmdnerl/models3/cmaq.html.

economic values across endpoints. It should be noted that the uncertainty BenMAP quantifies provides insight into how uncertain incidence and valuation estimates are with regard to uncertainty associated with statistical error and cross-study variability. However, BenMAP at this time is unable to provide a probabilistic, multiple-source uncertainty analysis. Key sources of uncertainty not captured by BenMAP include those associated with emissions estimates, air quality modeling, population projections, and aspects of health science and economic valuation not captured in the studies.

This paper first presents the basic framework for calculating the benefits of policy-related changes in air quality. We limit the discussion to estimating and valuing the health impacts directly linked to ambient levels of $PM_{2.5}$, although BenMAP is also able to estimate and value the health effects associated with other pollutants. We then present the results of two sample analyses. In the first case, we estimate the health-related economic benefits of the attainment of a national-level emissions control program. We then present the health-related economic benefits of California meeting the national annual standard for $PM_{2.5}$, and the incremental benefits of California meeting its own lower annual $PM_{2.5}$ standard.

METHODS

BenMAP uses a "damage-function" approach to estimate the health benefits associated with a change in air quality. This approach estimates changes in individual health endpoints, assigns values to these changes, and sums the values for all non-overlapping health endpoints to generate total benefits. It imposes no overall preference structure, and does not account for potential income or substitution effects, so that adding a new endpoint will not reduce the value of changes in other endpoints (Smith et al., 2002). This is the standard approach for most cost-benefit analyses of regulations affecting environmental quality, and it has been used in several recent published analyses (Ostro and Chestnut, 1998; Kunzli et al., 2000; Levy et al., 2001).

Figure 1 illustrates the major steps in the damage function approach: population estimate, population exposure, adverse health effects, and economic costs. BenMAP is built on block-level population data from the 1990 and 2000 U.S. Census and county-level population forecasts out to the year 2025. BenMAP maps the population data to air quality surfaces, which may be generated using either monitor-based air quality data, model-based air quality data, or a combination of the two.

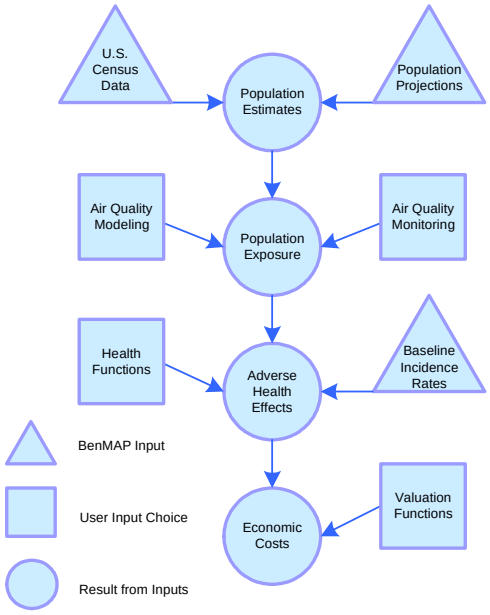


Figure 1. BenMAP policy analysis approach.

Typically, a user will create both a baseline population exposure surface, reflecting current or future-current air quality conditions, and a control surface, reflecting air quality conditions after the implementation of a policy scenario or a rollback of air quality conditions to a particular level, such as the attainment of a standard. The baseline and control population exposure surfaces yield changes in population exposure to ambient air pollution that are then input to health impact functions to generate changes in incidence of health effects. The resulting effects changes are then assigned monetary values. Finally, values for individual health effects are summed to obtain an estimate of the total monetary value of the changes in emissions.

Population

BenMAP population data is built from Census block data containing over two hundred race, gender, and age specific variables. This block level data is aggregated to the various grid definitions used in BenMAP analyses, including uniform grids such as that used by the REMSAD air quality model, and political boundary grids such as United States counties. BenMAP also includes population projections developed by Woods & Poole at the United States county level for each of these race, gender, and age specific variables. BenMAP uses these projections to estimate populations for the years 2001 to 2025 by using the county level ratios of the desired population projection and the year 2000 population. For years beyond 2025, BenMAP uses a simple linear extrapolation of the estimated 2024-2025 trend.

For United States county population data, estimating populations for 2001 and beyond is straightforward. Each race, gender, and age specific variable is scaled by the appropriate population projection ratio in each county. For grid definitions other than United States counties, BenMAP calculates a population-weighted average of the county level population projection ratios for each grid cell. That is, a grid cell might overlap two or more counties. In these cases, BenMAP uses the percentage of the total population in that grid cell which comes from each county to determine a population weighted average of the various county level population projections.[4]

Population Exposure

BenMAP provides multiple ways to estimate exposure, using a combination of air quality monitoring and air quality modeling data. Currently, it does not include emissions data or the ability to do air quality modeling, and instead has a database of air quality monitoring data and the ability to incorporate modeling data from several different air quality models. BenMAP has four broad approaches to estimating exposure: model direct, monitor direct, monitor and model relative, and monitor rollbacks. The air quality exposure estimates are referred to as air quality grids.

BenMAP creates air quality grids to estimate the average exposure to ambient air pollution of people living in some specified area, or domain, such as that delineated by REMSAD, CAMx, UAM-V and CMAQ models, as well as more irregular shapes, such as counties. It is assumed that all persons in a given grid-cell are exposed to the same pollution levels. When using modeling data directly, BenMAP simply converts the input modeling data into an air quality grid file that matches the structure of the model. These model specifications are presented in Table 1. However, when using monitor data, alone or in combination with modeling data, the user may select from two interpolation methods to move from point-based monitor data to the grid-cell-based exposure estimates: Closest Monitor and Voronoi Neighbor Averaging (VNA).

The Closest Monitor method simply assigns the monitor closest to a grid cell's center as its representative value. VNA first identifies the set of monitors that "surround" each grid cell's center (these monitors are referred to as the grid cell's neighbors), and then calculates an inverse-distance weighted average of these neighboring monitors. For each of the interpolation approaches, users specify parameters such as the maximum distance within which to include a monitor, with monitors beyond the specified maximum excluded from the analysis.

[4] The appendices in the BenMAP user's manual describe the population data, forecasting approach, and other aspects of the model in detail. The manual is available at http://www.epa.gov/ttn/ecas/models/modeldoc.pdf.

Table 1: Air quality model data structure.

Model	Pollutant [a]	Modeling Domain and Data File Description
REMSAD	$PM_{2.5}$, PM_{10}, and PM Coarse	Two REMSAD Modeling Domains REMSAD12: has grid cells that are 1/6 of a degree longitude wide and 1/9 of a degree latitude high, or about 12 kilometers by 12 kilometers. The modeling domain extends from longitude -126E to -66E and latitude 24E to 52E, with a total of 90,720 grid cells that completely cover the continental United States. REMSAD36: has grid cells that are 1/2 of a degree longitude wide and 1/3 of a degree latitude high, or about 36 kilometers by 36 kilometers. The modeling domain extends from longitude -126E to -66E and latitude 24E to 52E, with a total of 10,080 grid cells that completely cover the continental U.S.
CAMx and UAM-V	Ozone	Modeling Domain CAMx and UAM-V have grid cells that are 1/6 of a degree longitude wide and 1/9 of a degree latitude high, or about 12 kilometers by 12 kilometers. BenMAP assumes a boundary extending from longitude -127E to -67E and latitude 26E to 52E, with a total number of 84,280 grid cells that cover most of the continental United States, with the exception of the southern tips of Florida and Texas.
CMAQ	$PM_{2.5}$, PM_{10}, and PMC	Modeling Domain The modeling domain for CMAQ covers the entire continental United States. The size of each grid cell is roughly comparable to that of REMSAD36.

[a] Note that the different Grid types are limited to specific pollutants. Currently, BenMAP can only input REMSAD and CMAQ model data for particulate matter, and CAMx and UAM-V are limited to ozone.

Users may also conduct monitor rollback analyses, for one or more non-overlapping rollback regions. A region is simply a set of states with an associated set of rollback parameter values. Three rollback types are available - Percentage Rollback, Incremental Rollback, and Rollback to a Standard. Each of these rollback types has different rollback parameters associated with it.

Percentage Rollback. Percentage rollback involves setting only two parameters - a percentage and a background pollution level. The rollback procedure is similarly straightforward - each observation at each monitor in the region has the portion of its value which is above the background pollution level reduced by a percentage.

Incremental Rollback. Incremental Rollback similarly involves setting only two parameters - an increment and a background level. The rollback procedure is similar to the percentage rollback procedure - each observation at each monitor in the region has the portion of its value which is above background level reduced by an increment. The reduced values are not allowed to become negative, however - that is, they are truncated at zero.

Rollback to a Standard. Rollback to a Standard has two groups of parameters - those associated with the Attainment Test, which determines whether a monitor is in attainment (meets the standard), and those associated with the Rollback Methods, which are used to bring out of attainment monitors into attainment.

The Attainment Test parameters are Metric, Ordinality, and Standard. A monitor is considered in attainment if the n^{th} highest value of the metric is at or below the value specified by the standard, where n is the ordinal value. For example, if the $PM_{2.5}$ metric is twenty four hour average, the ordinality is four, and the standard is sixty five $\mu g/m^3$, a monitor will be considered in attainment if the fourth highest twenty four hour average is at or below sixty five $\mu g/m^3$. Ordinality does not apply for the annual average metric, since there is only a single metric value to work with.

For $PM_{2.5}$, the Rollback Method parameters are simply a Rollback Method and a Background Level. There are four supported rollback methods for $PM_{2.5}$ Rollbacks - Percentage, Incremental, Peak Shaving, and Quadratic. For each of these rollback methods, the following definitions are important:

Anthropogenic Out of Attainment Value: The out of attainment value is the metric value that caused the monitor to be considered out of attainment (the fourth highest value, in the above example). The anthropogenic out of attainment value is the portion of the out of attainment value left over after the background level has been subtracted from it.
Anthropogenic Standard: The portion of the Attainment Test Standard left over after the background level has been subtracted from it.

Anthropogenic Metric Values: The portion of each metric value left over after the background level has been subtracted from it (or zero, if the metric value is below the background level).

Non-Anthropogenic Metric Values: For each metric value, either the background level (if the metric value is higher than the background level), or the metric value (if the metric value is less than the background level).

Percentage. To generate rolled back metric values using Percentage rollback, BenMAP calculates the percentage required to reduce the anthropogenic out of attainment value to exactly the anthropogenic standard. This percentage reduction is then applied to all of the anthropogenic metric values. Finally, these reduced anthropogenic metric values are added to the non-anthropogenic metric values to give the final rolled back metric values.
Incremental. To generate rolled back metric values using Incremental rollback, BenMAP calculates the increment required to reduce the anthropogenic out of attainment value to exactly the anthropogenic standard. This incremental reduction is then applied to all of the anthropogenic metric values (but - they are not allowed to fall below zero). Finally, these reduced anthropogenic metric values are added to the non-anthropogenic metric values to give the final rolled back metric values.
Peak Shaving. To generate rolled back metric values using Peak Shaving rollback, BenMAP simply truncates all anthropogenic metric values at the anthropogenic standard. These reduced anthropogenic metric values are added to the non-anthropogenic metric values to give the final rolled back metric values.
Quadratic Rollback. The Quadratic rollback reduces large values proportionally more than small values while just achieving the standard - that is, the anthropogenic out of attainment value should be more or less at the anthropogenic standard after the rollback (though some small amount of error is involved).

Combining Monitoring and Modeling Data

When BenMAP interpolates air quality data using the Monitor and Model Relative option, monitor values are scaled using modeled air quality data. The concept of scaling is to use the modeling data to adjust the interpolated monitor data, and thus combine the advantages of both types of data - the real-world accuracy of the monitoring data and the omnipresence and ability to predict future trends of the modeling data. There are three scaling approaches: Spatial, Temporal, and Spatial and Temporal (combined).[5]

Spatial scaling. Spatial scaling involves an air quality modeling file that matches the same year as the monitoring data. BenMAP scales the concentrations of each neighboring monitor by the ratio of the modeled concentration at the grid cell to the modeled concentration at the grid cell containing the monitor. This approach takes into account what the air quality modeling reveals about spatial heterogeneity in pollution levels. For example, if the monitors are in relatively polluted urban areas, and the grid cell is in a relatively unpolluted rural area, then the scaling will result in multiplying the monitor values with ratios less than one, and thus produce lower values at the rural grid cell than would be estimated with interpolation of the unscaled monitor data.

Spatial scaling is useful because, while monitors provide invaluable information about historical conditions, there are only a limited number of monitors. Many areas, particularly rural areas in the U.S., are not close to monitors. Model data can provide additional information that improves the interpolated concentration estimates, and provides a more accurate picture of air quality.

Temporal scaling. Temporal scaling involves both a base year air and a future year air quality modeling file, and scales the concentrations of each neighboring monitor by the ratio of the modeled concentration at the grid cell

[5] Each scaling approach is documented in detail in Appendix C of the BenMAP user's manual, available at http://www.epa.gov/ttn/ecas/models/modeldoc.pdf.

containing the monitor in the future year to the modeled concentration at the grid cell containing the monitor in the base year. This approach takes into account what the air quality modeling reveals about the changes in pollution levels over time at the monitor sites. For example, if the modeling forecasts that in the future, pollution levels will decrease, then the scaling will result in multiplying the monitor values with a ratio less than one, and thus produce lower forecasts at the grid cell than would result with the unscaled monitor data.

Temporal scaling is useful because monitors cannot provide any information about future conditions. Model data can provide this information, which can then be used to project future monitor concentrations.

Spatial and Temporal Scaling. Using both spatial and temporal scaling involves base-year and future-year air quality modeling files, and is simply a combination of spatial scaling and temporal scaling. BenMAP scales the concentrations of each neighboring monitor first by the ratio of the modeled concentration at the grid cell in the future year to the modeled concentration at the grid cell containing the monitor in the future year (spatial scaling), and then by the ratio of the modeled concentration at the grid cell containing the monitor in the future year to the modeled concentration at the grid cell containing the monitor in the base year (temporal scaling). Notice, however, that the two future year concentrations at the grid cell containing the monitor cancel out, allowing the ratio used to be simply the modeled concentration at the grid cell in the future year to the modeled concentration at the grid cell containing the monitor in the base year.

Adverse Health Effects

BenMAP is able to calculate the adverse health effects related to hundreds of health impact functions as part of the evaluation of the effects of various $PM_{2.5}$ (and other pollutant) air quality scenarios. It also comes with a wide variety of incidence rate data necessary to establish the baseline health conditions prior to calculating a change in health effects, and it allows for complex aggregation and pooling, in order to combine multiple sources of information. Finally it keeps track of all of the assumptions used in an analysis through the use of configurations that may be re-used and edited for new analyses.[6]

To calculate point estimates of the changes in incidence of a given adverse health effect associated with a given set of air quality changes, BenMAP performs a series of calculations at each grid-cell. First, it accesses the health impact functions needed for the analysis, and then it accesses any data needed by the health impact functions. Typically, these include the grid-cell population, the change in population exposure at the grid-cell, and the appropriate baseline incidence rate. It then calculates the change in incidence of adverse health effects for each selected health impact function. The resulting incidence change is stored, and BenMAP proceeds to the next grid-cell, where the above process is repeated.

BenMAP calculates the uncertainty surrounding estimated incidence changes, resulting from the sampling uncertainty surrounding the pollutant coefficients in the health impact functions used, and produce a distribution of possible incidence changes rather than a single point estimate. To do this, BenMAP uses an N-point Latin Hypercube to represent the underlying distribution, and creates a corresponding distribution of incidence changes in each population grid cell, where N is specified by the user (Helton and Davis, 2002).[7]

For pollutant-health endpoint combinations estimated by more than one health impact function, BenMAP can pool the incidence estimates using a variety of techniques, including fixed and random effects and user-specified subjective weights. The fixed effects model assumes that there is a single true concentration-response relationship and therefore a single true incidence estimate that applies everywhere. Differences among incidence estimates

[6] The sources of prevalence and incidence data included in BenMAP are documented in Appendix E of the User's manual (http://www.epa.gov/ttn/ecas/models/modeldoc.pdf), all PM-related concentration response functions packaged with BenMAP are documented in Appendix F, and Appendix I describes in detail the uncertainty and pooling options available to the user.

7 The Latin Hypercube method is used to enhance computer processing efficiency. It is a sampling method that divides a probability distribution into intervals of equal probability, with an assumption value for each interval assigned according to the interval's probability distribution. Compared with convention Monte Carlo sampling, the Latin Hypercube approach is more precise over a fewer number of trials because the distribution is sampled in a more even, consistent manner.

derived from different studies are therefore simply the result of sampling error. The certainty of an estimate is reflected in its variance (the larger the variance, the less certain the estimate). Fixed effects pooling therefore weights each incidence estimate in proportion to the inverse of its variance.

The weighting scheme used in a pooling based on the random effects model is basically the same as that used if a fixed effects model is assumed, but the variances used in the calculations are different. This is because a fixed effects model assumes that the variability among the estimates from different studies is due only to sampling error (i.e., each study is thought of as representing just another sample from the same underlying population), while the random effects model assumes that there is not only sampling error associated with each study, but that there is also between-study variability -- each study is estimating a different underlying concentration-response relationship. Therefore, the sum of the within-study variance and the between-study variance yields an overall variance estimate.[8]

Economic Cost

Once BenMAP has estimated the incidence associated with a particular health effect, derived from a single health impact function or multiple pooled health impact functions, the user may estimate the economic value of that incidence based on hundreds of preloaded health effect-specific dollar values. In the same way BenMAP estimates health effects, it can also estimate both point estimates of incidence valuation and a Latin Hypercube-based distribution of incidence valuation reflecting both the uncertainty surrounding estimated incidence and the uncertainty surrounding the unit values. BenMAP also allows for the pooling of endpoint specific valuation in the same way incidence estimates are pooled.[9]

Reporting

BenMAP has a number of reporting options to allow users to document their analyses. The user can sum across monetized benefits (to create estimates of total benefits associated with a given policy analysis) and can export incidence and valuation results to spreadsheet compatible files. The user has a set of mapping tools to visually inspect inputs and results, and to export maps to shapefile formats for use in other GIS programs. And the user has access to an audit trail that keeps track of all user's decisions at each step of an analysis and to review decisions made in previous analyses.

BenMAP records each of the choices made when estimating the change in adverse health effects between a baseline and control scenario. This is referred to as a configuration. A configuration records the following choices: the air quality grids for the baseline and control scenarios; the year for the analysis; the threshold for the analysis; whether the analysis will focus on a single "point" estimate (Point Mode), or a range of results that mirror the variability in the inputs to the health impact functions (Latin Hypercube Points); the health impact functions to be used in estimating adverse health effects and associated pooling specifications; the unit values to be used in estimating the monetary value associated with the adverse health effects; and the pooling specifications used to estimate health effects and valuations based on more than one health impact function.

Once these choices are made, BenMAP saves the configuration file for future reuse. This is especially useful for recreating already existing policy analyses, such as those conducted by the U.S. EPA.

SAMPLE APPLICATIONS

We consider two sample applications. In the first, we analyze the benefits of a national-level air quality control program, and in the second, we examine the benefits associated with attaining two annual $PM_{2.5}$ standards in California, the national standard of 15 $\mu g/m^3$ and the more stringent California standard of 12 $\mu g/m^3$.

[8] Appendix I of the BenMAP user's manual provides the documentation and algorithms used for all weighting methods available to the user, including Fixed Effects and Random Effects weighting.
[9] Appendix H of the BenMAP user's manual documents the source and derivation of all health effect unit values included in the software (http://www.epa.gov/ttn/ecas/models/modeldoc.pdf).

National-Level Benefits Example

As part of its congressional mandate, the U.S. EPA promulgates regulations to improve the nation's air quality, controlling emissions from both mobile and stationary sources. The Clean Air Act allows the EPA to set emissions standards to protect human health without regard to compliance cost. However, Executive Order 12866 requires federal agencies to estimate the benefits and costs of major new pollution control regulations. As a result, EPA presents the costs and benefits of all economically significant new rules in a Regulatory Impact Analysis.

The form of air quality regulations take many shapes to control criteria pollutants, such as PM and ozone, as well as hazardous air pollutants. The example presented here assumes the enactment of a national control strategy to reduce stationary sources of pollutant emissions related to the formulation of $PM_{2.5}$. We assume the rule would reduce emissions of sulfur dioxide (SO_2), nitrogen oxides (NO_x) from fossil fuel-fired combustion units by approximately 70 percent from current levels. These mandatory emission reductions would be achieved through a cap and trade program. Federally enforceable emissions limits, or national caps, for each pollutant would be established. Sources would be allowed to transfer these authorized emission limits among themselves to achieve the required reductions for all pollutants at the lowest overall cost.

Such an approach would likely provide significant benefits to public health and the environment. Emissions reductions would start before 2010 and would increase significantly between 2010 and 2020. The program would cut SO_2 emissions by 73 percent, from year 2000 emissions of 11 million tons to caps of 4.5 million tons in 2010 and 3 million tons in 2018. It would cut emissions of NO_x by 67 percent, from year 2000 emissions of 5 million tons to caps of 2.1 million tons in 2008 and 1.7 million tons in 2018. Based on these emissions reductions, the cumulative health benefits of the program across the next two decades are likely to be significant.

In order to characterize the health impacts and health-related economic benefits of the emission reductions likely to occur from implementation of a program such as the one above, EPA conducted sophisticated modeling of emissions reductions from electric utilities and the fate and transport of those emissions. EPA used the Integrated Planning Model (IPM) to estimate emissions changes and REMSAD to estimate changes in ambient levels of $PM_{2.5}$. In the following section, we demonstrate how BenMAP can be used to estimate population level exposure to changes in ambient $PM_{2.5}$, changes in incidence of key health effects, and the value of those changes in health.

BenMAP Analysis of a National-Level PM Control Program

To estimate $PM_{2.5}$ exposure for this example, the U.S. EPA used REMSAD modeling data, with a 36 kilometer by 36 kilometer resolution, to spatially and temporally scale 2001 $PM_{2.5}$ monitoring data. We use the same the same data as a starting point for our results.

Recall that to create air quality grids BenMAP scales the concentrations of each neighboring monitor first by the ratio of the modeled concentration at the grid cell in the future year to the modeled concentration at the grid cell containing the monitor in the future year (spatial scaling), and then by the ratio of the modeled concentration at the grid cell containing the monitor in the future year to the modeled concentration at the grid cell containing the monitor in the base year (temporal scaling). Figures 2, 3, and 4 display the 2001 monitoring data and the base year (2001) and future year (2020) modeling data used to create air quality grids for the air quality control program analyzed here. The resulting air quality grid is displayed in Figure 5.

We derived health impact functions using the available published scientific literature to ascertain the relationship between particulate matter exposure and adverse human health effects. In general, we selected health impact functions from epidemiological studies that: 1) used $PM_{2.5}$, 2) covered the broadest potentially exposed population, 3) had appropriate model specification (e.g. controlled for confounding pollutants), 4) had been peer-reviewed, and 5) analyzed health effects to which we could place an economic value.

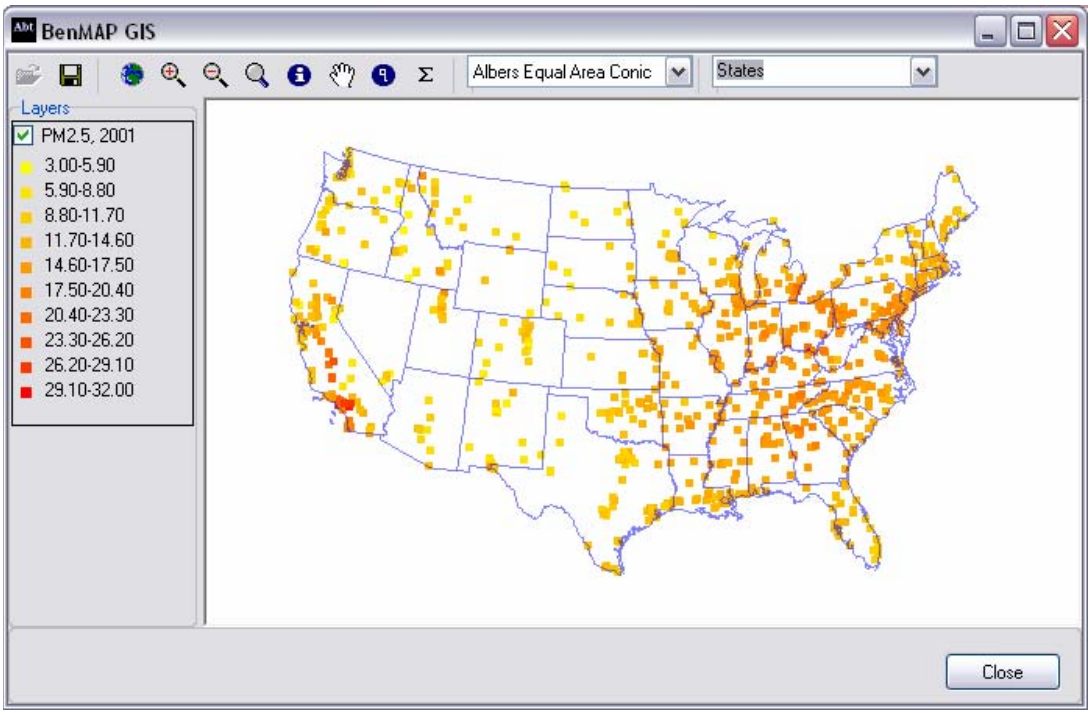


Figure 2. Year 2001 $PM_{2.5}$ monitors.

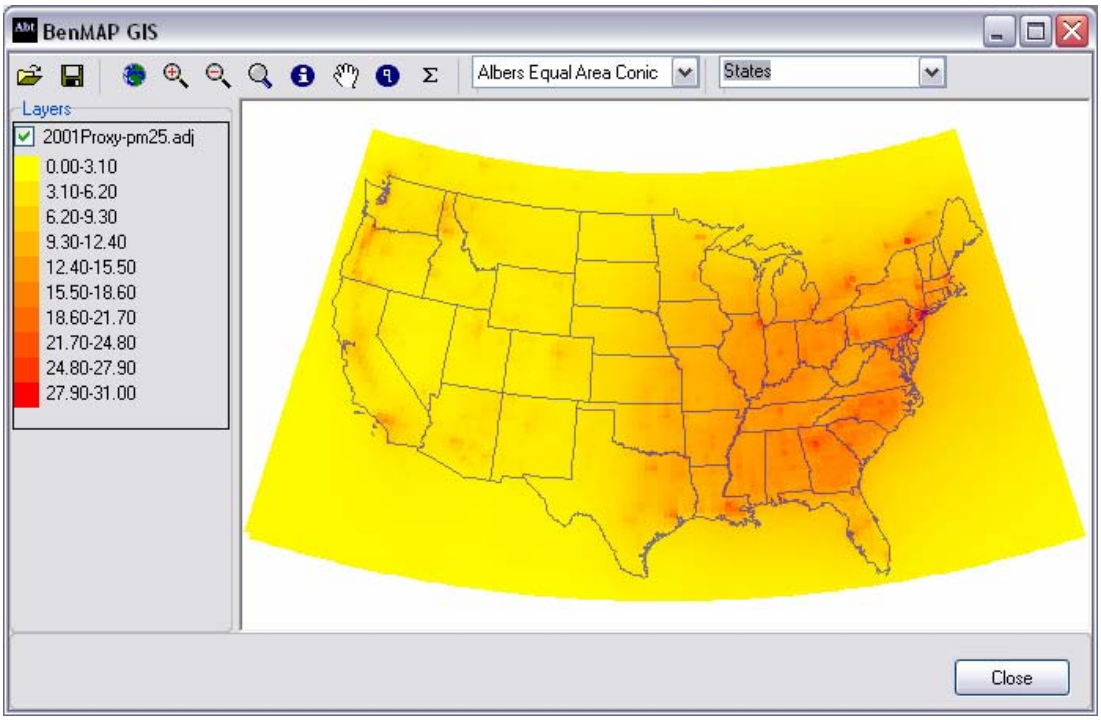


Figure 3. Base year (2001) REMSAD $PM_{2.5}$ modeling data used to spatially and temporally scale 2001 monitor data.

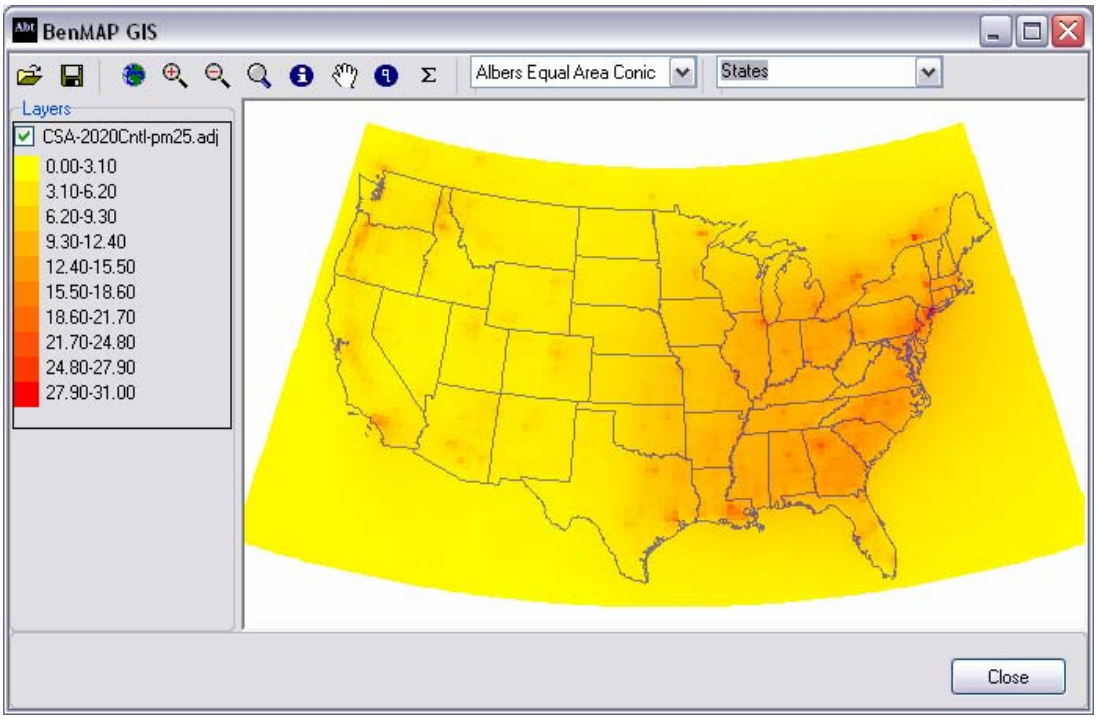


Figure 4. Future year (2020) REMSAD $PM_{2.5}$ modeling used to spatially and temporally scale 1001 monitor data.

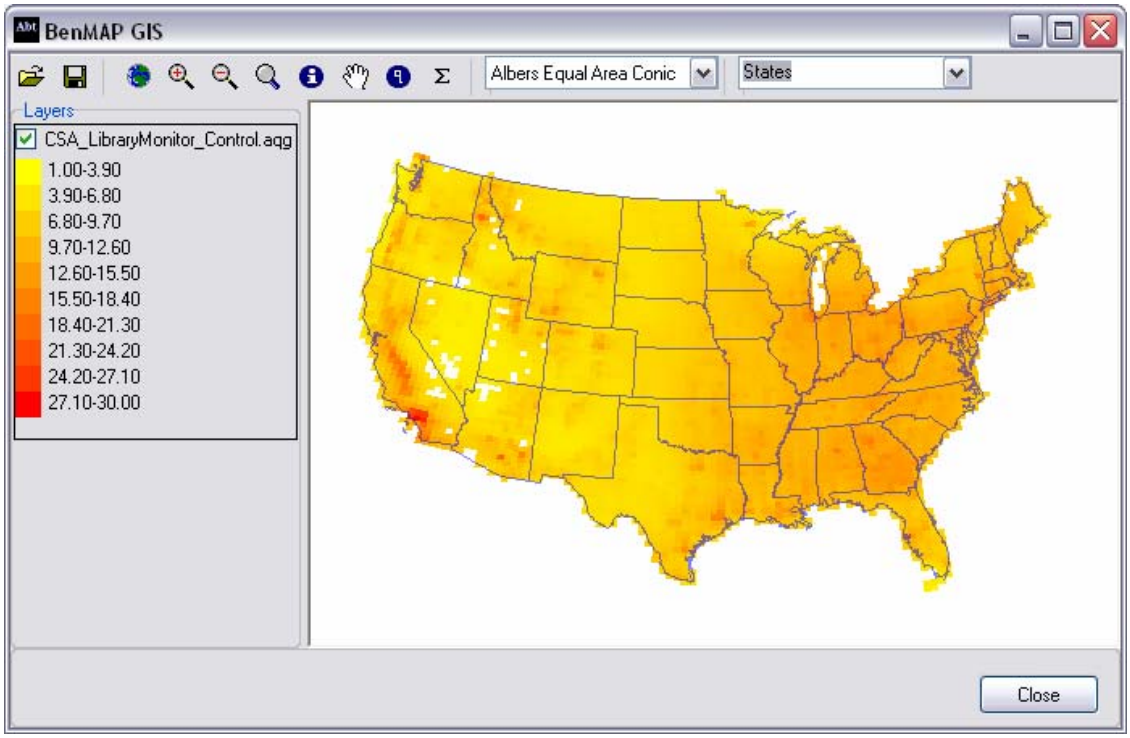


Figure 5. Scaled monitor data, reflecting a 2020 air quality scenario, interpolated to the REMSAD36 grid-cell level.

Table 2 lists a sample of the $PM_{2.5}$-related health impact functions used in the analysis, and the unit values that we used to estimate the monetary benefits. Table 3 presents the results of the analysis. Total benefits related to the selected $PM_{2.5}$ endpoints are approximately $114 billion (US 2000$), and are clearly dominated by the dollar benefits associated with premature mortality. Mortality, in fact, comprises 94 percent of total monetary benefits associated with the total. Though mortality dominates the economic valuation of benefits, BenMAP estimates many other adverse health effects that will be avoided in the future due to reductions related to the example national control scenario (Table 3).[10]

Table 2. Selected PM-related health effects and unit values.

Health Effect	Age	Epidemiological Study	$ /Case	Valuation Source
Mortality	30+	Krewski et al. (2000)	$6.0 million	Based on Viscusi (1992)
Chronic bronchitis	27+	Abbey et al. (1995)	$340,000	Based on Viscusi et al. (1991)
Non-fatal heart attacks	18+	Peters et al. (2001)	Vary by age	Based on Eisenstein et al. (2001) and Russell et al. (1998)
Respiratory Hospital Admissions	65+	Pooled estimate[a]: Moolgavkar (2000b) - ICD 490-496 Lippman et al. (2000) - ICD 490-496	$14,000	The cost-of-illness estimates (lost earnings plus direct medical costs) are based on ICD-9 code level information (e.g., average hospital care costs, average length of hospital stay, and weighted share of total category illnesses) reported in Agency for Healthcare Research and Quality, 2000 (www.ahrq.gov).
	18-64	Moolgavkar (2000b) - ICD 490-496 (less 493)	$12,000	
	65+	Lippman et al. (2000) - ICD 480-486	$18,000	
	<65	Sheppard, et al. (1999) - ICD 493	$8,000	
Cardiovascular Hospital Admissions	65+	Pooled estimate[a]: Moolgavkar (2000a) - ICD 390-429 (less 410) Lippman et al. (2000) - ICD 410-414, 427-428	$21,000	
	18-64	Moolgavkar (2000a) - ICD 390-429	$23,000	
Work loss days	18-64	Ostro (1987)	Vary by county	County-specific median annual wages divided by 50 (assuming 2 weeks of vacation) and then by 5 - to get median daily wage.
Minor restricted activity day	18-64	Ostro and Rothschild (1989)	$50	Based on Tolley et al. (1986)

[a] BenMAP generated pooled weights using the random/fixed effects approach.

Note that total benefits are the result of a dependent summation; summing across each individual endpoint's distribution of monetary benefits, we assume that the occurrence of a low (or high) estimate of incidence is shared across endpoints and that the occurrence of a low (or high) estimate of an endpoint's unit value is also shared across endpoints. The dependent summation assumption therefore makes it possible to sum across the 5^{th} and 95^{th} percentiles (or any other percentile).

[10] The quantified PM-related health effects presented here are a sample of those that BenMAP can quantify. Others include acute bronchitis, lower and upper respiratory illness, asthma exacerbations, respiratory symptoms, and infant mortality. Note that there are many other PM-related health effects that BenMAP is unable to quantify, but are known to be related to PM exposures, such as low birth weight, changes in pulmonary function, chronic respiratory diseases other than chronic bronchitis, etc.

Table 3. Selected $PM_{2.5}$-related health benefits associated with the national-level control scenario.

	Health Effects (cases)			Valuation (million 2000 $)[a]		
Health Effect	**5th**	**mean**	**95th**	**5th**	**mean**	**95th**
Mortality	8,100	14,000	20,000	15,300	107,000	261,000
Chronic Bronchitis	1,600	8,800	16,000	320	3,860	13,000
Heart Attacks	8,600	23,000	37,000	470	1,960	4,450
Resp Hosp Admissions	1,400	7,200	13,000	23	113	202
Cardio Hosp Admissions	-1,800	5,500	15,000	-39	124	316
Work Loss Days	1,400,000	1,600,000	1,800,000	182	208	235
Minor Restricted Activity Days	8,100,000	9,600,000	11,000,000	306	524	754
Total (dependent sum)	–	–	–	16,600	114,000	280,000

Note: For presentation purposes, the results were rounded.

[a] The benefits estimates include an adjustment to account for the growth in income over time, and a concomitant increase in willingness-to-pay for risk reduction.

In 2002, the Health Effects Institute (HEI) reported findings by health researchers at Johns Hopkins University and others raising concerns about aspects of the statistical methods used in a number of recent time-series studies of short-term exposures to air pollution and health effects (Greenbaum, 2002). Researchers found problems in the default "convergence criteria" used in Generalized Additive Models (GAM) and a separate issue about the potential to underestimate standard errors in the same statistical package. In response, the authors of studies affected by this problem began to reanalyze the results of several important time series studies to address these issues. In most, but not all, of the reanalyzed studies, it was found that risk estimates were reduced and confidence intervals increased. However, the reanalyses generally did not substantially change the findings of the original studies. At the time the authors conducted the case study presented here, the results of these reanalyses were not yet available.

Examination of the original studies used in this case study found that the PM-related health endpoints that were potentially affected by the GAM issues were limited only to reduced hospital admissions. All other quantified health effects, which account for over 99 percent of the total monetized benefits, were not affected by the GAM issue.

Attaining $PM_{2.5}$ Standard

In 1997, the U.S. EPA revised its National Ambient Air Quality Standards (NAAQS) for PM, as required under the Clean Air Act. The new standard required that monitored ambient $PM_{2.5}$ not exceed an annual average concentration of 15 $\mu g/m^3$. In response to their own pressing air quality problems, regulators in California set a more stringent annual $PM_{2.5}$ standard of 12 $\mu g/m^3$. Using the monitor rollback capabilities in BenMAP, we examined the benefits in California of attaining the national standard in 2002, as well as the extra benefits that California would achieve by attaining their more stringent State standard.

To estimate the benefits of the two standards, we used a population projection for 2002, aggregated the population data in a REMSAD grid of 36 kilometers by 36 kilometers, and then interpolated the monitor data using VNA with no maximum distance. Figure 6 displays the baseline air quality grid reflecting 2001 monitor data interpolated to the grid cell level. Also included on this Figure are the locations of all $PM_{2.5}$ monitors in California. Figure 7 presents the air quality associated with a rollback to 15 $\mu g/m^3$ for monitors that exceed the national standard, interpolated to the grid cell level. Figure 7 also displays the location of those monitors exceeding 15 $\mu g/m^3$ used in the rollback calculation. Figure 8 presents the air quality associated with a rollback to 12 $\mu g/m^3$ for monitors that exceed the more stringent standard, as well as the monitors used in the rollback calculation.

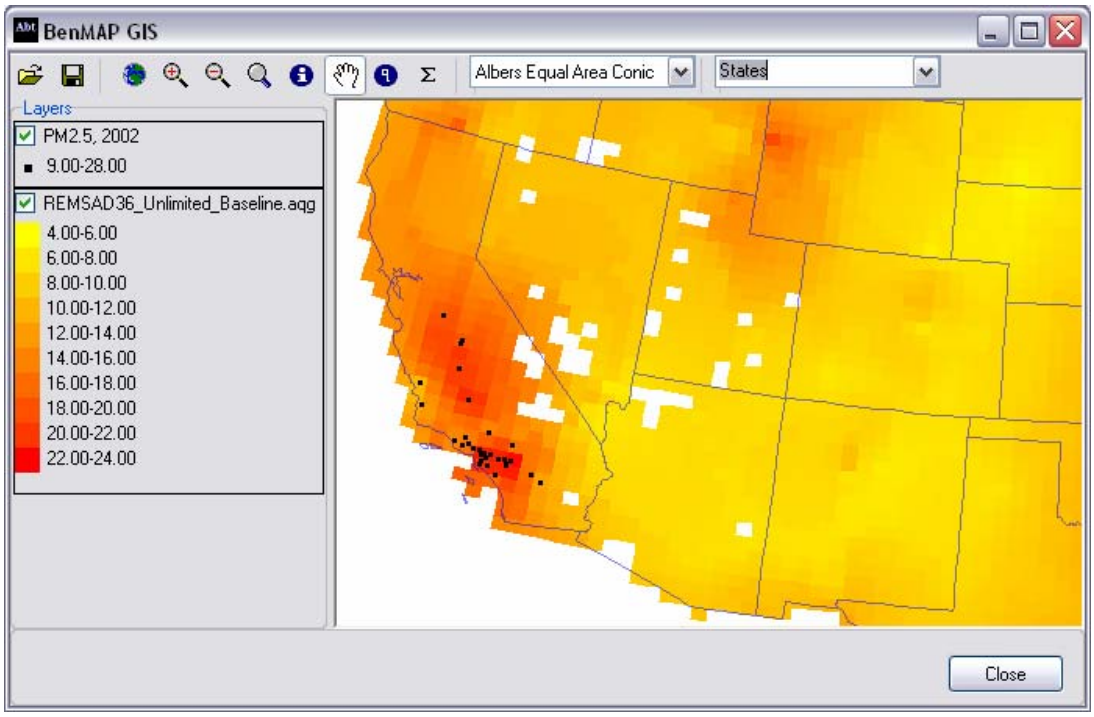


Figure 6. Baseline air quality grid reflecting 2001 monitor data interpolated to the grid cell level. Also included are the locations of $PM_{2.5}$ monitors in California.

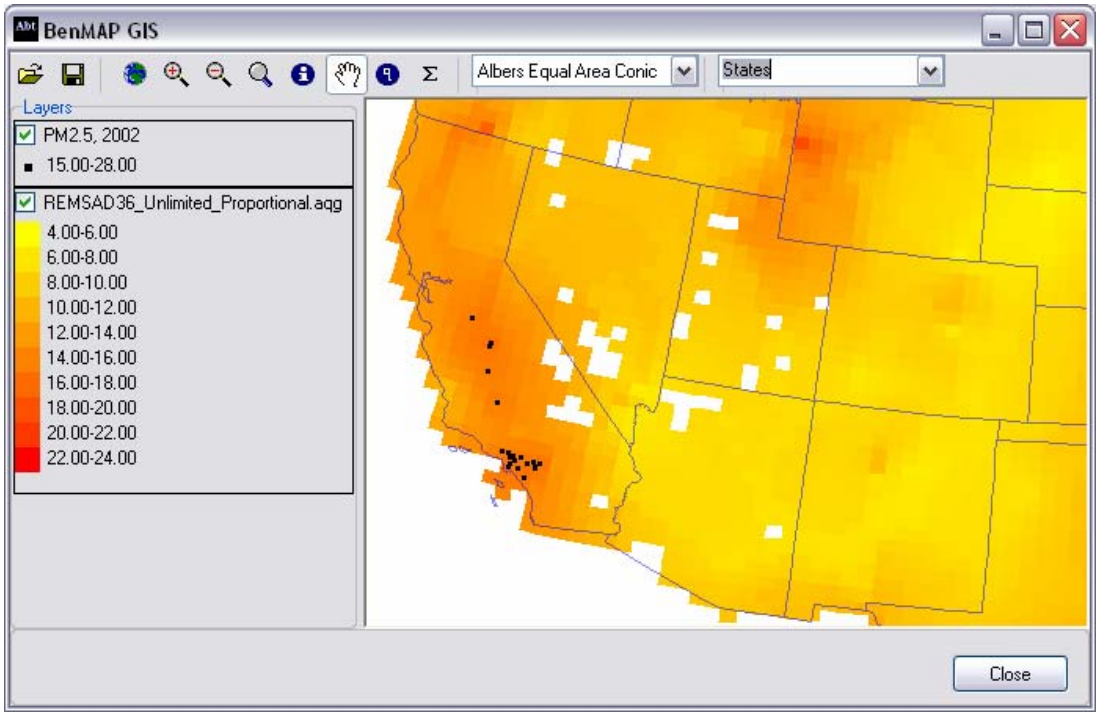


Figure 7. Air quality associated with a rollback to 15 $\mu g/m^3$ for monitors that exceed the national standard, interpolated to the grid cell level. Also included are the locations of those monitors exceeding 15 $\mu g/m^3$.

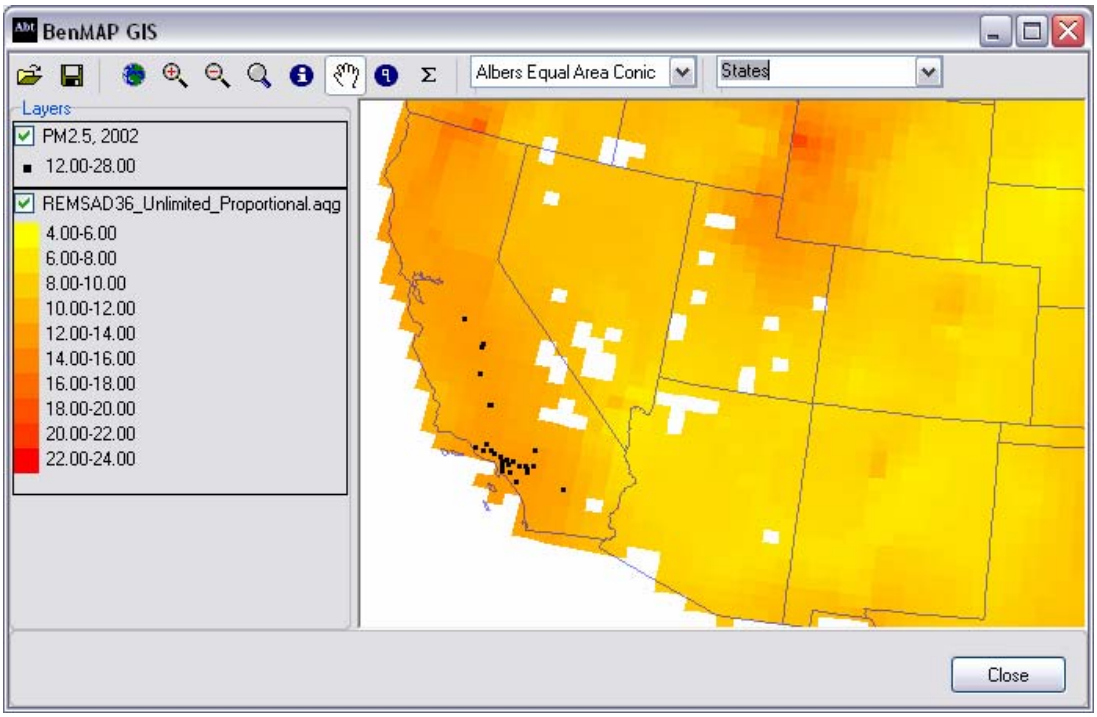


Figure 8. Air quality associated with a rollback to 12 μg/m^3 for monitors that exceed the California standard, interpolated to the grid cell level. Also included are the locations of those monitors exceeding 12 μg/m^3.

Table 4 presents the results of the analysis. The reduction in deaths associated with meeting the 15 μg/m^3 is substantial, and further lowering of the standard results in an additional 50 percent reduction. As expected, the results are sensitive to whether we used a proportional rollback or a quadratic rollback. Results are particularly sensitive to the maximum distance that we used to exclude monitors. Setting a maximum distance of 10 kilometers reduced the benefits by half. The type of grid that we used had some effect, with the smaller grid size producing a larger impact, presumably because the smaller grid is better able to capture "hot spots" with particularly high $PM_{2.5}$ levels.

Table 4. Reduction in premature mortality associated with achieving alternative annual $pm_{2.5}$ standards in California.

Grid	Max Monitor Distance	Rollback Type	Deaths Avoided by Attaining 15 μg/m^3 Standard	Additional Deaths Avoided by going from 15 to 12 μg/m^3 standard
REMSAD36	Unlimited	Proportional	5,100	2,800
REMSAD36	Unlimited	Quadratic	4,300	1,700
REMSAD36	10 km	Proportional	2,400	1,100
REMSAD36	50 km	Proportional	4,200	2,100
REMSAD12	Unlimited	Proportional	5,200	2,900
County	Unlimited	Proportional	4,000	2,500

Note: For presentation purposes, the results were rounded.

DISCUSSION

Environmental problems are increasingly a concern, and it is useful to have tools to analyze the magnitude of the problem and the benefits that may be achieved by policies to reduce the threat. BenMAP provides a useful to tool to analyze a range of air pollution problems. In particular, we have used it to estimate the benefits of reducing $PM_{2.5}$ levels through a national-level PM control program, and then to estimate the benefits of achieving alternative annual $PM_{2.5}$ standards in California.

BenMAP provides the user with a quick and flexible approach to estimating air pollution exposure, the associated health effects, and the economic benefits of avoiding these effects. In addition, it provides powerful mapping functions, the ability to characterize the uncertainty arising from the concentration-response coefficient and the unit value assigned to each health effect, and tools to aggregate and pool results in multiple ways. Another key feature of BenMAP is its transparency. BenMAP keeps track of all of the assumptions used in the analysis, allowing a user to easily replicate previous analyses, and to track the assumptions that others have made in their analyses. Through a command-line version of BenMAP, a user may also skip the graphical user interface in BenMAP, and generate the results for any number of sensitivity tests with a batch file.

The estimated benefits for the various scenarios that we have considered are significant, and highlight the serious nature of air pollution. It also highlights the usefulness of providing easily accessible results to policy makers and stakeholders. And at the same time, it makes clear the range of assumptions that underlie these types of analyses, and the need to adequately consider the inherent uncertainty and variability.

Sources of uncertainty in results

As in most complex analyses, there are many sources of uncertainty that affect the final benefits estimates. These include the emission inventories, air quality models (with their associated parameters and inputs), estimates of the future state of the world (i.e., regulations, technology, and human behavior), population estimates, epidemiological estimates of health impact functions, and estimates of dollar values per health effect, among others. For some parameters or inputs it may be possible to provide a statistical representation of the underlying uncertainty distribution. For other parameters or inputs, the necessary information is not available.
In our quantified estimate of uncertainty we have captured only two sources of uncertainty. We have quantified an estimate of uncertainty associated with the health impact function and with the valuation of adverse health effects. However, we have not included other sources of uncertainty such as in the estimation air quality, population exposure, incidence rates, and others. As a result the confidence interval can be misleading. However, it does provide at least some information about the impact of those two sources of uncertainty on the likely range of benefits.

In addition to being uncertain, the true magnitude of the benefits is also inherently variable due to the truly random processes that govern pollutant emissions and ambient air quality in a given year. Factors such as hours of equipment use and weather display constant variability regardless of our ability to accurately measure them. As such, the estimates of annual benefits generated by BenMAP should be viewed as representative of the magnitude of benefits expected, rather than the actual benefits that would occur every year.

There are a number of key assumptions that affected the calculation of our results, and how they should be interpreted. We have assumed that all types of $PM_{2.5}$ are equally harmful. While this is a reasonable interpretation of the epidemiological evidence, we note that the evidence is not definitive regarding the impacts of size and composition of particles. It could be that future research may point to certain particle types as being especially harmful, and the results of the present analysis would be substantially different. A second key assumption is in regards to the estimation and valuation of premature mortality. To estimate premature mortality, we have used a re-analysis by Krewski et al. (2000) of the American Cancer Society cohort. If we had used a daily time-series study such as that used by Burtraw et al. (2003, p. 658), our benefits estimate would have been lower by roughly a factor of three, and if we had valued mortality using the results from Mrozek and Taylor (2002) our estimate would have been reduced by another factor of three.

However, based on current advice from a number of scientific panels, we have chosen to rely on cohort epidemiological studies measuring the impact of long term exposure to $PM_{2.5}$ (U.S. EPA, 2001; National Research Council, 2002) and on an estimate of the value of statistical life consistent with the broader set of contingent valuation and hedonic wage literature (U.S. EPA, 2000). Finally, we have applied concentration-response functions only to the specific populations covered by the samples in the underlying epidemiological studies. In many cases, these samples were based solely on convenience (e.g., hospital discharge data from Medicare is limited to populations over 65) rather than on expectations about the populations potentially at

risk. This assumption will lead to an underestimation of the total health impacts associated with a given $PM_{2.5}$ reduction.

Future directions for BenMAP

The U.S. EPA is actively developing BenMAP to better characterize uncertainty and the key assumptions that drive the estimated benefits, develop better databases, and increase the public's ability to access BenMAP and the databases used in any given analysis. The improved databases will include additional monitoring data (including speciated $PM_{2.5}$), more spatially disaggregated estimates of incidence and prevalence (as well as projections of incidence and prevalence). Uncertainty will be improved in several respects, such as by including the uncertainty in incidence and prevalence estimates, including more structured correlation patterns between the available distributions, more systematic influence analysis tools, and incorporating some limited tests of the uncertainty in air quality. Finally, U.S. EPA has made BenMAP available on the web, along with the databases and assumptions used in their analyses of air regulations, in order to make the policy analysis process as transparent as possible. Interested users can download BenMAP at http://www.epa.gov/ttn/ecas/benmapdownload.html.

REFERENCES

Abbey, D.E., Ostro, B.E., Petersen, F., and Burchette, R.J. 1995. Chronic respiratory symptoms associated with estimated long-term ambient concentrations of fine particulates less than 2.5 microns in aerodynamic diameter ($PM_{2.5}$) and other air pollutants. *J. Expo. Anal. Environ. Epidemiol*. 5:137-159.

Burtraw, D., Krupnick, A., Palmer, K., Paul, A., Toman, M., and Bloyd, C. 2003. Ancillary benefits of reduced air pollutants in the United States from moderate greenhouse gas mitigation policies in the electricity sector. *J. Environ. Econ. Manage.* 45:650-673.

Eisenstein, E.L., Shaw, L.K., Anstrom, K.J., Nelson, C.L., Hakim, Z., Hasselblad, V., and Mark, D.B. 2001. Assessing the clinical and economic burden of coronary artery disease: 1986-1998. *Med. Care* 39:824-835.

Greenbaum, D. 2002. Letter to colleagues dated May 30, 2002. [Available at www.healtheffects.org]. Letter from L.D. Grant, Ph.D. to Dr. P. Hopke re: external review of EPA's Air Quality Criteria for Particulate Matter, with copy of 05/30/02 letter from Health Effects Institute re: re-analysis of National Morbidity, Mortality and Air Pollution Study data attached. Docket No. A-2000-01. Document No. IV-A-145.

Helton, J.C., and Davis, F.J. 2002. Illustration of sampling-based methods for uncertainty and sensitivity analysis. *Risk Anal.* 22:591-622.

Krewski, D., Burnett, R., Goldberg, M., Hoover, K., Siemiatycki, J., Jerrett, M., Abrahamowicz, M., and White, M. 2000. Reanalysis of the Harvard Six Cities Study and the American Cancer Society Study of Particulate Air Pollution and Mortality. Cambridge: Health Effects Institute.

Kunzli, N., Kaiser, R., Medina, S., Studnicka, M., Chanel, O., Filliger, P., Herry, M., Horak, Jr. F., Puybonnieux-Texier, V., Quenel, P., Schneider, J., Seethaler, R., Vergnaud J.C., and Sommer, H. 2000. Public-health impact of outdoor and traffic-related air pollution: a European assessment [see comments]. *Lancet* 356:795-801.

Levy, J.I., Carrothers, T.J., Tuomisto, J.T., Hammitt, J.K., and Evans, J.S. 2001. Assessing the public health benefits of reduced ozone concentrations. *Environ. Health Perspect.* 109:1215-1226.

Lippmann, M., Ito, K., Nádas, A., and Burnett, R. 2000. Association of Particulate Matter Components with Daily Mortality and Morbidity in Urban Populations. Report 95. Cambridge: Health Effects Institute.

Moolgavkar, S.H. 2000a. Air pollution and hospital admissions for diseases of the circulatory system in three U.S. metropolitan areas. *J. Air Waste Manage. Assoc*. 50:1199-1206.

Moolgavkar, S.H. 2000b. Air pollution and hospital admissions for chronic obstructive pulmonary disease in three metropolitan areas in the United States. *Inhal. Toxicol*. 12:75-90.

Mrozek, J.R., and Taylor, L.O. 2002. What determines the value of life? A meta-analysis. *J. Policy Anal. Manage*. 21:253-270.

National Research Council 2002. Estimating the Public Health Benefits of Proposed Air Pollution Regulations. Washington, DC: The National Academies Press.

Office of Management and Budget 2003. Draft 2003 Report to Congress on the Costs and Benefits of Federal Regulations. *Federal Register* 68:5492-5527.

Ostro, B., and Chestnut, L., 1998. Assessing the health benefits of reducing particulate matter air pollution in the United States. *Environ. Res*. 76:94-106.

Ostro, B.D. 1987. Air pollution and morbidity revisited: a specification test. *J. Environ. Econ. Manage*. 14:87-98.

Ostro, B.D., and Rothschild, S. 1989. Air pollution and acute respiratory morbidity - an observational study of multiple pollutants. *Environ Res*. 50:238-247.

Peters, A., Dockery, D.W., Muller, J.E., and Mittleman, M.A. 2001. Increased particulate air pollution and the triggering of myocardial infarction. *Circulation* 103:2810-2815.

Russell, M.W., Huse, D.M., Drowns, S., Hamel E.C., and Hartz, S.C. 1998. Direct medical costs of coronary artery disease in the United States. *Am. J. Cardiol*. 81:1110-1115.

Sheppard, L., Levy, D., Norris, G., Larson, T.V., and Koenig, J.Q. 1999. Effects of ambient air pollution on nonelderly asthma hospital admissions in Seattle, Washington, 1987-1994. *Epidemiology* 10:23-30.

Smith, V.K., Van Houten, G., and Pattanayak, S. 2002. Benefit transfer via preference calibration. *Land Econ*. 78:132-152.

Tolley, G.S., and Babcock, L. 1986. Valuation of Reductions in Human Health Symptoms and Risks. Prepared for U.S. Environmental Protection Agency. January.

U.S. EPA 2000. An SAB Report on EPA's White Paper Valuing the Benefits of Fatal Cancer Risk Reduction. Prepared by the Environmental Economics Advisory Committee (EEAC) of the Science Advisory Board, U.S. Environmental Protection Agency. Washington, DC. EPA-SAB-EEAC-00-013. July 27.

U.S. EPA 2001. Review of the Draft Analytical Plan for EPA's Second Prospective Analysis -- Benefits and Costs of the Clean Air Act 1990-2020: An Advisory by the Advisory Council on Clean Air Compliance Analysis. Prepared a Special Panel of the Advisory Council on Clean Air Compliance Analysis of the Science Advisory Board, Washington, DC: U.S. Environmental Protection Agency. EPA-SAB-COUNCIL-ADV-01-004. September 24.

Viscusi, W.K., Magat, W.A., and Huber, J. 1991. Pricing environmental health risks - survey assessments of risk - risk and risk - dollar trade-offs for chronic bronchitis. *J. Environ. Econ. Manage*. 21:32-51.

Viscusi, W.K. 1992. Fatal Tradeoffs: Public and Private Responsibilities for Risk. New York: Oxford University Press.

DEVELOPING RISK-BASED PRIORITIES FOR REDUCING AIR POLLUTION IN URBAN SETTINGS IN UKRAINE

Michael Brody, Jane Caldwell,[1] Alexander Golub[2]

ABSTRACT

Ukraine, when part of the former Soviet Union, was responsible for about 25% of its overall industrial production. This aging industrial infrastructure continues to emit enormous volumes of air and water pollution and wastes. The National Report on the State of Environment in Ukraine 1999 (Ukraine MEP, 2000) shows significant air pollution. There are numerous emissions that have been associated with developmental effects, chronic long-term health effects, and cancer. Ukraine also has been identified as a major source of transboundary air pollution for the eastern Mediterranean region. Ukraine's Environment Ministry is not currently able to strategically target high priority emissions and lacks the resources to address all these problems. For these reasons the US Environmental Protection Agency set up a partnership with Ukraine's Ministry of Environmental Protection to strengthen its capacity to set environmental priorities through the use of comparative environmental risk assessment and economic analysis – the Capacity Building Project. The project is also addressing improvements in the efficiency and effectiveness of the use of its National Environmental Protection Fund. The project consists of a series of workshops with Ukrainian Ministry officials in comparative risk assessment of air pollutant emissions in several heavily industrialized oblasts; cost-benefit and cost-effectiveness analysis; and environmental finance. Pilot risk assessment analyses have been completed. At the end of the Capacity Building Project it is expected that the use of the National Environmental Protection fund and the regional level oblast environmental protection funds will begin to target and identify the highest health and environmental risk emissions.

[1] US Environmental Protection Agency, Washington, DC.
[2] Environmental Defense, Washington, DC.
The views expressed in this article are those of the authors and do not necessarily represent those of the US Environmental Protection Agency.

INTRODUCTION

Ukraine, until recently, like the other countries of the Newly Independent States (NIS), and Central and Eastern Europe (CEE) operated with centrally planned economies. As they began to make the transition to a market economy and to build market institutions in the late 1980's and early 1990's, their economies came under severe pressure for a number of reasons. After a significant economic adjustment period, a gradual but slow economic recovery started first in the CEE countries and subsequently in the NIS countries. Many of the main government concerns in this period were slow economic growth, associated rates of unemployment, inflation, and social problems. Environmental concerns seemed to be less pressing due to sharp production declines resulting in reduction of all types of pollution, even though it was clear that these reductions would prove temporary. Nonetheless, countries with transition economies face certain specific problems. Environmental benefits are rarely considered in the short or even in the medium term, and willingness to pay for environmental services generally is low.

In this context, the concept of strengthening the capacity of the Ukraine Ministry of Environmental Protection was formulated during discussions between the USEPA and the MEP during meetings of the Environmental Work Group under the US - Ukraine Bi-National Commission for Economic and Technical Cooperation. This Commission was established as part of the US foreign assistance program in Ukraine. Through this process MEP representatives strongly supported more cooperation with USEPA on projects that focus directly on building their own institutional capacity. As a result, MEP and USEPA agreed on developing an environmental program with a clear emphasis on strengthening MEP's core institutional abilities. This approach was viewed as a necessary long-term strategy to enable Ukraine to successfully manage its environmental problems.

Essential elements of the CBP include: a) sharing analytical tools used in the US; b) results from other countries' (e.g., EU Tacis, Denmark) technical assistance programs in Ukraine; c) the work of multi-lateral organizations such as the Organization of Economic Cooperation and Development (OECD); and d) other relevant international experience and tools from neighboring countries, particularly Russia and to a lesser extent, Poland.

USEPA and MEP initially developed a series of analytical objectives for the Capacity Building Project (CBP). It was agreed that MEP needed to more rigorously set its environmental priorities, and that the analytical basis for these priorities should be comparative risk assessment. Due to the magnitude of pollution, the number of the pollutants emitted, and the number of significant pollution sources, Ukraine is faced with an extremely difficult task, and with very difficult choices about which sources to address and how to address them. Risk assessment, which uses estimates of exposure to hazardous pollutants and then characterizes the risk such exposure may pose, can be used to rank the relative risk of different industrial emissions. Risk assessment can provide decision criteria for MEP's actions. These analyses should be performed by Ukrainian experts, whether inside or outside the government. Risk assessments can be carried out to varying degrees of detail depending on the type of information on exposure and hazard available. Ukrainian experts and officials can use the best data sources available to them, whether the sources are national or international. At the very least, simple screening analysis can begin to provide critical decision support to the Ministry and support the case for the importance of environmental protection.

Through the efforts of this program MEP should be able to set more effective risk-based priorities in regional and national level environmental policies and increase the efficiency and performance of its environmental investments. Additionally, the Ukrainian government will have more tools with which to meet its international environmental treaty obligations. Finally, as a result of its efforts, MEP should be able to more effectively argue the significant health and economic benefits of environmental protection to the Ukrainian Government.

A Brief Synopsis of Approaches to Air Pollution Risk Assessment and Regulation in the United States

The purpose of this section is not to create a template for Ukraine to copy, but to provide a glimpse of the US experience in dealing with air pollution, and the ongoing need for flexibility and the ability to change approaches. The original Clean Air Act was signed into law in 1963 and was considered to be the first modern environmental law enacted by the United States Congress. It was the Clean Air Act of 1970 (amended by Congress in 1975 and 1977), that formed the basis of federal pollution control and it used health-based national ambient air quality standards as its approach. The standards were to be met by application of control technology with cost and technological

considerations to be subordinate to public health protection. All requirements were to be national with no facility having a competitive edge by having to meet less stringent controls. The EPA was responsible for carrying out the program (Clean Air Act: Law and Explanation, 1990, www.epa.gov/air/oaq_caa.html).

The EPA set two kinds of National Ambient Air Quality Standards (NAAQS) that specified acceptable concentrations of pollutants in outdoor air. Primary standards were set to protect human health while secondary ones were developed for plants and animals for six common pollutants (Criteria Pollutants). Lists of hazardous air pollutants (HAPs) were created (e.g., asbestos, beryllium, mercury, benzene) and regulations were developed to control sources of those pollutants. To limit pollution from mobile sources (e.g., cars) controls were placed on emissions (e.g., catalytic converters and use of unleaded gasoline). Thus, national standards were in effect for most major industries. The fifty States were assigned important roles in implementation. Both national and local authorities have responsibilities, cooperation has been a necessity, and there are checks and balances built into the system.

However, due to the ongoing difficulty in the attainment and setting of health-based air quality standards there have been further changes to the clean air laws. The 1990 Clean Air Act Amendments (CAAA) contained new and specific regulatory deadlines and actions to combat pollution. Almost all major cities in the United States did not meet the NAAQS for one or more pollutant with the most widespread problem being that of ozone. The 1990 CAAA imposed more stringent controls on automobiles for areas with greatest pollution. Emissions of the hazardous air pollutants were to be controlled through a technology standard. The HAPs were identified for control with a goal of 75% reduction within 10 years through use of maximum achievable control technology (MACT). Rules to control these pollutants under the original statute were often delayed while trying to produce enough evidence on risk to justify regulation. So the 1990 CAAA tried to define away the risk problem by identifying pollutants thought to pose a risk and then mandating the MACT standard. Yet, the 1990 CAAA did require risk assessments to determine whether further reductions would be needed. But a national scale assessment of HAPs concentrations in the air and their likely health risks had never been conducted. The overall health benefit of the MACT standards could not be easily ascertained. So it has not been possible to avoid the inevitable questions about the nature and scope of the health risks to be reduced.

The approaches taken to tackle the problem of understanding and targeting the risks posed by HAPs can be informative for other situations such as the Ukraine. Under the 1990 CAAA the HAPs include 188 specific pollutants and chemical groups, many of which are associated with adverse health outcomes including cancer, neurological, respiratory, reproductive, and developmental effects. Most known health effects of the HAPs are derived from animal and occupational studies with little information on potential health risks from chronic low level exposures to the public (USEPA, 1994; Leikauf et al., 1995). In contrast to Criteria pollutants, little monitoring data exist for the large number of HAPs listed. Their varied chemical nature and heterogeneous geographical distribution makes comprehensive monitoring exceedingly difficult. The USEPA has used a modeling approach to estimate long-term annual average outdoor emissions of the 188 HAPs originating from a myriad of sources (USEPA's Cumulative Exposure Project or "CEP"). Concentrations were estimated for every census tract in the continental United States for a base year of 1990 (Rosenbaum et al., 1999a; Rosenaum et al., 1999b). Modeled HAP concentrations were then compared with previously defined benchmark concentrations for cancer and non-cancer health effects (i.e., the 1/million extra risk for cancer level and the inhalation reference concentrations, [a level at which there is little probability of non-cancer effects]). Uses of the information included identification of pollutants that were ubiquitously high in several geographical locations, pollutants that may pose the greatest risk near population centers, and sources of such pollutants (Woodruff et al., 1998; Caldwell et al., 1998; Woodruff et al., 2000; Axelrad et al., 1999). This approach provides an estimate of health risks rather than measures of the frequency of disease occurrence. A use of the CEP approach has been applied to a more localized level for the state of California (Morello-Frosch et al., 2000) and is the basis of EPA's National Air Toxics Assessment (NATA).

The NATA is an ongoing effort, based on the approach used in the CEP, to track the risk posed through time. Goals of a national assessment are: i) to inform priorities for regulatory programs, ii) to assess progress toward national risk-based goals, iii) to inform efforts to allocate resources to further investigate (e.g. monitoring) problems on a broad or local scale and iv) to support prospective assessments of estimated benefits of air toxics programs (USEPA Science Advisory Board, 2000). The results of NATA are publicly available. EPA's extensive outreach efforts to

communicate the results include a website that includes information about the assessment, frequently asked questions, results (data, maps, charts) and discussions on interpretation. Even with the best national assessments there will always be uncertainties. One of the largest sources of uncertainty will continue to center around the inventory of the facility emissions. These data will always be the best available, but uniformity will be lacking; some will be from actual measurements, but most will be based on some method of emission factors or mass balance from the sources themselves. But it is important to emphasize that well-understood, peer-reviewed methods and tools exist for the hazard and exposure analyses of risk assessment.

Russian Experience with Risk Assessment

Ukraine was a very significant industrial region of the former Soviet Union. Comprehensive regulation of air pollution in the Soviet Union was established in the late 1970's. The maximum allowable ambient concentration (MAC) was a key environmental quality standard that was established for more than 100 pollutants. These standards established maximum values for one-time concentrations and daily average concentrations. Introduced in 1969, MAC standards complied with medical requirements and were very strict. For example, the MAC for SO_2 was 0.05 mg/m^3, as compared to 0.26 mg/m^3 in the US[3]. Such strict standards were in actual practice unattainable. Based on the MAC values, Maximum Permissible Levels of Emissions (MPL) for enterprises (stationary sources of pollution) were established. Standards for concentrations of harmful substances in emissions from mobile sources were also set. However, in practice only automobile emissions of CO have been controlled. MAC standards were applied to both new and existing enterprises. However, the stringency of the standards was offset by the lack of compliance due to poor monitoring and enforcement. While calculated for the individual source, MPL does not take into account background ambient concentration of the pollutant from other sources. Therefore, real concentrations of harmful substances in the atmosphere exceeded MAC's significantly. In addition, MPL's were difficult to use because maximum allowed emissions were established for each source of emissions, that is, for each stack, rather than for the facility as a whole. There are companies which have hundreds of individual pollution sources, and the standards are set for each of them. Many companies were not and are still not able to reach the MPL emissions. Therefore following the introduction of these standards, temporary standards (TSP) were also introduced which have become the real emission control tools.

By the end of the 1980's this system almost collapsed. But then a system of pollution fees was introduced (Golub and Strukova, 1994; Kolstad and Golub, 1993). However, this system inherited the major weak point of the previous regulation system, which was the enormous number of regulated pollutants. Introduction of pollution fees helped to improve the monitoring system, but it is very difficult to judge whether or not it had a measurable environmental outcome. During the economic crisis of the early post-Soviet period the level of air pollution declined, while emission per unit of output increased (Golub and Strukova, 1994; Kolstad and Golub, 1993).

In 1996 the first health risk analysis study was conducted in Volgograd (Larson et al., 1999). The study was based on the risk assessment methods of, and training provided by the USEPA, and was led by the Harvard Institute for International Development and the USEPA. Risk analysis was considered important to identify priorities given the scarce resources available in countries in transition. The risk assessment was based on the existing air and water emissions inventory of the multiple sources in Volgograd. Concentrations in the air were estimated using a Russian air dispersion model. USEPA health-based benchmarks and slope factors were used as a starting point. Russian air dispersion models designed to estimate ambient concentrations focused on 20 minute and 24 hour concentrations, which were the bases for regulation in that country. For risk assessments the annual average concentration is needed. For the Volgograd study the results of calculations were adjusted by experts (Larson et al., 1999). In an assessment in Nizhnii Tagil there was an attempt to calibrate the US air dispersion model and calculate annual average concentrations directly. Although the Russian modeling methodology has limitations, the benefits include the availability of data to run the model, and general acceptance of the methods. The Volgograd risk assessment estimated that among hundreds of pollutants only one of them: PM_{10} was responsible for more than 90% of the estimated mortality risk. Among dozens of industrial enterprises, only two were responsible for the major share of the impact on public health. These analyses were then used to develop risk mitigation priorities, with significant

[3] More examples of MAC's for various substances can be found in Golub and Strukova (1994), page 168.

input from epidemiology. Results from other health risk assessments in Russia are described in Oniszhenko (2002) and Danilov-Danilian et al. (2003).

A very significant proportion of Soviet industrial development occurred in Ukraine. By analogy to this first generation of risk assessments in Russia, it is fairly reasonable to assume that significant human health risks result from industrial pollution there. In 1999 approximately 2% of total mortality in Russia and 8% of morbidity (seen mainly as respiratory illness) was likely due to environmental pollution (calculations based on Bobylev et al., 2000). It is likely that environmental pollution takes a similar toll on public health in Ukraine. Ukraine also has some factors that may exacerbate the situation; these being higher population density and a fuel mix that probably produces more pollution. To generate more rigorous estimates for Ukraine it is necessary to carry out risk assessments within that country.

METHODS

Applying Risk Assessment to Setting Environmental Priorities in Ukraine

The numerous industrial facilities and mobile sources in Ukraine release very significant amounts of criteria air pollutants or their precursors, heavy metals, volatile organic compounds, and persistent organic pollutants. These emissions are linked to increased risk of numerous health effects including respiratory ailments, cancer, developmental, and/or neurotoxic effects. The extent of the problems and the constraints of cost require as a first step the careful weighing of relative harm of the many industrial emissions. Then setting priorities for control of these emissions can begin as the effectiveness and cost of control is considered.

Because risk assessment is relatively new to Ukraine to achieve the ultimate goal of creating acceptance and use of risk requires Ukrainian officials and technical specialists to learn the concepts and practice of risk assessment on their own. Risk assessment, risk management and their practical policy application must occur in their own country and environmental context. To accomplish this CBP objectives include: 1) the conduct of pilot risk assessments and priority-setting activity within several oblasts to show the effectiveness and benefits of the methodology and its relevance to Ukrainian conditions; 2) using “learning by doing” approaches to help Ukrainian specialists to acquire practical skills in risk assessment, and thereby train a core group of Ukrainian risk assessors; 3) harmonize Ukraine's existing retrospective, epidemiological approaches with the prospective approach of risk assessment; and 4) introduce the use of risk assessment to project evaluation procedures for the National and Oblast Environmental Protection Funds.

To set priorities for reducing the emissions, they must be ranked by their potential to cause harm. Setting priorities requires that we try to generate the most important pieces of information that we can learn about potentially harmful emissions. Perhaps the most difficult and uncertain but important step in conducting a risk assessment is to define the extent and content of emissions. In order for the assessment to have the greatest validity for defining the nature and scope of the Ukrainian pollution problem, it is essential to gather data specific to the areas or regions to be assessed. The information on facility emissions must not only describe what is being emitted but also in what amounts. To do the exposure analysis, appropriate population data must also be gathered. Pollutant dispersion models are already available from a number of sources. Toxicological information on the pollutants is also readily available. Medical epidemiological data from Ukraine can clearly be useful to understand the effects of industrial emissions on human health. However, epidemiology studies are by their nature retrospective and attempt to identify associations of health conditions with past emissions. This is typically difficult as these past emissions are among many factors influencing health. In the context of prioritizing the control of pollution it is important to include risk assessment's forward-looking, prospective view and to estimate the reduction in adverse health effects resulting from the reduction of emissions.

RESULTS AND DISCUSSION

Phase 1 CBP Activities

The Capacity Building Project (CBP) began with an Inter-ministerial workshop (November, 2002). It was designed with two purposes: 1) to build understanding of the approach by illustrating many of the principles of modern environmental policy; and 2) to facilitate partnership and to exchange experience between the two governments and share best practices available in multilateral international institutions. The conference was organized by the NGO Counterpart International, a grantee of the USEPA, in close consultation with both the EPA and the MEP. Presenters included representatives of the US and Ukrainian governments, the OECD and the World Bank, as well as Ukrainian, US and international experts in the fields of risk assessment, environmental economics and environmental finance. Most participants were officials from the MEP, both from the national level and nearly all of the oblasts. Additionally, the Cabinet of Ministers, Ministries of Economy, Finance and Health also sent their representatives.

While some of the specific topics covered in the workshop were familiar to various participants, the incorporation and integration of these ideas and practices into building core institutional capacity as a system was new and challenging. Open, transparent goal setting by government is the ultimate basis for good governance and broader public support. The task for MEP and the Government of Ukraine is to develop the capability of incorporating these practices into their own specific context. The workshop led to the formation of a workgroup that included all of the participating ministries and led to the selection of two oblasts to host pilot assessments. After the introductory workshop we began a series of parallel workshops to 1) begin the comparative risk assessment case studies and 2) promote risk-based management of Ukraine's environmental protection funds.

The first workshop to begin the pilot risk assessment on the local level was held in heavily industrialized Zaporyzhzhia oblast. Participants were from the oblast level Environment and Health Ministries, staff from the City Council, and representatives from industrial enterprises and local academic institutions. Presentations on US practice of risk assessment and its adaptation in Russia preceded sessions specific to the prospective local comparative risk assessment. This first "hands-on" workshop led to several key findings. The first confirmed the Russian experience with MAC standards – 400 priority chemicals – but attainment had been reached for very few. With preliminary emissions data from three industrial enterprises, it was possible to demonstrate an updated version of Russian air dispersion models that have now been linked to US geographic information system technology. Ukrainian environmental health specialists already have in-depth knowledge of their worst contaminants, but they do not always have easy internet access to the latest information that is available from sources such as the USEPA or the World Health Organization. But finally, and equally important to all of the technical issues was that these workshops are required to develop the trust and conditions for trying new approaches like comparative risk assessment. This was an opportunity to promote the benefit of risk assessment methodology and its relevance and practicality to Ukrainian conditions and context for risk management.

As a result, more detailed emissions data were made available and initial comparative risk assessments in this oblast has been completed. The first pilot risk assessment project in Ukraine was implemented in Zaporyzhzhia city. It used to be one of the most polluted cities in Ukraine. The study covered only three industrial enterprises since environmental authorities were reluctant to conduct a comprehensive study before testing the method. Exposure was calculated only for two districts in Zaporyzhzhia:

- Zavodskoy district with a population of about 61,000;
- Ordjonikidze district with a population of about 106,000.

The study applied adjusted dose-response coefficient for PM_{10} equal to 0.5% per 10 $\mu g/m^3$, and for SO_2 equal to 0.6% per 10 $\mu g/m^3$ SO_2. Baseline mortality for Zaporyzhzhia city is 8.4 per 1000. The results are provided in Tables 1 and 2.

Table 1. Annual additional mortality from PM_{10}.

Receptor point	PM_{10} Annual concentration μg/m^3	PM_{10}		Additional mortality (cases/year/per 1,000,000)
		Individual risk	Additional mortality (cases/year)	
Zavodskoy	0.137	$8.4*10^{-4}$	51	836
Ordjonikidze	0.085	$5.2*10^{-4}$	55	518

Table 2. Annual additional mortality from SO_2.

Receptor point	SO_2 Annual concentration μg/m^3	SO_2		Additional mortality (cases/year/per 1,000,000)
		Individual risk	Additional mortality (cases/year)	
Zavodskoy	0.125	$9.2*10^{-4}$	56	915
Ordjonikidze	0.118	$8.6*10^{-4}$	92	864

The analysis demonstrates that a dominant fraction of health risk comes from conventional pollution such as PM_{10} and SO_2. The share of health risk from carcinogens was relatively smaller. The results are similar to those obtained in Russian cities (Larson et al., 1999).

This pilot study demonstrates that this method could be successfully implemented using local primary data and local air dispersion modeling capacity, which is a critical aspect of the analysis gaining acceptance and use.

Based on this work, economic valuation will be used to help direct resources to those control projects which yield environmental benefits at the least cost. The most important categories of goods and services provided by the environment that need to be valued are environmental health risks, natural resources, and environmental services and amenities including bio-diversity. Valuation methodologies and benefit estimates based on health and environmental risk analyses would be a more standard approach, and would optimize environmental policy by targeting a reduction in human health risk from decreased environmental pollution at the least cost. Health risk reduction will likely become the leading criterion for deciding on the limited number of pollution control projects that are likely to be feasible. A pilot risk management workshop based on these analyses will be held to develop the risk-based decision making process. We believe that based on this preliminary effort, authorities are likely to make available emissions data from the other major enterprises. At that point the risk assessments can be expanded, and we can also begin to better develop other endpoints and routes of exposure.

In addition to workshops on risk, a parallel series of workshops are addressing environmental finance. Like Russia, Ukraine levies pollution fees on emissions. These fees go into a group of national, oblast and municipal environmental protection funds. These funds are major for environmental finance in Ukraine. The CBP is also addressing improvement of the management of these funds. Officials involved in the management of the funds are examining a series of issues including: updating financial management techniques, and consolidation and harmonization of funds at the different governmental levels. Explicit linkages between the use of the funds and the priorities set through comparative risk assessment are being explored.

CONCLUSIONS

The USEPA – MEP Capacity Building Project is still in its early stages and has not yet achieved concrete environmental outcomes. However, Ukraine is a country with severe urban air quality problems. A fundamental choice has to be made in any environmental assistance project that is carried out there. Resources can be used for a specific environmental pollution control project. Or resources can be directed to assist Ukraine in developing sustainable strategies and capabilities to manage environmental problems on their own, over the long term, with acceptance of the current set of constraints. Ukraine's MEP requested assistance based on this second path. The CBP, as described in this paper, has been designed to support that path. We have started reporting and will continue

to report on actual results as the CBP progresses.

Acknowledgements

This work was supported in part by US EPA cooperative agreements with Counterpart International and Environmental Defense. The authors would like to thank the efforts of Dr. Vadim Diukanov, Counterpart International, Kiev Ukraine for assistance in coordinating the risk assessment, and Dr. Simon Avaliani, Russian Academy of Advanced Medical Studies for direct assistance and quality control of the risk assessment.

REFERENCES

Axelrad, D., Morello-Frosch, R., Woodruff, T., and Caldwell, J. 1999. Assessment of estimated 1990 air toxics concentrations in urban areas in the United States. *Environ. Sci. Policy* 2:397-411.

Bobylev, S., Avaliani, S., Golub, A., Sidorenko, V., Safonov, G., and Strukova, E. 2000. Macroeconomic assessment of environment related human health damage cost for Russia. Moscow State University.

Caldwell, J., Woodruff, T., Morella-Frosch, R., and Axelrad, D. 1998. Application of health information to hazardous air pollutants modeled in EPA's cumulative exposure project. *Toxicol. Ind. Health* 14:429-454.

Danilov-Danilian, V., ed. 2003. Climatic Change, View from Russia. Center for Russian Environmental Policy and Environmental Defense. Moscow, TEIC.

Golub, A., and Strukova, B. 1994. Application of a Pollution Fee System in Russia. In *Economic Instruments for Air Pollution Control,* eds. Ger Klaassen and Finn R. Forsund, pp.165-184. Dordrecht/ Boston/ London: Kluwer Academic Publishers.

Kolstad, C.H., and Golub, A. 1993. Environmental protection and economic reform in Russia. *EPAT/MUCIA Policy Brief* No 2, July 1993.

Larson, B., Avaliani, S., Golub, A., Rosen, S., Shaposhnikov, D., Strukova, E., Vincent, J., and Wolff, S. 1999. The economics of air pollution health risks in Russia: A case study of Volgograd. *World Dev.* 27:1803-1819.

Leikauf, G., Kline, S., Albert, R., Baxter, C., Berstein, D., and Buncher, C. 1995. Evaluation of a possible association of urban toxics and asthma. *Environ. Health Perspect.* 106:253-271.

Morello-Frosch, R., Woodruff, T., Axelrad, D., and Caldwell, J. 2000. Air toxics and health risks in California: the public health implications of outdoor concentrations. *Risk Anal.* 20:273-291.

Oniszhenko, G., Avaliani, S., Novikov, S., Rakhmanin, U., and Bushtueva, K. 2002. Basis for human health risk assessment resulting from chemical pollutants. *Moscow: NII ECH&GOS.*

Rosenbaum, A., Axelrad, D., Woodruff, T., Wei, Y., Ligocki, M., and Cohen, J. 1999a. National estimates of outdoor air toxics concentrations. *J. Air Waste Manage. Assoc.* 49:1138-1152.

Rosenbaum, A., Ligocki, M., and Wei, Y. 1999b. Modeling Cumulative Outdoor Concentrations of Hazardous Air Pollutants: Revised Final Report. *Systems Applications International, Inc.* San Rafael CA.

Ukraine MEP. 2000. National Report on the State of Environment in Ukraine 1999. Rayevsky Scientific Publishers, Kyiv.

USEPA. 1994. Technical Background Document to Support Rulemaking Pursuant to Clean Air Act Section 112(g): Ranking of Pollutants with Respect to Human Health. *USEPA.* Research Triangle Park. NC.

USEPA, Science Advisory Board, 2000. Summary of July 2000 peer review of the draft document *Planning and Scoping the Initial National-Scale Assessment: An element of the EPA National Air Toxics Program*. (www.epa.gov/ttn/atw/nata/peer.html) August 27, 2003.

Woodruff, T., Axelrad, D., Caldwell, J., Morello-Frosch, R., and Rosenbaum, A. 1998. Public health implications of 1990 air toxics concentrations across the United States. *Environ. Health Perspect.* 106:245-251.

Woodruff, T., Caldwell, J., Cogliano, V., and Axelrad, D. 2000. Estimating cancer risk from outdoor concentrations of hazardous air pollutants in 1990. *Environ. Res.* 82:194-206.

AIR QUALITY IMPROVEMENTS WITH EUROPEAN ENVIRONMENT POLICIES: SO_2 CASE STUDY IN A COASTAL REGION IN PORTUGAL

M.C. Pereira, R.C. Santos, M.C.M. Alvim-Ferraz[1]

.

ABSTRACT

The European Union (EU) has defined several efforts for preventing and controlling air pollution that have led to a generalized reduction of health risks and environmental effects. One example of these efforts was the establishment of legislation that imposed a reduction of sulphur content in fuels, switching to cleaner fuels for industry and motor vehicles (Auto-Oil Directives). The goal of this work is to evaluate the first trends in the air quality of Oporto Metropolitan Area (Oporto-MA) related to implementation of these Directives in Portugal since 2000. Therefore, sulphur dioxide (SO_2) concentration data from air quality monitoring sites, were analysed for the period 1999 through 2003. It was observed that daily SO_2 concentrations in the industrial area are much higher than those of the urban and rural areas reflecting the influence of the industrial sector. The annual SO_2 concentrations for the consecutive years of analysis have decreased considerably for all the monitoring sites. A considerable decrease in SO_2 emissions was also observed for the same period. According to EU legislation, the exceedances of the SO_2 recommended limit values decreased significantly. In conclusion, this analysis shows evidence that SO_2 concentrations are decreasing, and that this is consistent with the implementation of the legislation and control of the sources of pollution. Nevertheless, the potential for human health risks still persists particularly in industrialized areas. It is expected that SO_2 concentration levels in ambient air of Oporto-MA will continue to decrease in the coming years following the present European trends.

[1] Universidade do Porto, Porto, Portugal.

INTRODUCTION

The generalized trend of increasing pollutant emissions to the atmosphere coupled with concerns about the potential impact of pollution control strategies on economic development make it essential that effective air quality management be employed to protect human health and the environment. Pressures on the environment are permanent and it is unrealistic to think that the presence of pollutants in the ambient air can be avoided, because in practice it is impossible to eliminate natural emissions as well as those associated with anthropogenic activities. Considering that corrective actions are more expensive and less efficient than preventive ones, it is important to define and employ effective pollution prevention policies. With this principle in mind, the European Union (EU) has defined several efforts for preventing and controlling air pollution, that have led to a generalized reduction of health risks and environmental effects (Leeuw et al., 2001; Larssen et al., 2002). One example of these efforts was the establishment of legislation known as Auto-Oil Directives (EC Directive, 1998a; EC Directive, 1998b) that imposed a reduction of sulphur content in fuels, switching to cleaner fuels for industry and motor vehicles. This legislation resulted from the Auto-Oil programme which was created in 1992 to assess the future trends in emissions, air quality and also to establish a consistent framework within which different policy options to reduce emissions would be assessed using principles of cost-effectiveness, sound science and transparency, and thus provide a foundation for the transition towards longer term air quality studies covering all emission sources (DGXI, 1996; CEC, 2000).

In Portugal, the limits of atmospheric concentrations of sulphur dioxide (SO_2) and other pollutants such as nitrogen dioxide, carbon monoxide, ozone, total suspended particles and lead were established for the first time in 1993 with the aim of reducing and controlling air pollutant emissions (Portaria nº 286/93). For SO_2 the scenario presented required a reduction in emissions of 56.5 % between 1993 and 2001. The Auto-Oil directives transposed to internal laws were another important step for the reduction of SO_2 emissions (Decreto-lei nº 104/2000, Decreto-lei nº 281/2000).

The atmospheric concentration limits are being changed according to the publication of the air quality Framework Directive (FWD) (EC Directive, 1996) that defines the new strategy for air quality management in Europe. The FWD was approved for the Member States, resulting in the Daughter Directive that sets the new limit-values for sulphur dioxide, nitrogen dioxide, PM_{10} and lead (EC Directive, 1999). The Daughter Directive was transposed to Portuguese internal law in April 2002 (Decreto-Lei nº 111/2002).

Oporto Metropolitan Area (Oporto-MA) is the second most populous Portuguese region with about 1.2 million inhabitants and a population density of 540 inhabitants per square kilometre. It is located at a latitude of 41º10' N and at a longitude of 8º40' E. The region is limited on the west by the Atlantic Ocean with 50 km of coastline, and crossed by the Douro River. In climatic terms this area is designated North maritime, characterized by warm and dry summers and mild and wet winters with two transition seasons: spring and autumn. Topographically the zone is rather uneven, with a maximum altitude of about 300 m that decreases with proximity to the coast. The annual average air temperature varies between 12.5 and 15 ºC. The difference between warmer and colder monthly averages is less than 10 ºC. Annual air humidity is between 75 and 80%, and the total annual mean precipitation is between 1000 and 1200 mm, with more than one hundred days per year with precipitation equal to or higher than 1.0 mm. Prevailing winds are from W and NW. In the north of Portugal, Oporto-MA is the region with the highest power consumption per capita and per industry, which is related to a relatively high industrial density. The motorization rate, defined as the number of motor vehicles per capita, is the second highest in Portugal.

In Oporto-MA the most important stationary sources of atmospheric pollutants are one oil refinery, one petrochemical plant, one thermoelectric plant working with natural gas, one incineration unit and one international shipping port. Nevertheless, motor vehicle traffic is estimated to be responsible for a significant amount of pollutants emitted to the atmosphere (Borrego et al., 2000).

The goal of this work is to evaluate the first trends in the air quality of Oporto Metropolitan Area (Oporto-MA) since the implementation of these Directives in Portugal in 2000. Therefore, SO_2 data from four selected monitoring sites (2 urban/traffic, 1 industrial and 1 rural), were analysed for the period 1999 through 2003.

METHOD

Four representative monitoring sites belonging to the Air Quality Monitoring Network of Oporto-MA, managed by the Commission of Coordination and Development of North Region (CCDR-N), under the responsibility of the Ministry of Environment, were selected to characterise the area according to the main kinds of emission sources: two urban/traffic (Paranhos and Boavista), one industrial (Custóias) and one rural (Vila Nova da Telha) (Figure 1).

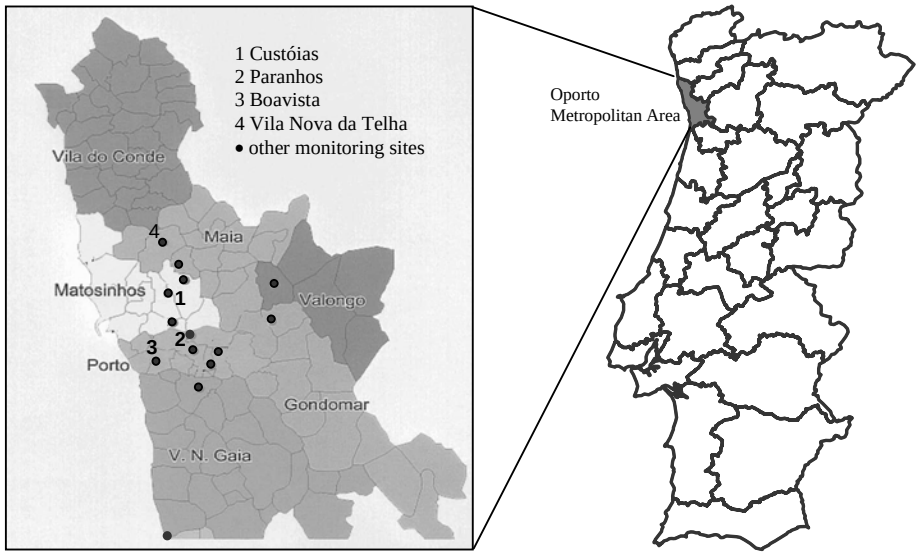


Figure 1. Map of Portugal and Oporto-MA showing the CCDR-N monitoring sites.

Paranhos (urban/traffic1) and Boavista (urban/traffic2) are located 8 km and 4 km, respectively, from the coastline, in densely populated areas of two urban/traffic zones in the city of Oporto. Several public institutions, shopping centres, high and secondary schools, and one university are located in these areas. These sites are near two of the most important access points to the motorway, which means that traffic emissions are the main source of atmospheric pollutants.

Custóias is situated about 5 km from the coastline, in an open area of a suburban industrial zone of the city of Oporto. It is situated to the west of the oil refinery and is influenced mainly by atmospheric emissions from this plant as well as by other industrial emissions transported by prevailing winds from W and NW. Road traffic and urban activities do not significantly influence air quality in a direct way at this site, except on Saturdays, when a large street market takes place.

Vila Nova da Telha is located 6 km from the coastline in a suburban rural area, the monitoring site being installed in a primary school playground. Because of the human activities in the surroundings, it is not significantly influenced by traffic, urban or industrial emissions in any direct way. As Portugal is not significantly influenced by pollutants coming from other countries, the air quality in this area is mainly influenced by intra-regional transport of pollutants, and is therefore appropriate to be considered as background concentration for atmospheric pollution in Oporto-MA.
SO_2 concentrations were obtained using the UV Fluorescence method according to EC Directive 1999/30/CE and Decreto-Lei nº 111/2002, using the AF21M equipment from Environment SA. This equipment is submitted to a rigid maintenance program and is calibrated periodically. Measurements are made continuously registering hourly averages of concentrations in $\mu g\ m^{-3}$.

The statistical analysis was performed only when valid data had been obtained on at least 90% of the maximum number of SO_2 measurements possible for each year (EC Directive, 1999). The exception was for Custóias, in 2000 the efficiency was only 75%. In the last three years, the efficiency of the SO_2 measurements in the

urban/traffic1 site was not sufficient for the statistical treatment. For this reason we selected the urban/traffic2 monitoring site that started measurements in September of 2000. Both areas have similar characteristics.

RESULTS

Statistical analysis of SO_2 data from 1999 to 2003, verified that daily SO_2 concentrations observed in the industrial area are much higher than those of the urban and rural areas (Figure 2). This reflects the influence of the industrial sector, which includes one oil refinery with a petrochemical plant and other industries.

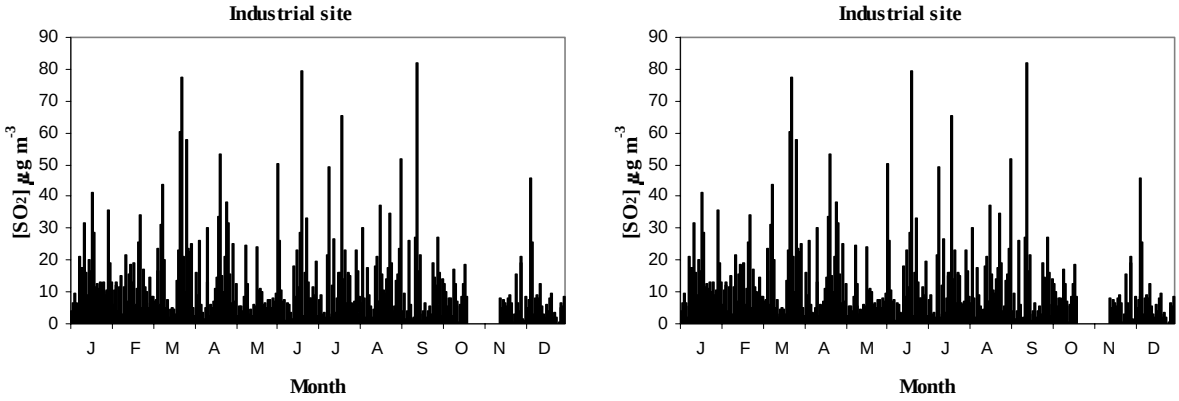


Figure 2. SO_2 daily averages for the industrial and rural sites in 2002.

The behaviour of SO_2 in Oporto-MA was analysed based on the SO_2 exceedances according to EC legislation (EC Directive, 1999). Thus the recommended SO_2 hourly limit value for protection of human health (350 µg m^{-3} not exceeded more than 24 times per year) was not surpassed during the analysed years (Table 1). The hourly limit value in the industrial area was reached 15 times in 1999, once in 2002 and was not reached in 2003, showing a significant decrease in the number of high values during the study period.

The daily limit value for protection of human health (125 µg m^{-3} not exceeded more than 3 times per year) was exceeded once in 1999 in the industrial site and not subsequently at any site (Table 1).

Table 1. Exceedances of SO_2 hourly and daily limits for protection of human health.

Site	Number of hours [1]					Number of days [2]				
	1999	2000	2001	2002	2003	1999	2000	2001	2002	2003
Urban/traffic[1]	4	0	*	*	*	0	0	*	*	-
Urban/traffic[2]	-	-	-	0	0	-	-	-	0	0
Industrial	15	9	3	1	0	**5**	0	0	0	0
Rural	2	0	2	1	0	0	0	0	0	0

* Not enough data for statistical treatment

[1] hourly limit (350 µg m^{-3}) not allowed more than 24 exceedances per year

[2] daily limit (125 µg m^{-3}) not allowed more than 3 exceedances per year

The annual averages of SO_2 concentrations for the consecutive years of analysis have decreased considerably for all the monitoring sites (Figure 3). For the industrial and rural sites the decrease between 1999 and 2002 was approximately 60%. For the industrial site between 2002 and 2003 the decrease was about 53%, which overall corresponds to a decrease of 81%. The rural site presented the lowest SO_2 annual concentrations and in the last three years, the values measured are similar to those found in other rural areas of Europe (WHO, 2000).

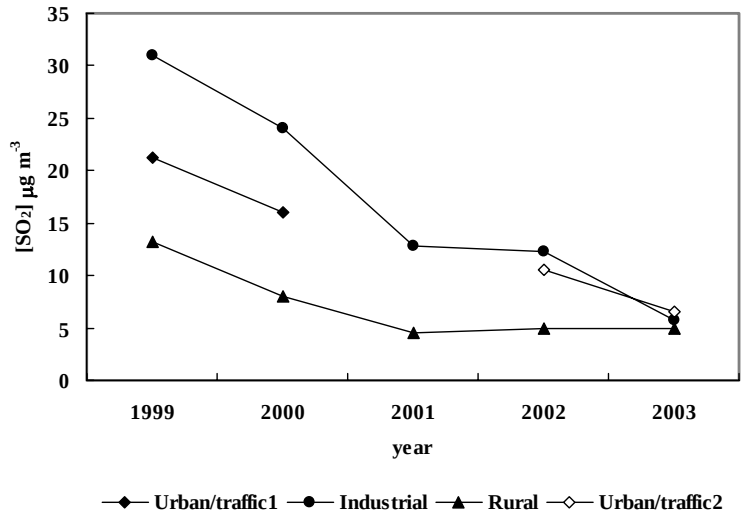


Figure 3. SO_2 annual average concentrations in Oporto-MA, 1999-2003.

The recommended SO_2 concentration annual limit value for protection of ecosystems (20 µg m^{-3}) was exceeded in 1999 both in the industrial and urban areas, with annual averages of 32.1 and 21.3 µg m^{-3} for the industrial and the urban/traffic sites, respectively. In 2000 the annual limit was exceeded only in the industrial area (23.9 µg m^{-3}). From 2001 to 2003 this limit was obeyed in all the monitoring sites. It should be emphasised that all the other sites of the Air Quality Monitoring Network of Oporto-MA, did comply with the European Directives for SO_2. Considering that the oil refinery is one of the main sources for the SO_2 emissions, it was possible to analyse the SO_2 emission trends in the Oporto-MA based on available data from the oil refinery. The emission values expressed both in tons and in kg/ton of refinery feed charge, were 18463 ton and 3.90 kg/ton in 1999; and 10631 ton and 2.7 kg/ton in 2003, (percentage of sulphur in the burn processual residue of 3.90 and 2.54%, in 1999 and in 2003, respectively). These estimated emission values were obtained by continuous measurements in the chimneys of the oil refinery. Besides those measurements, the refinery has an annual program of direct measurements in the chimneys to sustain the results attained by the continuous measurements. The refinery receives regular inspections by official certified entities to control the emission values. Since 2003 the legislation obliged the refinery to burn processual residue with less than 3% of sulphur.

The values presented show a significant decrease of the SO_2 emissions from 1999 to 2003. These data are consistent with the pronounced decrease of the SO_2 levels in ambient air at the industrial site and in all the sites of the network reflecting an improvement of the ambient air quality in Oporto-MA.

In contrast to the situation with SO_2, there was no clear tendency for a decrease in the PM_{10} concentrations observed for the period (Pereira et al., 2004). For the PM_{10} concentrations, the EC Directive considers two different implementation phases, the first one until 2005 and the second one until 2010, with limits each time more exigent (EC Directive, 1999). Thus, the comparison of the PM_{10} annual average with the corresponding limits (40 µg m^{-3} to be achieved before the end of the first phase of implementation, and 20 µg m^{-3} to be achieved before the end of the second phase) shows that only the rural site complied with the limit corresponding to the first phase of implementation in four of the years analysed (Figure 4). If the limit to be reached during the second phase of implementation is considered, it is clear that concentrations were considerably higher than the limit (1.5 to 2.5 times higher for the industrial site and 1.6 to 2.3 times higher for the rural). This analysis reveals that even in the rural areas not directly influenced by particle emissions there are high background concentrations that are related to the intra-regional transport of pollutants as mentioned above. It should be emphasized that none of the other sites of the Air Quality Monitoring Network of Oporto-MA, obeyed the European Directives for PM_{10}, which means that a general reduction of PM_{10} emissions is necessary to protect human health and the environment.

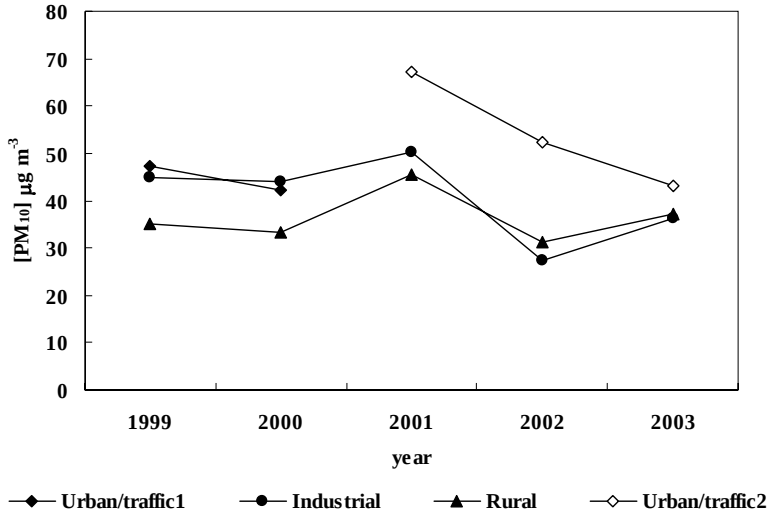


Figure 4. PM_{10} annual average concentrations in Oporto-MA, 1999-2003.

In Oporto-MA ozone also deserves special attention because the exceedances of the EC legislation (EC Directive, 2002) are frequent during the warmer months, April to October (Pereira et al., 2004). For the five years of analysis, the threshold for public notification that short-term exposure can cause health risks among groups particularly sensitive (180 µg m^{-3} for hourly average) was surpassed 15 times in the industrial site, 26 times in the rural site and just 2 times in 2003 for the urban/traffic2 site. During the same period, the threshold for public notification of health risks associated with short-term exposure of the population in general (240 µg m^{-3} for three consecutive hourly averages) was surpassed once in the industrial site and twice in the rural site. Therefore, frequent ozone episodes are affecting the air quality in suburban and other urban areas of the Oporto-MA.

DISCUSSION AND CONCLUSIONS

Based on the information presented here, SO_2 concentration levels in ambient air of Oporto-MA are clearly affected by the geographic location and proximity to emission sources.

Improvement in fuel quality, the application of emission control programmes and recent technological innovations in the past years are believed to be responsible for the significant decrease in the SO_2 concentrations and, consequently, the reduction in the number of health and environmental standards exceedances of the EC SO_2 limits in Oporto-MA.

Nevertheless, human health risks still persist particularly in industrialized areas. It is expected that SO_2 concentration levels in ambient air of Oporto-MA will continue to decrease in the coming years in line with the present European trends.

In conclusion, this analysis indicates reductions in SO_2 concentration levels in industrial, urban/traffic and rural areas. We suggest that "Auto-Oil" directives led to these reductions and hence improved the air quality of Oporto-MA. While improvements in air quality are evident for SO_2 it is important to consider that other pollutants, particularly PM_{10} and O_3, are also critical in the area and may deserve special attention. Policies based on the same principles should be enforced to help solve the problems related to emissions of these pollutants.

REFERENCES

Borrego, C., Barros, N., Gomes, P., Coutinho, M., Tchepel, O., Lopes, M., and Miranda, A.I. 2000. Application of the Air Quality Framework Directive to Portugal: A Strategic Evaluation. In J. Longhurst, D. Elson, H. Power (Eds.), *Air Quality Management*, pp. 1-39, WIT Press.

CEC. 2000. AOP II air-quality report. *Directorate-General for Regional Policies and Cohesion*, Brussels.

DGXI. 1996. Air quality report of the Auto-Oil Programme. *European Commission*, DGXI, Brussels.

Decreto Lei nº 104/2000, Ministério da Economia, Diário da República, 129, I-Série A.

Decreto Lei nº 281/2000, Ministério da Economia, Diário da República, 260, I-Série A.

Decreto Lei nº 111/2002, Ministério do Ambiente e do Ordenamento do Território, Diário da República, 89, I-Série A.

EC Directive 1996, Council Directive 96/62/EC, relating to evaluation and management of ambient air quality. OJ L 296, 55-63.

EC Directive 1998a. 98/69/EC of the European and Parliament and the Council relating to measures to be taken against air pollution by emissions from motor vehicles.

EC Directive 1998b, 98/70/EC of the European and Parliament and the Council relating to the quality of petrol and diesel fuels.

EC Directive 1999. Council Directive 99/30/EC, relating to limit values for sulphur dioxide, nitrogen dioxide and oxides of nitrogen, particulate matter and lead in ambient air. OJ L 163, 41-60.

EC Directive 2002, 2002/3/EC of the European Parliament and the Council relating to ozone in ambient air. OJ L 67, 14-30.

Larssen, S., Barret, K.J., Fiala, J., Goodwin, J., Hagen, L.O., Henriksen, J.F., Leeuw, F., and Tarrason, L. 2002. Air quality in Europe: state and trends 1990-99. Copenhagen: European Environment Agency.

Leeuw, F.A.A.M, Moussiopoulos, N., Bartonova, A., and Sahm, P. 2001. Air quality in larger cities in the European Union: a contribution to the Auto-Oil II programme. Copenhagen: European Environment Agency.

Pereira, M.C., Alvim-Ferraz, M.C.M., and Santos, R.C. 2004. Relevant Aspects of Air Quality in Oporto (Portugal): PM_{10} and O_3. Environmental Monitoring and Assessment. 1-19 (Prepublication 03/09/04).

World Health Organization (WHO). 2000. Air quality guidelines. Copenhagen: WHO Regional Office.

HEALTH IMPACT ASSESSMENT OF PM_{10} EXPOSURE IN THE CITY OF CAEN, FRANCE: IS ELIMINATING AIR POLLUTION PEAKS ENOUGH?

P. Glorennec, F. Monroux[1]

ABSTRACT

Air pollution in Caen, a city in northwestern France, comes mainly from motor vehicles. The aim of this study was to assess the public health impact of both acute (with immediate or short-term effects) and chronic (with long-term effects) exposure to PM_{10} (particulate matter <10 µm). The standard World Health Organisation (WHO) methodology for a Health Impact Assessment (HIA) was used to calculate the attributable deaths and hospital admissions. Population exposure was estimated from PM_{10} concentrations collected by the local air quality measurement network. The relative risks were modelled with exposure-risk functions established in epidemiologic studies in the general population. The APHEA-2 program, which combines European time-series studies, was used to assess effects from acute exposures and a meta-risk was calculated from cohort studies to assess the effects of chronic exposure. The health impact of chronic exposure from 1998 through 2002 was estimated at 168 (101-238) deaths. Acute exposure (relative to a baseline level of 10 µg/m^3) led to 26 (17-35) deaths and 43 (22-67) hospital admissions during this period. A 10% daily decrease in pollution would reduce the number of expected deaths from short-term exposure by 19%, while achieving compliance with European Union regulations (daily mean in 2010: 50 µg/m^3) would reduce them by less than 3%. Because the health impact of the pollution in Caen is due mainly to relatively moderate levels, reducing everyday pollution levels through long-term regulation would be more beneficial than avoiding pollution peaks.

[1] Ecole Nationale de la Santé Publique, Rennes Cedex, France.

INTRODUCTION

Methodological progress in epidemiologic studies since the 1990s (Bell et al., 2004; Goldberg et al., 2003) has made it possible to show that air pollution today, when industrial pollution has decreased but automobile pollution has increased, still affects population health.

The metropolitan area of Caen is home to 173 000 inhabitants, nearly all living and working in this area (INSEE, 1999). The quality of air in this city is influenced primarily by motor vehicle emissions (Fontell et al., 1997). In 1999 the short-term health impact of urban air pollution in Caen was assessed, on the basis of data from October 1997 through October 1998 (Glorennec et al., 2002). More recent literature includes epidemiologic studies estimating the long-term impact of pollution on mortality and articles on the respective advantages and limitations of short- and long-term health impact assessments (HIA) (McMichael et al., 1998; Quenel et al., 1999). In November 2000, the World Health Organisation released recommendations on this topic (WHO, 2001). This progress in epidemiology led public health professionals to consider whether HIAs that also cover long-term effects would be useful at local levels.

The goals of this study were thus to meet the regulatory obligations of the French Clean Air Act (Journal officiel de la Republique Francaise, 1997) to assess the health effects of air pollution and to provide information about its local effects to the public. Accordingly, we sought to assess the public health impact of both acute (with short-term effects) and chronic (with long-term effects) exposure to PM_{10}. The presentation of epidemiologic results cannot always be easily interpreted by local decision-makers, who nonetheless manage French transportation policy; the interface between science and policy must be strengthened. Accordingly, in order to help improve the effectiveness of public policy-making, we sought to compare the health impact of two types of strategies: avoidance of pollution peaks and decreasing the mean pollution level.

Methods

Health impact was assessed by calculating the number of attributable cases, according to the method recommended by WHO (1999). The proportion of cases attributable to a causal risk factor is described by (Glorennec and Quénel, 1999):

$$Pa = \frac{f\,(RR - 1)}{1 + f\,(RR - 1)} \text{ or } \frac{RR - 1}{RR} \text{ if } f = 1$$

where
Pa: proportion or fraction of attributable cases;
RR: relative risk;
f: prevalence of exposure.
$Na = Pa * N$

where
Na: number of cases attributable to air pollution; and
N: total number of cases (incidence).

The endpoints we studied were effects that have been demonstrated in epidemiologic studies conducted in the general population at comparable exposure levels and for which local incidence data were available. They included mortality for all causes except accidents (ICD S00-X59) and hospital admissions for respiratory (ICD J00-J99) and cardiovascular (ICD I00-I99) causes.

We used exposure-risk functions reported by time-series epidemiologic studies to examine immediate effects of PM_{10} exposure and cohort studies to look at the long-term effects. The estimators we used (cf. Table 1) are those recommended by the French Institute for Public Health Surveillance (Pascal and Cassadou, 2003) and are based on studies from the European APHEA-2 program (Atkinson et al., 2001; Künzli et al., 2000). The relative risks were calculated from these exposure-risk relations.

Table 1: Relative risks (RR) for a 10 µg/m^3 increase of PM_{10}.

Pollutant	Indicator	Lower	Mid	Upper	source
PM_{10}	Short term mortality	1.004	1.006	1.008	APHEA2
	Short term respiratory morbidity > 65 years	1.006	1.009	1.013	APHEA2
	Short term cardiac morbidity	1.002	1.005	1.008	APHEA2
	Short term cardiac morbidity > 65 years	1.004	1.007	1.01	APHEA2
	Short term mortality, adults > 30 years	1.026	1.043	1.061	Kunzli 2000

The exposure levels were measured by AirCOM, the local air quality measurement network. Daily exposure levels (Table 2) were based on the arithmetic mean of the daily data for the two available PM_{10} sensors from 1 April 1998 through 31 March 2002.

Table 2. Estimated outdoor PM_{10} exposure (µg/m^3). Caen 1998-2002.

	PM_{10} (µg/m^3)
Min	2
P 5	7
P 25	12
P 50	17
P 75	25
P 95	42
Max	87
Daily mean	19
Standard deviation	11.4
% missing values	1%

CepiDC provided the data for mortality from all causes except accidents (ICD 10 = S00-X59) for adults (>30 years) residing in the study area: 562 deaths during winter months and 526 in summer (Caen April 1998-April 2002). The hospital admissions data came from the medical information department of the Caen University Hospital Centre (Table 3).

Table 3: Number of hospital admissions for cardiovascular (ICD I00 to I99) and respiratory (ICD J00 to J99) disorders at the Caen UHC.

	I00 to I99 15-64 years	**≥ 65 years**	**Total**	**J00 to J99 15-64 years**	**≥ 65 years**	**Total**
Total study period	2033	4841	6899	1559	2218	5167
Annual mean	407	968	1380	312	444	1033
Total summer period	934	2381	3329	731	969	2161
Mean annual summer	187	476	666	146	194	432
Total winter period	1099	2460	3570	828	1249	3006
Mean annual winter	220	492	714	166	250	601

We used Excel©-based EIS-PA software (Franke, 2003) to calculate health impact, relative to a reference level of 10 µg/m^3, considered as the "no pollution" (or background) level. Daily time series of pollution levels make it possible to calculate the impact of each exposure level.

For acute exposures, we assessed:

- the health impact of a 10% reduction in daily pollution levels, which would correspond to "moderate" improvement in air pollution,
- daily compliance with the EU recommended daily mean for 2010 (50 µg/m^3).

Results

Table 4 summarises the impact of the air pollution over the study period.

Table 4: Health impact of outdoor PM_{10}. Caen 1998-2002.

	Health effect	Attributable cases (95% CI)
	Mortality	26 (17-35)
Short-term	Hospital admissions, respiratory causes	16 (11-23)
	Hospital admissions, cardiovascular causes	27 (11-44)
Long-term	Mortality	168 (101-238)

As shown in Figure 1, if we consider the effects due to short-term exposure, a daily reduction of 10% in the pollution level would reduce the number of deaths by 19%, while daily compliance with the European standard of 50 µg/m^3 would yield an improvement of only 2.5 %.

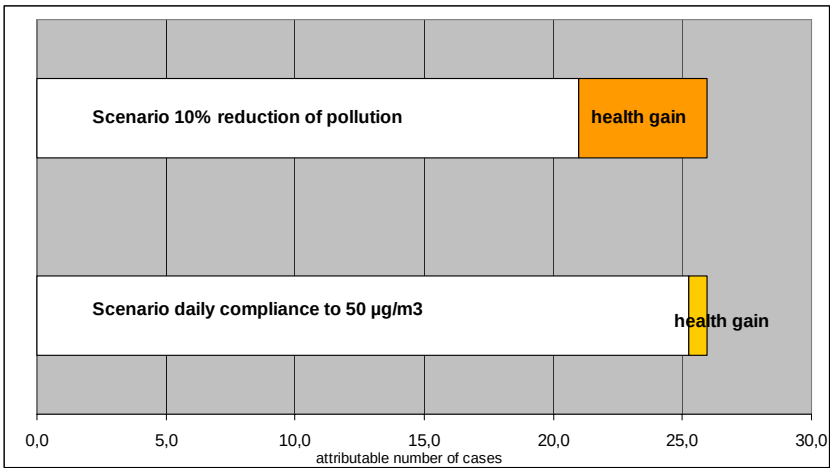


Figure 1. Health effectiveness of particulate air pollution policy scenarios, 1998-2002.

Figure 2 illustrates the health impact of different levels of exposure to air pollution. Approximately two thirds of this effect comes from pollution levels less than 40 µg/m^3.

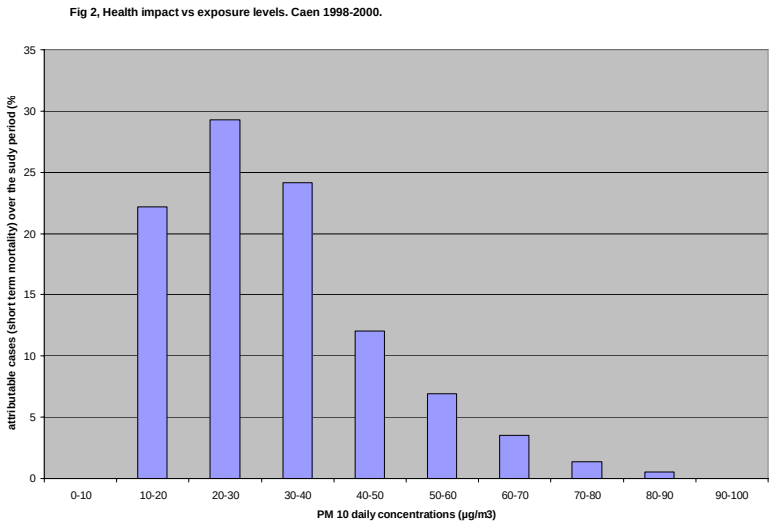


Figure 2. Health impact vs exposure levels. Caen 1998-2000.

DISCUSSION AND CONCLUSIONS

The concept of attributable risk presupposes that the relation between air pollution and health is causal. That is, the risk attributable to a risk factor can be calculated only if there is serious evidence of causality between exposure and disease (Coste and Spira, 1991). This causality is increasingly evident for particles (Dab et al., 2001). Because of the correlations between different air pollutants, however, this concept of attributable risk must be understood to apply to exposure to air pollution as a whole. We approach this complex chemical mixture through pollution indicators, here PM_{10}.

In the field of urban air pollution, exposure-risk functions are based on observations in humans at low exposure levels; extrapolation from animals to humans or from high to low doses is not required. On the other hand, we applied relations established elsewhere to our study area, although we cannot be certain of the validity of this procedure. That is, the same pollution indicator may be a tracer of different pollution profiles, especially, for example, if the automobile fleets in the two places differ (France has a larger proportion of diesel engines than many other countries). The exposure-risk functions we used for the short-term effects come from the APHEA-2 (Katsouyanni et al., 2001) program and are based on observations in several cities of western Europe. They should thus be robust and applicable to France. For the long-term effects, the same functions used by Künzli et al. (2000) from US cohort studies (Dockery et al., 1993; Pope et al., 1995) were applied. These studies estimated exposure based on an annual mean level of PM_{10}, data which is also available for Caen. We chose not to use the study by Hoek et al. (2002), although it was European, because its exposure indicator was constructed in conditions different from ours: it used a mix of regional and local exposure data. We also chose not to use the most recent and most powerful study, by Pope et al. (2002), because its exposure-risk relations were based upon $PM_{2.5}$ and not PM_{10}. On the other hand, we used the risk function from Pope et al. (2002) and a $PM_{2.5}/PM_{10}$ coefficient of 0.7 to analyse the sensitivity of our results: this yielded 109 (27-218) deaths for long-term exposure in the study area during the study period, compared with 168 (101-238) with the risk function we used initially.

For morbidity, we took into account only the respiratory effects leading to hospitalisation, even though it is likely that only a small fraction of the population with pollution-related respiratory problems go to the hospital; the impact of pollution on morbidity is therefore underestimated. Künzli et al. (2000), for example, calculated that there are approximately 135 000 asthma attacks annually in France, but “only” 19 000 hospital admissions attributable to air pollution. Local asthma prevalence data are required to assess the impact of pollution on such health endpoints as asthma attacks, and local authorities must plan to conduct asthma surveys if they want this relevant impact to be assessed.

Use of only two PM sensors to assess exposure undoubtedly results in less than optimally reliable estimates. Nevertheless these two sensors are located in areas representative of the study area, and wider-scale air quality measurements were taken previously to choose the sensor locations.

The baseline exposure level is an especially sensitive parameter in calculating impact. Choosing a lower baseline (e.g., 7.5 $\mu g/m^3$), as Künzli et al. (2000) did, increases the impact of the same pollution measurements. With this lower baseline, there are 32 (21-42) short-term deaths, compared with 26 (17-35) with a 10 $\mu g/m^3$ baseline.

Because of these limitations, the results should be interpreted as orders of magnitude. These results are not sufficiently accurate to measure the precise impact of particulate air pollution in Caen nor to predict the exact impact of policies. They are, however, accurate enough to yield general conclusions that can be used in setting local air quality policies.

The results show that air pollution, even at moderate levels below the current recommended standards, affects the population's health. Moreover, although the relative risks associated with air pollution are modest, the large fraction of exposed persons leads to a substantial collective impact.

This impact is expressed by effects that are simultaneously short- and long-term. The long-term effects are, as our results suggest (cf. Table 4), higher in terms of mortality; they exceed the simple addition of the short-term effects (Künzli et al., 2001). Long-term exposure promotes the development of chronic diseases (Künzli et al., 2001), including lung cancer (Pope et al., 2002; Cohen, 2000). The short-term effects are often described as affecting specific population groups, children or the elderly, often with preexisting diseases (Filleul et al., 2003; Gouveia and Fletcher, 2000). Nevertheless, those affected were not often terminally ill (Schwartz, 1994;

Brunekreef and Holgate, 2002). Estimates of reduced life expectancy remain uncertain; Finkelstein recently showed (Finkelstein et al., 2004) that living near a heavily-trafficked road reduces life expectancy by 2.5 years.

Because these long-term effects are due to exposure over long periods of time, improvement of the air quality will only slowly produce long-term benefits (Leksell and Rabl, 2001). Accordingly, for example, the effects in the study by Pope et al. (2002) are associated with exposure averaged over 16 years: a short-term diminution of x% will thus take 16 years to produce the totality of its expected effects.

Episodes of high pollution often monopolize the debate about air pollution. The underlying idea is that if their consequences were mastered, the air pollution issue would be resolved. That is simply not true (Dab et al., 1998), even when we consider only the effects of acute exposures. Although "high pollution" days have the greatest daily impact, their low frequency (cf. Figure 2) means that they play a limited role compared with the consequences of moderate air quality throughout an entire year. As we showed, daily compliance with the European standard of 50 $\mu g/m^3$ would yield an improvement of only 2.5%. Even reducing PM_{10} levels above 40 $\mu g/m^3$ (EU regulatory level for an annual mean in 2005), by limiting traffic when bad dispersion conditions are forecast, would reduce the annual air pollution health burden by only 6%. Moreover, pollution peaks do not contribute much to long-term exposure, the impact of which is greater than acute exposure. In practice, this means that a local risk management policy aimed exclusively at avoiding exposures exceeding regulatory levels will have only a marginal impact on public health.

REFERENCES

Atkinson, R.W., Anderson, H.R., Sunyer, J., Ayres, J., Baccini, M., Vonk, J.M. Boumghar, A., Forastiere, F., Forsberg, B., Touloumi, G., Schwartz, J., and Katsouyanni, K. 2001. Acute effects of particulate air pollution on respiratory admissions: Results from APHEA 2 project. Air Pollution and Health: A European Approach. *Am. J. Respir. Crit. Care Med*. 164:1860-1866.

Bell, M.L., Samet, J.M., and Dominici, F. 2004. Time-series studies of particulate matter. *Annu. Rev. Publ. Health* 25:247-280.

Brunekreef, B., and Holgate, S.T. 2002. Air pollution and health. *Lancet* 360:1233-1242.

Cohen, A.J. 2000. Outdoor air pollution and lung cancer. *Environ. Health Perspect.* 108:743-750.

Coste, J., and Spira, A. 1991. Proportion of cases attributable to public health: definition(s), estimation(s) and interpretation. *Rev. Epidemiol. Sante* 39:399-411.

Dab, W., Le Moullec, Y., Dusseux, E., Medina, S., and Le Tertre, A. 1998. Les effets sanitaires des pics de pollution hivernaux. Extrapol 12 *Rev. Pollution Atmosphér*. 156:I-III.

Dab, W., Segala, C., Dor, F., Festy, B., Lameloise, P., Le Moullec, Y., LeTertre, A., Medina, S., Quenel, P., Wallaert, B., and Zmirou, D. 2001. Air pollution and health: correlation or causality? The case of the relationship between exposure to particles and cardiopulmonary mortality. *J. Air Waste Manage. Assoc*. 51:220-235.

Dockery, D.W., Pope, C.A, III, Xu, X., Spengler, J.D., Ware, J.H., Fay, M.E., Ferris, B.G. Jr., and Speizer, F.E. 1993. An association between air pollution and mortality in six U.S. cities. *N. Engl. J. Med.* 329:1753-1759.

Filleul, L., Medina, S., and Cassadou, S. 2003. Urban particulate air pollution: from epidemiology to health impact in public health. *Rev. Epidemiol. Sante* 51:527-452.

Finkelstein, M.M., Jerrett, M., and Sears, M.R. 2004. Traffic air pollution and mortality rate advancement periods. *Am. J. Epidemiol*. 160:173-177.

Fontelle, J.P., Chang, J.P., Audoux, N., Levy, C., and Rivière, E. 1997. Inventaires d'émissions dans l'atmosphère dans la cadre des plans régionaux pour la qualité de l'air. Centre Inter-Professionnel d'Etude de la Pollution Atmosphérique (CITEPA).

Franke, F. 2003. EIS-PA. (Evaluation d'impact Sanitaire-Pollution Atmospherique). St Maurice, France: Institut de Veille Sanitaire.

Glorennec, P., Nourry, L., and Quenel, P. 2002. Impact sanitaire à court terme de la pollution atmosphérique urbaine dans le Nord-ouest de la France. *Environ. Risques et Santé* 1:157-163.

Goldberg, M.S., Burnett, R.T., and Stieb, D. 2003. A review of time-series studies used to evaluate the short-term effects of air pollution on human health. *Rev. Environ. Health* 18:269-303.

Gouveia, N., and Fletcher, T. 2000. Time series analysis of air pollution and mortality: effects by cause, age and socioeconomic status. *J. Epidemiol. Commun. Health* 54:750-755.

Hoek, G., Brunekreef, B., Goldbohm, S., Fischer, P., and van den Brandt, P.A. 2002. Association between mortality and indicators of traffic-related air pollution in the Netherlands: a cohort study. *Lancet* 360:1203-1209.

INSEE (Institut National de la Statistique et des Etudes Economiques) 1999. Recensement général de la population. INSEE Paris.

Journal officiel de la République Française 1997. Loi n96-1236 du 30 décembre 1996 sur l'air et l'utilisation rationnelle de l'énergie. 1-1-1997.

Katsouyanni, K., Touloumi, G., Samoli, E., Gryparis, A., Le Tertre, A., Monopolis, Y., Rossi, G., Zmirou, D., Ballester, F., Boumghar, A., Anderson, H.R., Wojtyniak, B., Paldy, A., Braunstein, R., Pekkanen, J., Schindler, Ch., and Schwartz, J. 2001. Confounding and effect modification in the short-term effects of ambient particles on total mortality: results from 29 European cities within the APHEA2 project. *Epidemiology* 12:521-531.

Krzyzanowski, M. 1997. Methods for assessing the extent of exposure and effects of air pollution. *Occup. Environ. Med*. 54:145-151.

Künzli, N., Kaiser, R., Medina, S., Studnicka, M., Chanel, O., Filliger, P. Herry, M., Horak, F. Jr., Puybonnieux-Texier, V., Quenel, P., Schneider, J., Seethader, J., Vergnaud, J-C., and Sommer, H. 2000. Public-health impact of outdoor and traffic-related air pollution: A European assessment. *Lancet* 356:795-801.

Künzli, N., Medina, S., Kaiser, R., Quenel, P., Horak, F. Jr., and Studnicka, M. 2001. Assessment of deaths attributable to air pollution: should we use risk estimates based on time series or on cohort studies? *Am. J. Epidemiol*. 153:1050-1055.

Leksell, I., and Rabl, A. 2001. Air pollution and mortality: quantification and valuation of years of life lost. *Risk Anal*. 21:843-857.

McMichael, A.J., Anderson, H.R., Brunekreef, B., and Cohen, A.J. 1998. Inappropriate use of daily mortality analyses to estimate longer-term mortality effects of air pollution. *Int. J. Epidemiol*. 27:450-453.

Pascal, L., and Cassadou, S. 2003. Evaluation de l'impact sanitaire de la pollution atmosphérique urbaine: Actualisation du guide méthodologique. Institut de Veille Sanitaire pp. 1-38.

Pope, C.A. III, Burnett, R.T., Thun, M.J., Calle, E.E., Krewski, D., Ito, K., and Thurston, G.D. 2002. Lung cancer, cardiopulmonary mortality, and long-term exposure to fine particulate air pollution. *J. Am. Med. Assoc*. 287:1132-1141.

Pope, C.A. III, Thun, M.J., Namboodiri, M.M., Dockery, D.W., Evans, J.S., Speizer, F.E., and Heath, C.W. 1995. Particulate air pollution as a predictor of mortality in a prospective study of U.S. adults. *Am. J. Respir. Crit. Care Med*. 151:669-674.

Quenel, P., Zmirou, D., Dab, W., Le Tertre, A., and Medina, S. 1999. Premature deaths and long-term mortality effects of air pollution. *Int. J. Epidemiol*. 28:362.

Schwartz, J. 1994. What are people dying of on high air pollution days? *Environ. Res*. 64:26-35.

Van Dingenen, R., Raes, F., and Putaud, J.P. 2004. A European aerosol phenomenology - 1: physical characteristics of particulate matter at kerbside, urban, rural and background sites in Europe. *Atmos. Environ*. 38:2561-2577.

World Health Organisation (WHO) European Center for Environment and Health 2001. Quantification of the health effects of exposure to air pollution. WHO.

World Health Organisation (WHO) 1999. Strategies of ambient air quality monitoring for health impact assessment. WHO Regional publications ES. WHO regional office for Europe, Copenhagen.

PART V - SCIENCE-POLICY ISSUES

FINE PARTICLES: FROM SCIENTIFIC UNCERTAINTY TO POLICY STRATEGY

Rob Maas[1]

ABSTRACT

Gaps in our knowledge should not be a reason for not making policy decisions. How can we define robust policy strategies given the various uncertainties in the sources, exposure and causes of health effects? Which uncertainties are most important? What are its policy implications? This contribution will describe policy strategies for particulate matter (PM) abatement that are consistent with certain sets of assumptions, as well as the risks that are associated with such strategies. What is an optimal strategy assuming that the fine particles ($PM_{2.5}$) are the main cause of health effects? And what would be the 'mistake' of such a strategy, if 'in the end' PM_{10} or carbonaceous particles prove to be the 'real' cause? How can we make the policy strategy more robust and minimise its financial risks and health risks. The contribution will describe a systematic way of dealing with the knowledge gaps in the policy process.

[1] Netherlands Environmental Assessment Agency - RIVM/MNP, Chair UNECE Task Force Integrated Assessment Modelling

UNCERTAINTY AND INCERTITUDE

Within the source-effect chain of particulate matter one can find large uncertainties almost everywhere. Emission data are not very reliable, if compared with for instance SO_2 or CO_2. Emission data for PM_{10} and $PM_{2.5}$ seem to be incomplete. Currently we can only partly explain the background concentrations from these primary emissions and from the production of secondary particles in the atmosphere. Anthropogenic PM-emissions might be underestimated, e.g. the emissions from wood burning, from the aberration of tires and brakes, or the re-suspension of particles. Moreover, a substantial amount of the primary and secondary natural emissions are not taken into account (such as Sahara dust and biogenic organic particles), and the transcontinental influx might be underestimated. We can explain the formation of secondary inorganic particles from SO_2, NO_x and NH_3 emissions relatively well, but our understanding of the secondary organic particles is still poor. We also know little about the water-content of particles. Furthermore, large scale models can only partly estimate the actual exposure of the population in the streets and in buildings.

All the uncertainties mentioned above seem to be mainly caused by the lack of (reliable) data and can basically be solved by better measurements. With statistical techniques (such as error propagation) we can estimate the likelihood that an abatement measure will reduce human exposure based on our estimates of the uncertainty margins in the input data.

However the biggest challenge in the PM-case is our lack in knowledge about the fraction that is really causing the problem. Or is it too much knowledge, because there are several 'competing' theories to frame the problem? This incertitude cannot be treated in the same way as the uncertainties mentioned above (Hisshemöller and Hoppe, 1996; Van der Sluijs et al, 2003; Maas 2003). We cannot attach a probability to either one of the other theories. And as long as one of these theories is not falsified, there remains a fair chance that such a theory might be correct. The real challenge in the PM-debate is to acknowledge that there is not just one best theory, but that there are several ways to describe 'reality' and to deal with this incertitude in the policy process.

CHANGING REALITIES

If we look back and see how the science has developed over the past decade, we can conclude that some modesty about our current knowledge is appropriate. In the 1950s we thought that Black Smoke was causing lung diseases and we reduced its emissions by over 80%. In the 1970s it was sulphur, which by now is also reduced by more than 80%. In the 1990s PM_{10} was identified as the cause. Current policies aim at a reduction of PM_{10} by more than 80% in the EU. But still if we look at health indicators like lung cancer and prevalence of Chronic Obstructive Pulmonary Disease (COPD) we do not see much change. Other determinants like 'lifestyle' seem to be more dominant than air pollution (Murray and Lopez, 1996). In the past decades in the Netherlands, for example, lung cancer and COPD per 1000 women increased due to an increase in smoking (National Compass Public Health, 2005). A much closer look into the available epidemiological data is required to detect an association between PM-exposure and health. Currently - based on American studies (e.g. Pope et al. 2002) – the association between $PM_{2.5}$ exposure and health seems to be stronger than between PM_{10} and health, and although an effect of PM_{10} is not completely ruled out, $PM_{2.5}$ seems to replace PM_{10} as the focus of the policy makers' attention (Melcher et al, 2002) . In Europe this finding is not yet confirmed. $PM_{2.5}$ and PM_{10} are part of the conventional theory, although toxicological explanations behind its damaging effects are hypothetical (Buringh and Opperhuizen, 2002a, b).

Relatively little epidemiological research funds are focused on the alternative views that for instance specific PM-compounds such as carbonaceous particles (organic carbon, elementary carbon, PAH) could be causing the problem (WHO 2004; Kameda et al., 2005), or certain heavy metals, or the ultrafine particles. Incidentally for some of those PM-species more toxicological evidence is available than for $PM_{2.5}$ or PM_{10}! So the PMx case cannot be excluded either. Moreover strong associations can also be found between cardiopulmonary diseases and for instance traffic noise (Kempen et al., 2002), the quality of housing and the diet of low income families (Eschenroeder and Norris, 2003). It cannot be ruled out that health effects are caused by an accumulation of causes in low income neighbourhoods close to highways of which PM-exposure and NO_2-exposure are a only a minor part.

Do we know for sure that $PM_{2.5}$ is the final answer? Science should not suggest more certainty by selecting only one of the possible explanations, despite that correlations show that such a theory is (a bit) more likely. If scientists cannot falsify a competing theory, policy makers have a problem. They have to deal with the possibility that one of the theories is correct, but they cannot know which one (Jasanoff, 1990; Slovic, 1999). At least not before say 2010 or 2020. They could of course gamble on one theory, but this might be a waste of money (despite the advanced monetarised benefit estimates that go with the theory). How can policy makers minimise the risk of wasting money? How can they minimise health risks? Incertitude is no reason for political lethargy! Especially not as the health effects that occur are real.

DIFFERENT REALITIES REQUIRE DIFFERENT POLICY STRATEGIES

For the remainder of this analysis I distinguish four 'stylised' ways to define the PM-problem or four different perspectives on how reality works. The four theoretical 'families' are:

- '$PM_{2.5}$' – focus on transboundary air pollution and secondary inorganic particles
- 'PM_{10}' – a large part will be $PM_{2.5}$ but also the abatement of primary emissions of coarse particles becomes part of the strategy
- 'PMx' – focus on the traffic related carbonaceous particles and ultrafines
- 'No PM' – focus on living conditions in low income neighbourhoods

Each perspective has its own policy strategy (see Table 1).

Table 1: Alternative policy strategies and approaches.

Strategy	**Policy Approach**
Strategy 1: $PM_{2.5}$ is the problem	Focus on secondary particles from SO_2. NO_x, NH_3, VOC and on primary $PM_{2.5}$. Define cost-optimised national emission ceilings for these substances. Take into account the synergy with ozone and NO_2-exposure and the acidification and eutrophication problem.
Strategy 2: PM_{10} is the problem	Focus on cost-effective measures for primary PM sources that contribute most to local exposure, e.g. dust from building industry, cement, wood burning. A transboundary strategy is less important, only EU-harmonisation of technical standards in order to prevent 'false' competition at the expense of the health of the local citizens.
Strategy 3: Specific traffic related species are the problem (PMx)	Apply stricter international standards for vehicles, ships and refineries (focussed on elemental and organic carbon (EC/OC), heavy metals and primary ultrafines), substitute diesel in captive fleets of urban areas by zero emission vehicles, close densely populated streets for heavy traffic, encourage the use of public transport.
Strategy 4: It is mainly a socio-economic problem (no PM)	Improve living conditions and economic opportunities in poor neighbourhoods (prevent further segregation), circulate heavy traffic around (or under) those neighbourhoods. Design neighbourhoods in such a way that car traffic is minimised. Only reduce those air pollutants that also cause other problems like acidification, eutrophication and ozone formation.

IN SEARCH OF ROBUSTNESS

If we look at the different strategies, it seems hard to find a way out of the policy dilemma. But there is some light at end of the tunnel:

a. we can identify measures that are *'no-regret'*, i.e. relatively cheap measures that the can be part of each strategy (although they are not always the most effective option);
b. we can identify measures with large *co-benefits* for other environmental or health related problems;
c. we can identify potential *'regret'* measures, i.e. measures that can worsen the problem if seen from another perspective, e.g. when a trade-off exists between different species of PM, or between other policy fields and the PM-problem.

Measures that are good from all perspectives are for instance measures to reduce car traffic in living areas, either via traffic circulation schemes, the design of communities and the stimulation of public transport. Also measures like cleaner fuels and improvement of the energy efficiency are cost-effective and robust, as well as a ban on tobacco, wood burning and barbecues. Some of these measures are cost-effective but limit the freedom of individual and might thus raise political resistance.

Measures with high co-benefits are for instance the reduction of carbonaceous particles, because they also contribute to the greenhouse effect. Also the reduction of secondary particles (via further reduction of SO_2, NH_3, NO_x and VOC) entails co-benefits because this reduces acidification, eutrophication, ozone formation and NO_2-exposure.

All particle abatement measures contribute to improvement of the visibility, but it remains unclear how much this co-benefit is worth in the European policy arena.

'Trade-off' measures that are good from one perspective but bad from another perspective are, for instance, technical PM-measures that increase the emissions of ultrafine particles. There are also measures that are part of the greenhouse strategy that would be bad for PM-exposure, such as a shift from petrol to diesel, or the promotion of biomass and waste burning (if not accompanied with strict emission standards).

After scrutinising the different measures that can be part of a robust strategy, still a number of measures will remain that are really gambles. For these measures the policy maker has two options: 1. don't select them and take the risk that harmful health effects will remain to occur or: 2. select them (all) and take the risk of wasting money.

For these measures science can help to quantify the risks in terms of health and money by applying a so-called paradox approach: first select those measures that are part of the optimal solution within a certain perspective (so-called utopian worlds), but then look at the effectiveness of such measures when applied in another perspective (the so-called dystopian worlds). Via the confrontation of perspectives the maximum mistake can be quantified, and this might help the policy maker in finding the right balance between environmental and economic precaution (see Figure 1).

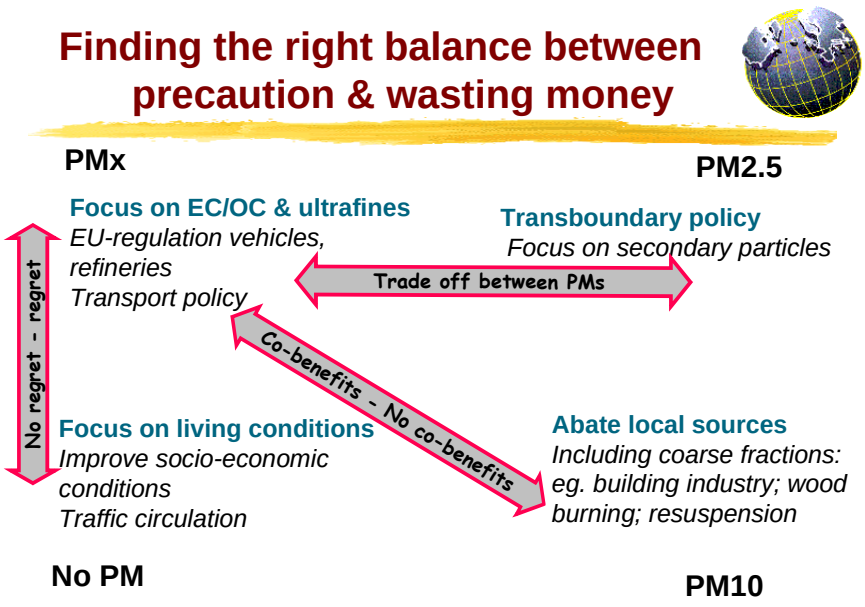


Figure 1: Finding the right balance between precaution and wasting money.

In order to minimize possible mistakes, politicians can combine different strategies and not base their decisions on one theory alone. But in the end they have to decide whether they want to be on the safe side and against what price.

REFERENCES

Buringh, E., and Opperhuizen, A., 2002a. On health risks of ambient PM in the Netherlands. Executive Summary report. RIVM, ISBN 90-6960-100-1.

Buringh, E., and Opperhuizen A. (Eds), 2002b. On health risks of ambient PM in the Netherlands. Full report. RIVM report 650010032.

Eschenroeder, A., and Norris, G. 2003. Should socioeconomic health effects be included in risk assessments? *Environ. Sci.* 1:27-58.

Hisschemöller, M., and Hoppe, R. 1996. Coping with Intractable Controversies: The case for problem structuring in policy design and analysis. *Knowl. Policy* 8:40–60.

Jasanoff, S., 1990. The Fifth Branch: Science Advisers as Policymakers. Cambridge, MA: Harvard University Press.

Kameda, Y., Shirai J., Komai, T., Nakanishi, J., and Masunga S. 2005. Atmospheric polycyclic aromatic hydrocarbons: size distribution, estimation of their risk and their depositions to the human respiratory tract. *Sci. Total Environ.* 340: 71-80.

Kempen, E.E.M.M., van Kruize, H., Boshuizen, H.C., Ameling, C.B., Staatsen, B.A., de Hollander, A.E.M. 2002. The association between noise exposure and blood pressure and ischemic heart disease: a meta-analysis. *Environ. Health Perspect.* 110:307-17.

Maas, R., 2003. Are we sure? Some thoughts about uncertainty treatment in integrated assessment. ASTA-yearbook. IVL Swedish Environmental Research Institute.

Mechler, R., Amann, M., and Schöpp, W. 2002. A Methodology to Estimate Changes in Statistical Life Expectancy Due to the Control of Particulate Matter Air Pollution. IR-02-035. International Institute for Applied Systems Analysis: Laxenburg, Austria.

Murray, C.J., and Lopez, A.D., 1996. The Global Burden of Disease, a Comprehensive Assessment of Mortality and Disability from Disease, Injury and Risk Factors in 1990 and Projected to 2020. Harvard University Press.

National Compass Public Health (in Dutch) 2005. Development of asthma and COPD in The Netherlands: http://www.rivm.nl/vtv/object_document/o1810n18082.html.

Pope, C.A., Burnett, R., Thun, M.J., Calle, E.E., Krewski, D., Ito, K., and Thurston, G.D. 2002. Lung cancer, cardiopulmonary mortality and long-term exposure to fine particulate air pollution. *J. Am. Med. Assoc.* 287:1132-1141.

Slovic, P., 1999. Trust, emotion, sex, politics, and science: surveying the risk assessment battlefield. *Risk Anal.* 19:689-701.

Van der Sluijs, J.P., Risbey, J.S., Kloprogge, P., Ravetz, J.R., Funtowicz, S.O., Quintana, S.C., Guimar aes Pereira, A., De Marchi, B., Petersen, A.C., Janssen, P.H.M., Hoppe, R., and Huijs, S.W.F. 2003. RIVM/MNP Guidance for Uncertainty Assessment and Communication: Detailed Guidance. Utrecht: Utrecht University.

WHO 2004. Health Effects of Air Pollution - Results from the WHO Project Systematic Review of Health Aspects of Air Pollution in Europe. Report E83080. Copenhagen: WHO Regional Office for Europe.

CLEAN AIR STRATEGY: AN ENGO PERSPECTIVE ON THE SCIENCE-POLICY INTERFACE

Max K. Wallis[1]

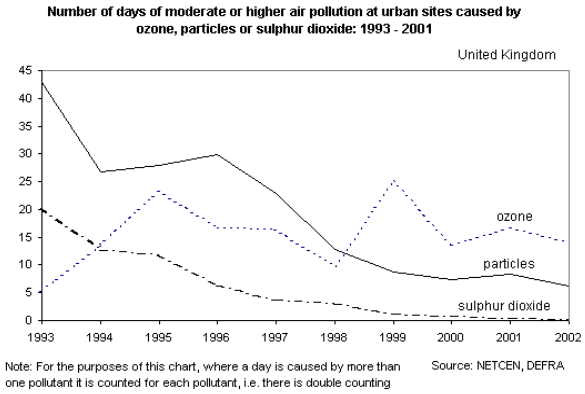
ABSTRACT

Environmental Non-Government Organisations (ENGOs) in Europe have been hardly involved in AIRNET, in part because the technicalities deter access. But ENGOs should be seen as representing the public as well as acting as stakeholders in their own right. They can be intimately involved in assessing the public information, as in the UK example criticised below. They can spotlight an issue like ultrafine particulate pollution that is being avoided for reasons of institutional inertia and special interests. For risk analysis and integrating a precautionary approach, ENGO participation within a stakeholder process is vital. Policies designed to combat air pollutants have to change profoundly how people live, travel and work (Maynard et al., 2003), yet policy-makers tend to duck the hard choices. Winning the public to make such changes requires enrolling the major campaigning ENGOs as allies, in the context of effective stakeholder communication and accountability in public information and policy setting.

[1] Cyfeillion y Ddaear, Wales UK

PUBLIC INFORMATION ON AIR POLLUTION

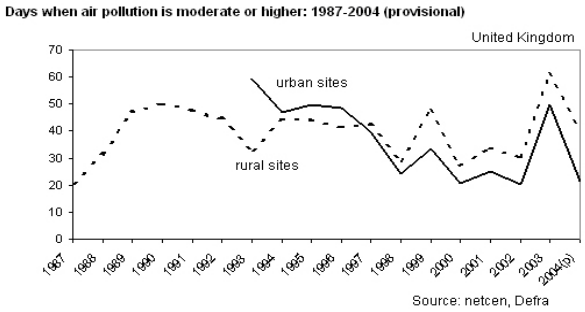
The UK uses "Headline Indicators" as an instrument of government - the indicator on air pollution (DEFRA, 2004) is supposed to inform the public of government performance on reducing pollution levels. It involves counting the number of days on which particles (PM_{10}) exceed the 50 $\mu g/m^3$ standard, or on which ozone exceeds 50 ppb for one or more hours in that day. The days on which one or more of PM_{10} or ozone (or sulphur dioxide) exceed these numbers are summed for each monitoring site and the average calculated over all UK national network sites. The results in Figure 1 show ozone and particles (PM_{10}) contribute, but sulphur dioxide hardly.



Note the 'particles' data are based on under-readings of PM_{10} by TEOM monitors, but this goes unmentioned.

Figure 1: Days of pollution exceedances, UK Urban sites.

Figure 2 shows that on this measure urban air is improving and that rural air is now worse than urban. This reflects the pattern of Figure 1 where ozone is currently assessed as worse than PM_{10}. However, giving equal weight to PM_{10} and ozone exceedances does not correspond to the actual harm (the ozone standard is for 8-hour not 1-hour average, while PM_{10} is now seen as many times more harmful than when the 50 $\mu g/m^3$ standard was set). Nitrogen oxides don't register, despite concerns over this urban traffic pollutant. Though scientifically unsound, the UK's Headline Indicator is retained for political expediency as it implies – wrongly – that the urban pollution problem is under control. The consequence has been public cynicism, bolstered by official excuses over the anomalous 2003 data. Secondly, air pollution dropped down the government's agenda and funding started to be cut back. The parliament's Public Accounts Committee (2002) questioned the spending, when air pollution modelling is very imprecise and there are considerable uncertainties and gaps in the evidence of the health effects of air pollution. The Committee required the government to address the most important gaps in its knowledge, such as which sizes of particles are the most dangerous, but this issue is still being ducked.



Note the 'particles' data are based on under-readings of PM_{10} by TEOM monitors, but this goes unmentioned.

Figure 2: Pollution exceedances, rural v. urban sites.

When Labour took over UK government in 1997, with healthy air as one of their priorities, they said they wanted straight talk. Instead of "air quality," we should talk of "air pollution." Instead of describing air quality as PM_{10} *very good*, PM_{10} *satisfactory* or PM_{10} *poor/very poor*, we now describe pollution levels as *Low, Moderate, High, Very high.*

These levels are defined numerically, but as Figure 3 shows (taken from DEFRA, 2002), they are described in relation to individual perception and likely symptoms. Maynard and Coster (1999) discuss more fully this banding system used for public information purposes.

numerical index /Pollution band	Health effect
1-3 low	Effects are unlikely to be noticed, even by people who know they are sensitive to air pollutants
4 - 6 moderate	Mild effects are unlikely to require action, but sensitive people may notice them
7 - 9 high	Sensitive people may notice significant effects, and may have to act to reduce or avoid them (for example, by reducing time spent outdoors). Asthmatics will find that their reliever inhaler should reverse the effects of pollution on their lungs
10 very high	The effects of high levels of pollution on sensitive people may worsen when pollution becomes very high
Sensitive individuals are people who suffer from heart and lung diseases, including asthma, particularly if they are elderly.	

*any recommended concentration other than zero implies that a view has been taken on the tolerable levels.

Figure 3. Bandings for UK Public Information (DEFRA, 2002).

Descriptors like "satisfactory" require judgement via substantive public input - and can be disputed by sensitive and vulnerable individuals (Maynard, 2003). The Toronto medical community (Toronto Public Health, 2001) likewise contested Ontario's use of similar descriptors.

ADVISING THE PUBLIC ON PRECAUTIONARY ACTION

There is no element in this system of advising prudent avoiding action (e.g. avoiding energetic exercise when ozone is high) despite the chronic ill-effects now known. Though the UK now accepts that the chronic effects of PM exposure are many times greater than the acute effects (COMEAP, 2001), the public information was not changed. Only "sensitive" people are advised to take action - and asthmatics are told the pollution is not harmful as their inhaler reverses the effects (Figure 3). This reflects the UK regulatory culture - to avoid alarming the public.

The UK regime is based on the approach that "the risk to healthy individuals is very small at all levels of air pollution likely to be experienced in the UK" (Maynard & Coster, 1999)

Does this reassuring statement imply that only ill persons, with respiratory or heart conditions are at risk? In practice very many do suffer from air pollution (10-20%, including asthmatics) and they are called "sensitive." Infants and children are not, however, considered sensitive (Figure 3). During high pollution days, children with asthma are advised to continue take part in physical sports as normal, but just to use their inhaler more (DEFRA, 2002).

HEALTH-BASED STANDARDS – FAILURE FOR PM

The UK standards were intended to be health-based, using e.g. the "lowest effect" levels for SO_2 and NO_2. The standard for PM_{10} was set at 50 $\mu g/m^3$ (24 hr), despite WHO saying there's no threshold and despite evidence of effects down to 20 $\mu g/m^3$. The difference (30 $\mu g/m^3$) was deemed "acceptable," and calculated to give one hospital case per day in a city of 1 million people. Thus the current UK standard is based on the 1995 decision that PM_{10} if under 50 $\mu g/m^3$ denotes "very good" air quality.

The UK authorities are well aware of the American Cancer Society (ACS) and Six Cities long-term studies with the HEI "reanalysis" (see Craig & Shortreed, 2003), which established that PM has chronic effects on health, not just the acute effects from high pollution episodes as was accepted in 1995. The chronic effects cause ill-health in many of the supposed "healthy" individuals. The loss of life-years was given by the UK's Committee on Medical Effects of Air Pollutants (COMEAP, 2001) to be over 10 times higher than the loss via acute effects, probably over 25 times higher. They likened the harm to health to that from heart disease caused by 'passive smoking,' which is termed a 'substantial public health hazard'. However, the chronic effects of PM pollution apply to the whole population, not only the 20% partners of smokers.

This sets the context for the question - how did the UK government respond to this substantial public health hazard? They confirmed weakened 24-hour standards on PM_{10} (allowing 35 exceedances/yr rather than 3) and proposed worsening the annual PM_{10} target for London (now 23 $\mu g/m^3$ instead of 20 $\mu g/m^3$). How was public information on air pollution changed? No change was made to the low/moderate description for PM_{10} in the 2002 version, as referred to above. The only ill-effects from PM were still given as worsening of heart and lung diseases. Just a sentence was added saying "experts suggest" long-term exposure to PM changes life expectancy. It does seem anomalous that the UK has scientific leaders in the field and that COMEAP produces a careful review concluding PM pollution is a serious issue, yet no UK scientists speak out over this cavalier treatment by government.

NEED FOR INFORMATION CRITERIA AND ACCOUNTABILITY

As the Conference *Concept Document* (Craig & Shortreed, 2003) states, it's important that messages concerning health effects and health protection provide an accurate reflection of the risks in a form that is easily understood. The Maynard-Coster criteria read, in amended form:

- to assist the public to understand the impact of air pollution on their health and the wider environment;
- to encourage the public to reduce emissions of pollutants within their control;
- to alert those who may be affected by air pollution to take timely precautions to avoid such effects; and
- to enable the public to assess progress in reducing air pollution towards harmless levels.

Assessing the UK provision according to such criteria is still needed as the criticisms above indicate. Toronto provided an impressive example of mismatch between information and reality, with the May 2000 study that showed all smog alerts in Toronto are triggered by ozone, while ozone is responsible for less than 5% of the premature deaths and about 30% of the hospitalisations attributed to air pollution. The study (Toronto Public Health, 2001) found 92% of Toronto's smog-related premature deaths and hospitalisations occur when the city's air quality is classified as "good" or "very good" by the Air Quality (AQ) index. The medical community rightly demanded restructuring of the AQ indicator and messaging system. Accountability in air pollution regulation, such as set out in the HEI methodology (2003), needs to be included as an integral element.

ADDRESSING THE PM-SIZE ISSUE

Ultrafine Particles – mismatch of science and UK policy

In Friends of the Earth, we caught onto the idea that ultrafine particles (UFP) are potentially the serious pollutant in 1996 when we were campaigning on industrial emissions. Seaton and colleagues formulated the hypothesis that particles smaller than 0.1 μm may be harmful just because such small particles cause inflammatory reactions in the lung. By the year 2000, the Royal Society meeting and book *Ultrafine Particles in the Atmosphere* (Royal Society,

2003) marked a watershed in acceptance that ultrafine particles are probably the most harmful fraction of PM and, second, that controlling PM_{10} fails to address the hazard of ultrafines because they contribute little to the total mass. Yet for public information and policy purposes, government officials and their expert advisors disregarded this science verdict.

Let me summarise the sad story of the treatment of PM by EPAQS, the advisory Expert Panel on Air Quality Standards. First, EPAQS (1995) recommended a UK Air Quality standard for airborne particulate matter PM_{10} of 50 $\mu g/m^3$ (as a running 24-hour average), which Government accepted. It was adopted into law in 1997 and was to be achieved by 2005 (99%ile). The favoured measuring device was the TEOM, of which many were purchased and installed (before the substantial under-reading due to heat-induced evaporation was discovered).

By 1999, international trends and lobbying from UK scientists had forced government to re-think; they accepted

- PM toxicity may lie in a finer fraction of the particles, perhaps below 2.5 µm or smaller
- PM_{10} measurements may include re-suspended dust of probable low toxicity
- difficulties in compliance where wind-blown dust or sea spray include 2.5 to 10 µm particles

The government asked EPAQS to consider an additional/alternative standard, taking account of the smaller size range thought to be responsible for toxic effects. However, EPAQS's draft report in 2000 proposed no change. It declared that a standard based on PM_{10}

> "would provide adequate protection for the public" (EPAQS, 2001).

The draft was heavily criticised, particularly for omitting much evidence on ultrafines and ignoring the increasing international use of $PM_{2.5}$, including by COMEAP. However, EPAQS (2001) stuck to their 'no change' stance, saying:

- PM_{10} is a metric that has stood the test of time
- no evidence that any alternative metric shows a closer and more reliable association with health outcomes
- They misrepresented critics as saying "it is only necessary to control the finer fraction in order to limit effects on health."
- They claimed that *in the UK*, $PM_{2.5}$ and PM_{10} are inseparable.

EPAQS did acknowledge that toxicological effects *may be* found in a finer fraction $PM_{2.5}$ or in ultrafine particles smaller than 0.1µm in diameter, poorly represented by PM_{10}. They nevertheless concluded that the "metric PM_{10} provides the *most appropriate basis* for a *UK* Air Quality Standard" (EPAQS, 2001). Only late in 2003, did EPAQS move reluctantly to accepting the international adoption of $PM_{2.5}$.

European Pressure

Pressure for action on PM and use of $PM_{2.5}$ has been building in Europe via the CAFE process and scientist committees in WHO (e.g. the January 2003 declaration that urban PM shortens lifetimes by 1-2 years (WHO, 2003)). This led to the European Commission's (2003) *Airborne particles and their health effects in Europe* in March, which tried to cut through politicians' reluctance by stating firmly

> "present levels of airborne particles are causing severe damage to human health..."

Unusually it needed the Commission to lead with firm statements:

- no apparent threshold of PM... no safe level may be defined
- reduction in life expectancy up to a few years
- $PM_{2.5}$ found to be more hazardous than the coarser fraction ($PM_{10} - PM_{2.5}$)

The Commission put down a marker on ultrafines:

- Ultrafine particles may... more research is needed to quantify.

The UFP story reveals a systemic failure

The Commission marker on ultrafines is too weak in view of the 4 years since the Royal Society (2000) watershed meeting (Royal Society, 2003) and abundant research findings. The issue that the Commission should have posed is - what kind of ultrafine PM monitoring and control system could be set up, as a first step for later refinement.

The mismatch between particulate toxicology and controls is increasingly glaring, the former finding that most and the most severe health impacts are related to ultrafines (nanoparticles), while the latter centres on micron-particles measured by $PM_{2.5}$. The main reason for the mismatch is the heavy investment in $PM_{2.5}$ monitoring and epidemiology, driven by the regulators, plus the attitude that controls on ultrafine particles are impractical (even "doomed to failure" said DEFRA's Martin Williams; "insufficient data on which to base a standard" said EPAQS (2001)). The limited evidence shows the UFP problem as worsening - data collected for the ULTRA project showed that ultrafine particles (<30nm, i.e. 0.03μm) increased through the 1990s in Erfurt while the levels of PM_{10} were steady and $PM_{2.5}$ were decreasing (Wichmann & Peters, 2000). Thus policies to tackle PM_{10} may well miss the major culprit for ill-health.

That the Background Papers and the *Concept Document* for the Rome AIRNET meeting missed out any mention of ultrafine particles is a significant failing, due in part to the issue being hardly appreciated in North America. The Policy Strategies and Options paper (van Bree et al., 2003) did quote the view that standard setting on the basis of the PM indexes is likely to be a sub-optimal strategy for managing particle pollution (Maynard et al., 2003) because of the different components. But it failed to raise for discussion the postulate that ultrafine size alone is the strongest indicator of adverse health impact - and strong enough for setting controls on urban pollution. Thus the AIRNET organisers' ducking of the ultrafines issue appears to mirror that of the Commission and much of the air pollution community.

The UK systemic failure on PM is perhaps unusual. But the ultrafine particle story shows a wider institutional failure within Europe, considering European scientists have been leaders in toxicology, epidemiology and measurement of ultrafines (Donaldson et al., 1999, Wichmann & Peters, 2000) and the 3-year old HEAPSS project focuses on this (Aalto et al., 2003, Lanki et al., 2003). There can be no simple answer, but openness is surely key as follows:

- scientists to work separately from policy people, as in WHO (not mixed as in EPAQS);
- international comparisons, scrutiny and best practice examples - where AIRNET helps;
- public access and involvement - which in practice must be via ENGOs;
- direct access of non-governmental groups (scientists and ENGOs) to the Commission.

RECOGNISING SUSCEPTIBLE AND VULNERABLE PERSONS?

It is increasingly recognised that individuals are susceptible not just through respiratory disease, but because their defence systems are immature (children), genetically limited, or compromised in exposure or health history. It is easy to criticise the blindness of EPAQS, when proposing their PM_{10} standard of 50 $\mu g/m^3$ (EPAQS, 1995), when they stated "the very large majority of individuals should be unaffected." One of their number has taken it even further by declaring only "exquisitely sensitive" individuals are affected (when giving evidence for an incinerator applicant at a public inquiry). The problem is not simply that scientist-experts ignored the susceptible, but that they did not realise they were stepping outside their science remit and exercising judgements that properly belong to a wider community and even to the political field. The UK has strong guidance (RCEP, 1998) that science advisory committees should not stray in this way, but the old culture remains strong (among officials who steer committees).

RCEP's guidance took EPAQS to task over an example of a pollutant for which there is no safe level of exposure, saying

> "any recommended concentration other than zero implies that a view has been taken on the tolerable levels of harm. Such a decision should not be taken by an ostensibly scientific body."

A clear separation of functions is required in handling scientific, economic, social and value issues in risk assessment and management. When air pollution standards are set and or other judgements made about environmental issues, the guidance says decisions must be informed by an understanding of people's values.

POLICY GUIDANCE ON SUSCEPTIBLE SUB-POPULATIONS

The EU policy guidance on susceptible persons has not been clear and consistent. The Fifth Environmental Action Plan said

> "all people shall be effectively protected against recognised health risks from air pollution"

The word "all" surely covers susceptible sub-populations and action is implied once risks are recognised. Questioning how are risks to be recognised, leads to the appreciation that implementing this proposition is not just science but a social/political process.

The Ambient Air Directive 96/62/EC ducked out of stating any principled objective. It talked in terms of ..avoid, prevent, reduce.. to limit values, or of setting target values (for ozone) where an absolute objective might be difficult or long-term. Till recently, WHO (1999) has led with the principle

> "to maintain a quality of air that protects human health and welfare… must also provide protection of animals, plants (crops, forests and natural vegetation), ecosystems, materials and aesthetics."

The EU's Sixth Environmental Action Plan's relevant objective (Article 7) reads

> "achieving levels of air quality that do not give rise to significant negative impacts on and risks to human health and the environment,"

which drops the promise to cover everyone. However priority actions (2a of Article 7) include

> "re-examination, development and updating of current health standards and limit values, including where appropriate, the effects on potentially vulnerable groups, for example children or the elderly."

Including the Unborn

Thus susceptible sub-populations are now explicitly covered. But are the unborn included? The AIRNET background papers refer to several recent studies showing pre-birth damage from air pollutants. "Saving the Children" (*Health & Clean Air*, 2003) also reviews the issue. CO and SO_2 are associated with low birth weight. Abnormal development (CO - holes in the heart; Ozone - defective heart valves etc.; PM - birth defects) appears to correlate with maternal exposure in second month of pregnancy.

There is no doubt that air pollution policy needs to include the unborn. They could be considered a 'vulnerable group' for the Sixth Environmental Action Plan, and legal clarification is desirable. In the meantime, let's proceed as if the unborn are included!

SCIENCE-STAKEHOLDER-POLICY INTERPLAY

The AIRNET Science-Policy Interface work group suggested in their conference draft (of 30/10/03) that scientists, policy-makers, stakeholders and the public are natural allies. Clashes at Rome on the conference floor showed, however, rather strong differences in views and interests between ENGOs, industry and policy-makers. Judgements on air pollution ranged from serious to marginal effects on health. The critical appraisal of UK policy-making in the text above also shows that the "natural allies" concept is over-simple. Picturing "science" as neutral is also problematic. Science can be divorced from reality and elitist as implied in Figure 4. ENGOs and the public are told only "sound science" counts, that results must be robust, that any challenge to established science requires

"compelling evidence." Statistical correlations revealed via epidemiology are no proof. Thus science and scientists are raised on a pedestal. Questioning and campaigning are rejected as public hysteria, lay arguments are characterised as ‘thin pieces of evidence stitched together by strident activists to create an alarmist picture’.

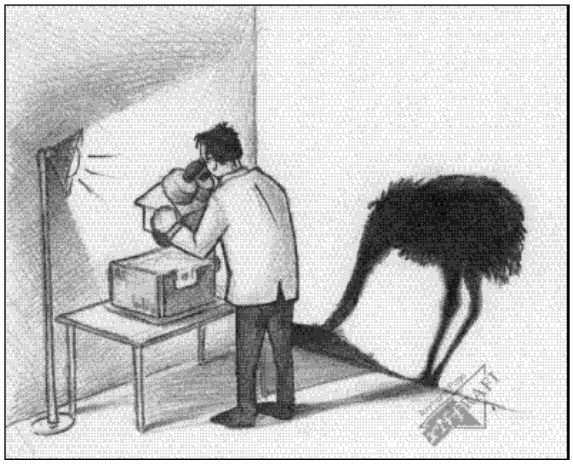


Figure 4: Science divorced from the real world.

The approach taken in the opening paper *AIRNET in Focus: Status report* (Brunekreef 2003) shows a way to counter this. This reminds us of limitations of the scientific process, from underestimating uncertainty and measurement error to publication and institutional bias. Weighing the evidence requires judgement that brings in a range of non-science factors; while scientists can try to minimise these they can't necessarily exclude them. In medical practice, the validity of public perception and knowledge, at some level, is increasingly recognised - the same should happen for health impacts of air pollution. Brunekreef also pointed out that asking for ever more solid scientific evidence ("sound science") can be an excuse for delaying decision-making, and so lead to "unsound" public health policy. Outstanding past failures from asbestos to nuclear radiation have been reviewed by the European Environment Agency (EEA, 2001) - this work includes a scholarly presentation of methodology for integrating the precautionary principle into science policy-making.

ENGOs and the public are becoming well versed in criticising risk-assessment toxicology, especially when presented with black-box numerology of complex computer programs. We tell the modellers they don't know if they've included all pathways, they lack data on combinations of pollutants, they cannot include unknown (trace) pollutants and their model of the standard human does not accommodate the wide range of susceptibilities. Modellers can reach opposite conclusions because of assumptions on data gaps and data extrapolation (Reid et al., 2005). We also remind the modellers that stakeholder participation is vital for risk analysis. Public acceptability of risks requires public participation in the decisions that create and manage such risks, including the consideration of values, attitudes and overall benefits. Sound public policy-making on issues involving science therefore requires more than good science: ethical as well as economic choices are at stake (EEA, 2001). Submerging risk in cost-benefit analysis does not avoid this and other criticism. It can help determine relative cost-effectiveness, but cannot tell us how far to abate air pollutants. The twist in the background paper (Rabl et al., 2005) of valuing a person's life according to her/his country's GDP per head runs up against objections of equity and of failure to include the non-market (social and informal) economy.

CONCLUSION

AIRNET is the type of activity that can moderate if not overcome the antagonisms. But there are histories of experience that cannot be disregarded. ENGOs generally see power hierarchies and privileged access, and use public campaigning to demand to be heeded by decision-makers. As far as we integrate in the Science-Policy Interface - reserving our right to campaign outside it too - we'd consider a model like that in Figure 5. This indicates how, for

European air quality, science currently feeds into a pretty open deliberative process with CAFE at the centre and outputs to EU level institutions. The question is - where should AIRNET position itself? If it chooses the right hand side of Figure 5, AIRNET claims privileged access to the Commission and policy-making processes. In that case, it is surely required to be accessible to all stakeholders, to be broadly representative and to operate democratically. AIRNET cannot fulfil that, if only because of its limited life. So AIRNET should be content with the left-hand position in Figure 5, helping Science feed into the deliberative mêlée, and designing its outputs for the participating parties and stakeholders. AIRNET can help integrate accountability into air pollution policy-making (HEI, 2003), via international comparisons and audits. The AIRNET project can make use of ENGOs as surrogates for the public stakeholder and aim to win them as allies in securing policy change.

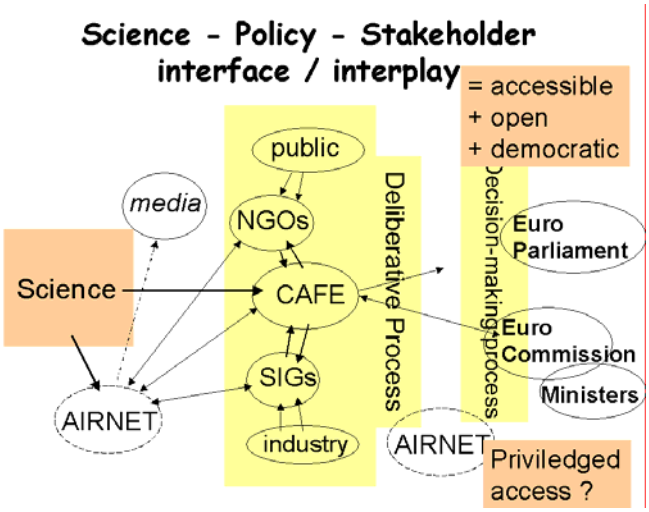


Figure 5: Where should AIRNET position itself?

Acknowledgement

Max Wallis is a research fellow in the Cardiff Centre for Astrobiology (Cardiff University, UK) and works with the Welsh ENGO Cyfeillion y Ddaear environmental network. He served for several years till 2002 as an ENGO representative (for Friends of the Earth) on the UK government's consultative Air Quality Forum. His presentation slides are found at http://airnet.iras.uu.nl/resource/flash/rome/0915_wallis_day3.php.

REFERENCES

Aalto, P., Paatero, P., Kulmala, M., Hämeri, K., Forastiere, F., Cattani, G., Marconi, A., Cyrys, J., Von Klot, S., Zetzsche, K., Peters, A., Bouso, L., Castaño, G., Palacio, J.A., Sunyer, J., Lanki, T., Pekkanen, J., Sjöval, B., Berglind, N., Bellander, T., and Nyber, F. 2003. Aerosol number concentration measurements in five European cities during HEAPSS, AIRNET/NERAM Conference. http://airnet.iras.uu.nl/resource/posters/rome/ airnet_poster65_f_forastiere.pdf.

Brunekreef, B. 2003. AIRNET in focus: status report, AIRNET/NERAM Conference. http://airnet.iras.uu.nl/resource/flash/rome/0900_brunekreef_day1.php.

COMEAP (Committee on the Medical Effects of Air Pollutants) 2001. Statement on long-term effects of particles on mortality, April 2001. www.doh.gov.uk/comeap/ statementsreports/longtermeffects.pdf.

Craig, L., and Shortreed, J. 2003. Strategies for Clean Air and Health, AIRNET/NERAM Conference Concept Document, draft 24/10/03. http://www.irr-neram.ca/rome/concept.pdf.

DEFRA 2002. *Air Pollution - What it means for your health*, London: The Stationery Office May 2002. www.defra.gov.uk/environment/airquality/airpoll/index.htm.

DEFRA 2004. *Headline Indicator*. www.sustainable-development.gov.uk/indicators/ headline/h10.htm.

Donaldson, J., Stone, V., and MacNee, W. 1999. The toxicology of ultrafine particles. In *Particulate Matter: Properties and Effects Upon Health*, eds. R.L. Maynard, C.V. Howard, pp. 115-129. Oxford:BIOS.

EEA (European Environmental Agency) 2001. Late lessons from early warnings: The precautionary principle 1896–2000. Environmental issue report No 22. www.eea.eu.int.

EPAQS 1995. *Particles.* London: The Stationery Office. www.defra.gov.uk/environment/ airquality/aqs/particle/index.htm.

EPAQS 2001. *Airborne particles: what is the appropriate measurement on which to base a standard?* TSO London. www.defra.gov.uk/environment/airquality/aqs/air_measure/ index.htm.

European Commission 2003. *Airborne particles and their health effects in Europe.* Information Note ENV.C1/AZr.

Health & Clean Air 2003. *Saving the Children* newsletter, Spring 2003. www.healthandcleanair.org.

HEI (Health Effects Institute) 2003. *Assessing Health Impact of Air Quality Regulations: Concepts and Methods for Accountability Research*. HEI Communication 11, Sept. 2003. www.healtheffects.org.

Lanki, T., Tiittanen, P., Forastiere, F., Nyberg, F., Paatero, P., Pekkanen, J., Peters, A., Sunyer, J., and HEAPSS Group 2003. Air pollution and hospitalisations for first myocardial infarction (MI) in the HEAPSS cohort. AIRNET/NERAM Conference. http://airnet.iras.uu.nl/posters/pdf/airnet_poster66_t_lanki.pdf.

Maynard, R. 2003. Scientific information needs for regulatory decision making. *J. Toxicol. Environ. Health* 66:1499-1501.

Maynard, R.L., and Coster, S.M. 1999. Informing the Public about Air Pollution. In *Air Pollution and Health*, eds. S.T. Holgate, J.M. Samet, H.S. Koren, and R.L. Maynard, pp. 1019-1033. San Diego, CA:Academic Press.

Maynard, R., Krewski, D., Burnett, R. T., Samet, J., Brook, J.E., Granville, G., and Craig, L. 2003. Health and air quality: Directions for policy-relevant research. *J. Toxicol. Environ. Health* 66:1891-1904.

Public Accounts Committee 2002. Treasury Minutes on the reports of the Committee of Public Accounts 2001-2002. 34th Report. *Policy Development: Improving Air Quality*, House of Commons July 2002.

Rabl, A., Nathwani, J., and Hurley, F. 2005. Tools and strategies for improving policy responses to the risk of air pollution, AIRNET/NERAM Conference background paper. *J. Toxicol. Environ. Health*. This issue.

RCEP 1998. Royal Commission on Environmental Pollution *Twenty-first Report, Setting Environmental Standards*. London: The Stationary Office.

Reid, N., Misra, P.K., Amman M., and Hales, J. 2005. Air quality modelling, AIRNET/NERAM Conference background paper. *J. Toxicol. Environ. Health*. This issue.

Royal Society 2003. *Ultrafine particles in the Atmosphere*, eds. L. M. Brown, N. Collings, R.M. Harrison, A.D. Maynard, and R.L. Maynard (issued also as *Phil. Trans. R. Soc. Lond.* A358, 2561-2797, 2000). London: Imperial College Press.

Samet, J., and Krewski, D. 2005. Health effects associated with exposure to ambient air pollution, AIRNET/NERAM Conference background paper. *J. Toxicol. Environ. Health*. This issue.

Toronto Public Health 2001. Condition Critical. Fixing our Smog Warning System. Toronto, Ontario. www.city.toronto.on.ca/health/sws_index.htm.

van Bree, L., Vandenberg, J., and Craig, L. 2003. *Policy Strategies and Options* discussion draft, AIRNET/NERAM Conference. http://airnet.iras.uu.nl/resource/flash/rome/ 1345_van_bree_day2.php.

WHO 1999. *Guidelines for air quality*. www.who.dk/document/e71922.pdf.

WHO 2003. Health Aspects of Air Pollution with Particulate Matter, Ozone and Nitrogen Dioxide. Report on a WHO Working Group, Bonn, Germany, 13–15 January 2003. www.who.dk/document/e79097.pdf.

Wichmann, H.E., and Peters, A. 2000. Epidemiological evidence of the effect of ultrafine particle exposure. *Phil. Trans. Roy. Soc. Lond.* 358:2751-2769.

CLOSING THE GAP BETWEEN SCIENCE AND POLICY ON AIR POLLUTION AND HEALTH - THE *AIRNET* ENTERPRISE [1]

Leendert van Bree, Nina Fudge[2], Jouni T. Tuomisto[3], Bert Brunekreef[4]

ABSTRACT

This paper discusses critical issues underlying the interface between air quality science, stakeholder participation and policy development within the context of the European AIRNET Network multi-stakeholder project. The paper argues that it is not only the content of air pollution and health issues that stakeholders consider important, but also the process and mechanisms by which the interface operates. A visual representation of the interaction between science, society and stakeholders in the development, dissemination and evaluation of effective air quality policy strategies is provided. The paper discusses the role of AIRNET in supporting the Clean Air for Europe (CAFE) program and assesses the AIRNET experience in establishing a network to bridge the gap between air quality policy, stakeholders, the public, and scientific communities.

[1] The European Network AIRNET was initiated within the Quality of Life and Management of Living Resources programme of the European Commission, QLRT-2001-00441, Key Action 4, Environment and Health. The content of this paper has formed the basis for the AIRNET Science-Policy Work Group end-report discussions to sharpen its focus on the science-policy-stakeholder interplay on air pollution and health. It has also functioned to generate the outline of the draft report presented at the AIRNET-NERAM Rome Conference on Strategies for Clean Air and Health, November 5-7, 2003. The authors are grateful for the many suggestions from the Work Group.
[2] Netherlands Environmental Assessment Agency (MNP), Bilthoven, The Netherlands.
[3] National Public Health Institute, Kuopio, Finland.
[4] Institute for Risk Assessment Sciences, Utrecht University, Utrecht, The Netherlands.

SCIENCE-POLICY-STAKEHOLDER INTERPLAY

Past and recent improvements in environmental quality, including air quality, are the result of weighing ecological, economic, and social interests in decision-making. Despite this success, air pollution is still one of the major environmental factors affecting human health (Brunekreef and Holgate, 2002) and concern over the substantial health impact remains high. In addition, substantial investments have to be put into further emission reductions to decrease the remaining risks, which may trigger debate on "value for money." The science-policy-stakeholder interplay in the area of air pollution becomes therefore even more important when developing health-effective and cost-efficient control strategies and measures that are transparent and sound, and carry public support (Figure 1).

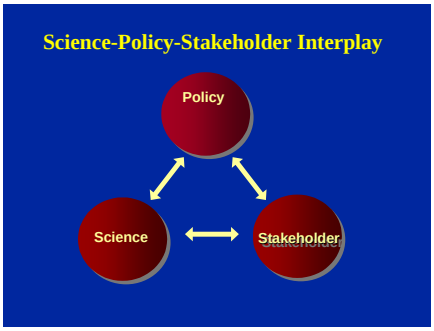


Figure 1. Science-policy-stakeholder interplay in effective air pollution control strategies.

It seems difficult to define what the science-policy-stakeholder interplay (or interface, as it is often called) is really about. Bringing scientists, policy makers, stakeholders, and the public together is a difficult process. Ideally, these parties should be allies in efforts to promote more sustainable approaches in environment and health to bring the exposure and health risk down to healthy or acceptable levels, but the interface between the players is often under-developed or functions poorly (Samet and Lee, 2001; Ginsburg and Cowling, 2003) and, in addition, traditionally such parties may have been more antagonistic than cooperative. Scientists may not view policy makers as legitimate clients for their research results. Policy makers may not perceive the research community as the producer of relevant information for decision-making processes. Policy makers often desire a level of certainty which science cannot offer (Matanoski, 2001).

There are a number of critical issues in this interface. Firstly, there is a need for better communication of scientific information to those who need it, ask for it, or have the right to know about it. Secondly, there is a need for scientists to better understand stakeholder interests and perspectives. Finally, there is a need for developing views on what science-policy-stakeholder interplay is, or how it should operate as a process. Noteworthy, many institutions develop risk communication strategies nowadays to break these communication barriers down, and the approaches have developed into a discipline in its own right.

Therefore, it seems that it is not only the content of air pollution and health issues that players consider important, but also the process and mechanisms by which the interface operates. This interplay and co-operation between science, society, policy, and stakeholders could be visualised as displayed in Figure 2, based on the premise that they all play a role in the generation, dissemination, and evaluation of policy options.

Why do we need this interplay? The use of models and techniques to assess the various impacts and the application of aggregated impact indexes to support scenario analyses and outlook-type of assessments make it necessary to get broad support from scientists, policy makers, and stakeholders. On the one hand one needs to know what the different information needs of the various stakeholders are in order to get the outcomes as informative and targeted as possible. On the other hand one needs agreement on problem definition and framing, various risk characterization aspects, decision rules, preferred methodology and assessment models, and necessary input data in order to increase confidence and support for the outcomes and their implications for decision-making. Furthermore, integrated assessments of air pollution and of control policy options and strategies have also to be considered from the perspective of sustainable development. Development and monitoring of effectiveness of sustainability strategies therefore also need broad interplay between the various players in this field. Current views on environmental and health consider three different perspectives:

- "*human-social perspective*" *(people)*
- "*ecological perspective*" *(planet)*
- "*economic perspective*" *(profit)*

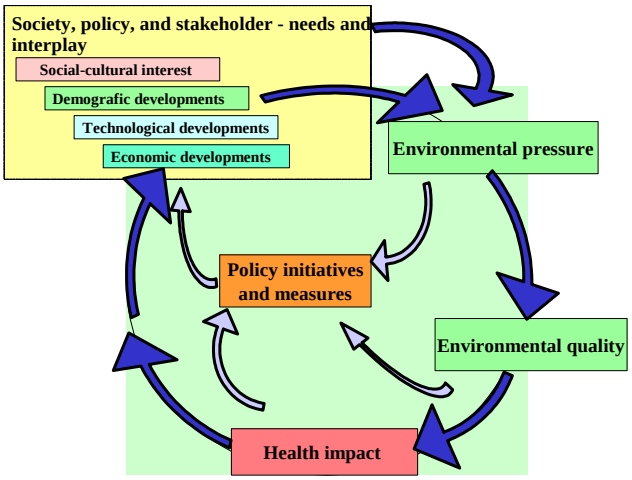


Figure 2. Science, society, policy and stakeholder interaction and co-operation in generation, dissemination, and evaluation of risk-based policy options and priorities.

Therefore, such an interplay is also needed to allow "people" and "planet" players to take their responsible positions to achieve a sustainable development in a (many times) market-driven economy ("profit").

AIRNET

The European AIRNET Network is a multi-stakeholder project in the field of air pollution and health (http://airnet.iras.uu.nl). The main objective is to create a widely supported basis for public health policy related to improving air quality in Europe and regulatory needs to achieve that goal.
The specific objectives are:

- To develop an interpretation framework for the (final and intermediate) result of research supported by the 4th and 5th EU Framework Programmes, as well as nationally funded studies;
- To collect, discuss and interpret the (final and intermediate) results of research supported by the FP4 and FP5 programmes, as well as nationally funded studies;
- To draw policy-relevant recommendations from the activities mentioned.

AIRNET is co-ordinated by the Institute for Risk Assessment Sciences, Utrecht University, Utrecht, the Netherlands, and the Netherlands Environmental Assessment Agency, RIVM, Bilthoven. AIRNET, and the Science-Policy Work Group in particular, functions as a structure and forum to get the science-policy-stakeholder interplay working for air pollution and health (AIRNET News 2, 2003). Within AIRNET the scientific input that is needed to better meet the end-user's needs is discussed to support authoritative institutions to start building a more targeted and sustainable policy. The objective of the AIRNET Science-Policy Working Group is to facilitate an interactive communication and review forum in air pollution and health. The aim is to discuss and interpret the outcomes of air pollution health risk research and impact assessments and to link these outcomes with the various end-users' needs, policy issues, and abatement strategies and control measures. Thus, it forms a bridge-building network between policy, stakeholders, public, and scientific communities on end-user relevant research outcomes and its implications. The actual structure of AIRNET as a network in Europe can be visualized as displayed below (Figure 3):

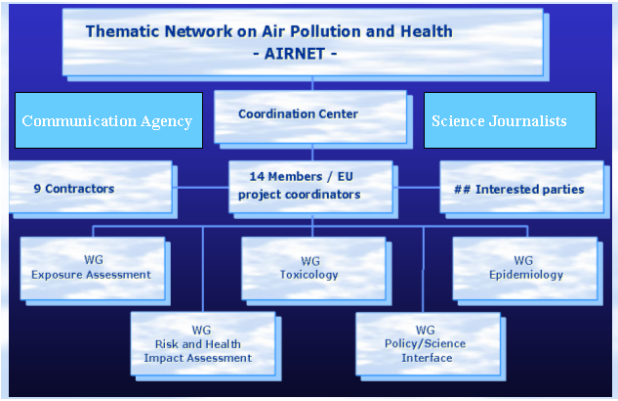


Figure 3. The AIRNET multi-stakeholder network structure.

AIRNET'S ROLE IN THE "CLEAN AIR FOR EUROPE" (CAFE) PROGRAMME

The Commission of the European Union has initiated the Clean Air for Europe (CAFE) programme to strengthen its air pollution policy (http://europa.eu.int/comm/environment/air/cafe). This will be based on the best available science and will be created in a broad, open, and transparent dialogue with the scientific community, as well as with the public and stakeholders. The major objectives of CAFE are:

- To review existing air quality standards and national emission ceilings as set out in recent legislation, and to contribute to the review of international protocols on the basis of the best and most recent scientific and technical information, taking into account experience of implementation of existing legislation and protocols;
- To develop new, flexible and comprehensive mechanisms for gathering information leading, in the longer term, to the further development of objectives and indicators for outdoor air quality;
- To identify where there may be a need for additional measures to reduce emissions from specific sources;
- To propose and update a strategy at regular intervals which defines appropriate air quality objectives for the future and cost-effective measures for meeting those objectives.

The AIRNET Network supports the work of CAFE by providing new research findings in an interpreted manner as a result of a process in which the views of the various end-users and stakeholders have been incorporated explicitly. The five separate reports produced by AIRNET on exposure, toxicology, epidemiology, health impact and science-policy include some major findings directly of interest to decision-makers. Among the major highlights are: (i) the relevance of population and personal exposures to estimate and interpret health effects and contributing sources, (ii) the strengths and weaknesses in the health effects evidence of short-term and long-term exposures, (iii) new ways to assess quantitatively health effects and burden of diseases, (iv) new insights in particulate matter components which seem mostly relevant for health effects (primary anthropogenic particles), (v) framing of risk issues including uncertainty analysis, and (vi) developing a science-stakeholder-policy interplay based on end-user needs and methods of communication and (understandable) information transfer. This output improves the two-way communication between CAFE and the various communities (international, national, and local decision makers, civil society representatives, interest groups, industry etc.), through science-policy-stakeholders interplay, and focussing on building transparency and trust. The output of AIRNET has been included in the reference list.

STRATEGIES FOR CLEAN AIR AND HEALTH

Air pollution and health issues and targeted control strategies apply to the level of both supranational, national and local/domestic policy makers. Keeping this in mind, it is obvious that the society-policy-stakeholder interplay should also function at these levels and has therefore to incorporate the different needs generated at these various levels. For AIRNET this means that one has to offer research outcomes, evaluations, and decision-making recommendations at the various levels of decision-making interest (international, national, local). It is this science-policy interplay perspective that the AIRNET management team focussed the second Annual

AIRNET-NERAM 2003 Conference on “Strategies for Clean Air and Health” and generated a targeted communication strategy. In its final year 2004, AIRNET has tried to implement this communication strategy in so-called ‘network days’. These days were held in the Netherlands, Sweden, Hungary, and Spain to try to cover different regional aspects of air pollution issues. At each of these occasions an interactive working format was chosen to stimulate the science-policy-stakeholder interactions (roundtables, speaker’s corner, breakout groups, plenary sessions, and regional European air pollution input and information needs i.e. West, East, North, or South). Also the AIRNET 2004 Prague Conference has been organised in this way incorporating also the outcome of the various network days. The AIRNET-NERAM 2003 conference statement is available for downloading (http://www.irr-neram.ca or http://airnet.iras.uu.nl). The interplay network model used is displayed in Figure 4 including the main issues and working formats of interest.

Figure 4. The multi-stakeholder communication set-up and framework applied in AIR-NETwork days and annual conference.

AIRNET’S LESSONS LEARNED

To what extent has AIRNET “bridged the gap” and what are the lessons learned to act as a multi-stakeholder network? Historically, the wish to create an overarching “science-policy” network in Europe was generated by scientists themselves. The participation of the few stakeholders at the beginning and identifying their information gaps rapidly resulted in a need to increase their number, to incorporate different levels of stakeholders, and to try to identify their information needs more thoroughly. AIRNET also learned that the science-stakeholder interface is interpreted differently by the different players, is not naturally occurring, does not work by itself, and requires lots of planning, and structure and energy (e.g. with AIR-NETwork days). Furthermore, with respect to dissemination of research findings, scientists found it hard to write in a concise and non-specialist way; therefore AIRNET provided guidelines, and also contracted science journalists to help. And because effective communication between stakeholders also appeared crucial; AIRNET contracted a communication agency to help to increase its effectiveness in that direction. Finally, organisation of a network requires considerable planning; there was no “recipe” available on how to optimally run a network. Knowing WHAT you wish to achieve is not sufficient, one should also know HOW you wish to achieve it. In fact many participants also recognized that science-policy interface must be considered as a continuous (learning) process rather than a static consultation type of interaction mechanism. It also appeared difficult to get all “players” actively involved; therefore defining a clear task and responsibility seems of crucial value. Despite the learning process during the lifetime of the project, AIRNET has been greatly appreciated for its efforts to have structurally built such a network in Europe.

REFERENCES

AIRNET News Issue 2. October 2003. http://airnet.iras.uu.nl.

Brunekreef, B., and Holgate, S.T. 2002. Air pollution and health. *Lancet* 360:1233-1242**.**

Dybing, E., and Totlandsdal, A.I. 2004. Air pollution and the risks to human health – a toxicological perspective. http://airnet.iras.uu.nl

Fudge, N., Totlandsdal, A.I., and Sanderson, E. 2003. AIRNET stakeholder survey: a report of end-users' air pollution and health information needs. http://airnet.iras.uu.nl.

Ginsburg, E.O., and Cowling, E.B. 2003. Future directions in air quality, science, policy and education. *Environ. Intern.* 29:125-135.

Hoek, G., and Katsouyanni, K. 2004. Air pollution and the risks to human health – epidemiology. http://airnet.iras.uu.nl.

Hurley, F., and Sanderson, E. 2004. Air pollution health impact assessments – an introduction. http://airnet.iras.uu.nl.

Janssen, N., and Sanderson, E. 2004. Air-pollution exposure assessment. http://airnet.iras.uu.nl.

Matanoski, G. 2001. Conflicts between two cultures: implications for epidemiological researchers in communicating with policy makers. *Am. J. Epidemiol.* 154:S36-S42.

Samet, J.M., and Lee, N.L. 2001. Bridging the gap: perspectives on translating epidemiological evidence into policy. *Am. J. Epidemiol.* 154:S1-S3.

Van Bree, L., Fudge, N., and Tuomisto, J.T. 2004. Air pollution and the risks to human health – Science/Policy Interface. http://airnet.iras.uu.nl.