Report of an Expert Panel to Review the Socio-Economic Models and Related Components Supporting the Development of Canada-Wide Standards for Particulate Matter and Ozone

To

The Royal Society of Canada
La Société royale du Canada

At the request of a multi-stakeholder Sponsors Group representing the Canadian Council of Ministers of the Environment, for the Federal/Provincial/Territorial Canada-Wide Standards Development Committee for Particulate Matter (PM) and Ozone and its Core Advisory Group of industrial and non-governmental stakeholders, including environmental, health and aboriginal organizations; and the Aluminium Association of Canada, Canadian Association of Petroleum Producers, Canadian Electricity Association, Canadian Foundry Association, Canadian Gas Association (through GRI Canada), Canadian Petroleum Products Institute, Canadian Manufacturers & Exporters, Canadian Steel Producers Association, Cement Association of Canada, and Forest Products Association of Canada.

"studiis eodem diversis nitimur"
"different paths, one vision"

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And with the Support of a Technical Secretariat comprising:

The Centre for Research in Earth and Space Technology (CRESTech)
An Ontario Centre of Excellence
and
The Network for Environmental Risk Assessment and Management (NERAM)

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Expert Panel to Review the Socio-Economic Models and Related Components Supporting the Development of Canada-Wide Standards for Particulate Matter and Ozone

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June 12, 2001

William Leiss, President
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283 Sparks Street
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Dear Dr. Leiss:

We are pleased to enclose a copy of the report entitled: *A Review of the Socio-Economic Models and Related Components Supporting the Development of Canada-Wide Standards for Particulate Matter and Ozone.* The report has been prepared by the Expert Panel established by the Committee on Expert Panels of the Royal Society of Canada, in response to a request from a multi-stakeholder group of sponsors identified on the title page and representing a number of industrial, governmental and non-governmental organizations. Our report addresses the questions that were posed in the Terms of Reference which were accepted by the Panel on December 10, 1999 at our first meeting, which included a public session with the sponsors and other interested parties.

The enclosed report responds to the request by carefully considering the documentation that we were provided and by offering our critical appraisal of the approaches taken by the Canada-Wide Standards Development Committee. Our efforts proved to be challenging because of the scope and dynamic nature of the Development Committee task. At times we were frustrated by the moving and somewhat ill-defined target that we found ourselves attempting to critique. Despite these challenges, we are satisfied that we have provided a beneficial and constructive critique of the Cost-Benefit Analysis that was done for the Canada-Wide Standards Development Committee for Particulate Matter (PM) and Ozone. Our conclusions and recommendations are framed in light of the enormous import of these standards and the level of analysis that we believe such important standards warrant.

We are happy to note that the enclosed report represents the consensus view of the Panel. Our consensus was not reached without extensive and sometimes intense debate. These discussions reflect the healthy discourse that should be expected in addressing a subject involving so much uncertainty. Accordingly, some members may favour stronger statements in some areas and more cautious statements in others, however, all have agreed to the wording that has been adopted in the report. The Expert Panel has agreed that every member should be free to express his or her own individual interpretations and points of difference freely. The RSC Committee on Experts Panels has chosen wisely for us to have had a panel of this size and diversity of expertise and opinion and yet remain able to work in such a highly collegial and collaborative manner. I would like to thank all members of the Panel for the considerable energy and time they devoted to the completion of this report.

This is a large and complex topic. We were able to obtain and analyze the information needed largely because of the able support provided to the Panel by the technical secretariat drawn from CRESTech and NERAM. We are indebted to them for their steadfast assistance. We are particularly grateful to our technical writer, Lorraine Craig, for assuring that our thoughts read so well.

We trust that the Canada-Wide Standards Development Committee for Particulate Matter (PM) and Ozone and the multi-stakeholder groups involved will find our recommendations useful in future analyses of Canada-Wide Standards for PM and ozone.

Yours sincerely,

Steve E. Hrudey, PhD, PEng
Chair, RSC Expert Panel on CWS for PM and Ozone

Prefatory Note

In February of 1999, a multi-stakeholder group of sponsors identified herein, approached the Royal Society of Canada with a request to commission an Expert Panel to review and report on the socio-economic models and related components supporting the development of Canada-wide standards for particulate matter and ozone. The Society agreed to do so and the Committee on Expert Panels undertook the task of screening and selecting the individuals for panel service whose names now appear as the authors of this report.

The report, entitled *A Review of the Socio-Economic Models and Related Components Supporting the Development of Canada-Wide Standards for Particulate Matter and Ozone*, represents a consensus of the views of all of the panelists whose names appear therein. The Committee wishes to thank the panel members and panel chair, the peer reviewers, and the technical secretariat for completing this very important report within a relatively short period of time.

The Society has a formal and published set of procedures, adopted in October 1996, which sets out how Expert Panel processes are conducted, including the process of selecting panelists. Interested persons may obtain a copy of those procedures from the Society. The Committee on Expert Panels will also respond to specific questions about its procedures and how they were implemented in any particular case.

The terms of reference for this Expert Panel are reproduced elsewhere in this report. As set out in our procedures, the terms are first proposed by the study sponsor, and accepted provisionally by the Committee. After the Panel is appointed, the terms of reference are reviewed jointly by the panelists and the sponsor; the panelists must formally indicate their acceptance of a final terms of reference before their work can proceed, and these are the terms reproduced in this report.

The Panel first submits a draft of its final report in confidence to the Committee, which arranges for another set of experts to do a peer review of the draft. The peer reviewer comments are sent to the Panel, and the Committee takes responsibility for ensuring that the panelists have addressed satisfactorily those comments. The Panel's report is then released to the public without any prior review and comment by the study sponsor. This arm's-length relationship with the study sponsor is one of the most important aspects of the Society's Expert Panel process.

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The Sponsors Group which requested this Report and proposed the terms of reference is composed of the following parties: Canadian Council of Ministers of the Environment, for the Federal/Provincial/Territorial Canada-wide Standards Development Committee for Particulate Matter (PM) and Ozone and its Core Advisory Committee of industrial and non-governmental stakeholders, including environmental, health and aboriginal organizations; and the Aluminium Association of Canada, Canadian Association of Petroleum Producers, Canadian Electricity Association, Canadian Foundry Association, Canadian Gas Association (through GRI Canada), , Canadian Pulp and Paper Association, Canadian Petroleum Products Institute, Canadian Manufacturers & Exporters, Canadian Steel Producers Association, Cement Association of Canada, and Forest Products Association of Canada.

The Technical Secretariat, comprising jointly the Centre for Research in Earth and Space Technology (CRESTech) and the Network for Environmental Risk Assessment and Management (NERAM), administered this panel project on behalf of the Society. The Society wishes to acknowledge the expert assistance of Dr. Dan McGillivray (Director, Business Development & Technology Transfer - Earth Systems, CRESTech) and his collaborators in carrying out this task, and the role of Sandy Jackson, the Society's administrative assistant, in maintaining our liaison with the Technical Secretariat.

Inquiries about the Expert Panel process may be addressed to the Chair, Committee on Expert Panels, Royal Society of Canada.

Dr. Geoffrey Flynn, FRSC
Chair, Committee on Expert Panels

on behalf of the Committee Members for this Panel:

Christopher Garrett, FRS, FRSC, University of Victoria
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June 20, 2001

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Acronyms and Abbreviations

AQVM	Air Quality Valuation Model
ASEP	Atmospheric Science Expert Panel
BS	black smoke
CB	cost-benefit
CBA	cost-benefit analysis
CCME	Canadian Council of Ministers of the Environment
CHA	cardiac hospital admission
CO	carbon monoxide
COH	coefficient of haze
COI	cost of illness
COPD	chronic obstructive pulmonary disease
CWS	Canada-Wide Standards
EPA	U.S. Environmental Protection Agency
ERV	emergency room visit
GHG	greenhouse gas
GVRD	Greater Vancouver Regional District
MT	megatonne
NH_3	ammonia
NOx	nitrogen oxides
PM	particulate matter (both PM_{10} and $PM_{2.5}$)
PM_{10}	particulate matter less than or equal to 10 microns in diameter
$PM_{2.5}$	particulate matter less than or equal to 2.5 microns in diameter
QALY	quality adjusted life years
RADM/RTP	Regional Acid Deposition Model with a Regional Particulate Matter module
RDIS	Residual Discharge Information System
RCM	Regional Climate Model
RHA	respiratory hospital admission
SO_2	sulfur dioxide
TAF	Tracking and Analysis Framework Model
TB	transboundary
TPM	total particulate matter
TSP	total suspended particulates
UAM	Urban Airshed Model
UV	ultraviolet
VOC	volatile organic compound
VSL	value of a statistical life
VSLY	value of statistical life year
WQC	Windsor-Quebec corridor
WTA	willingness to accept
WTP	willingness to pay

Glossary of Terms

accounting stance defines the scope of a proposal being considered and sets out the boundaries for the assessment of costs and benefits. These boundaries can be geographical, temporal and sectoral.

abatement is the reduction of the degree or intensity of emissions or pollutants.

AERCo$t is a model developed for Environment Canada for the purpose of estimating costs of air pollution control options.

aerosols are suspensions of solid or liquid particles in air.

Air Quality Valuation Model (AQVM) is a spreadsheet model owned and controlled by Environment Canada and Health Canada to value the human health and welfare benefits (referring to the value of the reduction in adverse impacts) associated with changes in Canada's ambient air quality

anthropogenic sources produce both gaseous and particulate emissions as a result of human activity such as fossil fuel combustion in electrical power plants, automobiles, industrial boilers and residential heating.

background levels of PM and ozone are the natural concentration that would result in the absence of anthropogenic emissions. PM can be produced directly from natural sources such as forest fires, blown dust, sea spray and emissions from trees. Ozone is a secondary product of the interaction of VOCs, NO_x, and sunlight, all of which have natural sources. The stratosphere is also a source of tropospheric ozone.

baseline in economic analyses refers to the health, environmental and economic conditions that occur in the absence of a proposed policy intervention.

biogenic emissions of PM are sources of low vapour pressure organic compounds that can condense and can also include biological material such as pollen.

box models are atmospheric transport models, often with very complex chemistry that treat the emissions of natural and anthropogenic species into the atmosphere and their subsequent reactions under sunlight as if they occurred in a chemical reactor, i.e. under isolated conditions. They are useful for isolating important chemical processes and can qualitatively simulate conditions appropriate to urban pollution. They can be modified in a simple manner to make some allowance for transport or exchange of air from outside the reactor.

Canada-Wide Standards (CWS) are established under the Environmental Harmonization Accord of the Canadian Council of Ministers of the Environment (CCME) and its Standards Sub-Agreement. For $PM_{2.5,}$ the CWS to be achieved by year 2010 is 30 micrograms per cubic metre, 24 hour averaging time, based on the 98^{th} percentile annual value averaged over three consecutive years. For ozone, the CWS to be achieved by 2010 is 65 ppm, 8 hour averaging time, based on the 4^{th} highest annual measurement, averaged over three consecutive years.

chemokines are a group of proteins that attract white blood cells. The chemokines are involved in a wide variety of acute and chronic types of inflammation, infectious diseases, and cancer.

coagulation refers to the aggregation of suspended particles in air, water or bodily fluids.

conjoint analysis is the application of design of experiments to obtain individual consumer preference information. Because the results provide information on individual preferences, it can be used to construct measures of value for attributes.

contingent valuation method is a survey-based economic valuation method that is often used to quantify in dollar terms the value of an environmental quality or health status change.

cost of illness measures include only medical costs and lost income as a proxy for work loss and thus do not reflect the total welfare impact of an adverse health effect.

cost-benefit analysis is an economic technique applied to public decision-making that attempts to quantify in dollar terms the advantages (benefits) and disadvantages (costs) associated with a particular policy.

cost-effectiveness analysis aims to determine the least expensive way of achieving a given environmental quality target, or the way of achieving the greatest improvement in some environmental target for a given expenditure of resources.

cytokines are soluble substances secreted by cells, which have a variety of effects on other cells, e.g. Interleukin 1 (Il-1).

damage function approach (DFA) is the overall approach used in AQVM to estimate the monetary value of changes in health and welfare effects associated with changes in ambient levels of air pollution. The DFA involves up to a five step process: (1) changes in emissions, by type and location for a policy or scenario are determined (2) change in emissions is translated into changes in ambient air pollution concentrations (3) changes in ambient air pollution concentrations are translated into changes in human health and welfare impacts using concentration-response functions (4) human health and nonhealth effects are assigned economic values (5) benefits are computed and aggregated over the different impacts, locations and time periods.

discounting is a method used by economists to determine the dollar value today of a project's future costs and benefits. This is done by weighting money values that occur in the future by a value less than 1, or "discounting" them. Because environmental decision-makers are increasingly forced to evaluate policies with costs and benefits that will be spread out over tens -- perhaps hundreds -- of years, discounting is used to help evaluate the value of measures that deal with problems such as stratospheric ozone depletion, global climate change, and the disposal of low- and high-level radioactive wastes.

dispersion modeling is based on the use of Gaussian plume models which use analytically based solutions that calculate the transport of inert tracers from a point source by diffusive processes. They are not able to readily incorporate the effects of vertical wind shear nor include chemical reactions. They are normally used for investigating the distribution of pollutants from stacks from power plants and such point sources and can be modified to handle multiple sources.

distributional effects are the net costs and benefits of a regulatory policy across the population and economy, divided up in various ways -- for instance, by income groups, race, gender, and industrial sector. Distributional effects of a regulation may also span over several generations.

emission factors for stationary sources are based on the relationship between the amount of pollution produced and the amount of raw material processed or burned. For mobile sources, emission factors are based on the relationship between the amount of pollution produced and the number of vehicle miles traveled. By using the emission factor of a pollutant and specific data regarding quantities of materials used by a given source, it is possible to compute emissions for the source. This approach is used in preparing an emissions inventory.

emission inventory is an estimate of the amount of pollutants emitted into the atmosphere from major mobile, stationary, area-wide, and natural source categories over a specific period of time such as a day or a year.

general equilibrium theory demonstrates the advantage of looking beyond first-stage effects. In the context of climate policy, it implies that the various parts of an economic system are interrelated, and the net effect of an action may be markedly different from the initial effect.

harvesting refers to air pollution exposures advancing death by only a few days or weeks. It is a factor affecting the interpretation of the response coefficients obtained from daily mortality time-series studies.

hedonic pricing approach derives values by decomposing market prices into components encompassing environmental and other characteristics through studying property values, wages and other phenomena. The premise of the approach is that the value of an asset depends on the stream of benefits derived, including environmental amenities.

household material soiling is a non-health endpoint in AQVM that estimates the economic effects to households from PM soiling based on studies of household cleaning expenditures.

IL-8 is a glycoprotein secreted by a variety of leukocytes (cells that help the body fight infections and other diseases. Also called white blood cells (WBCs)) which have effects on other leukocytes.

isoprene (C_5H_5) is a very reactive organic compound because of its double bonds, whose light and temperature sensitive emissions from plants can be comparable to anthropogenic organic emissions. Because of its high reactivity its degradation products can react with NO to produce important amounts of ozone.

multi-attribute analysis or multi-criteria analysis is a method of evaluating trade-offs over various attributes of a situation.

Monte Carlo analysis is a tool for evaluating uncertainty and variability. The basic goal of a Monte Carlo analysis is to characterize, quantitatively, the uncertainty and variability in estimates of exposure or risk. A secondary goal is to identify key sources of variability and uncertainty and to quantify the relative contribution of these sources to the overall variance and range of model results.

net present value is the current value of net benefits (benefits minus costs) that occur over time. A discount rate is used to reduce future benefits and costs to their present time equivalent.

neutrophils are a type of white blood cell that defends the body against foreign matter.

nitrogen dioxide (NO_2) has both natural and anthropogenic sources largely as a result of combustion of fuels with air. It can damage trees and lead to acid rain, which can harm aquatic and terrestrial ecosystems through effects on lakes, streams and soils and also corrode exposed materials. In the presence of sunlight and volatile organic compounds, NO_2 can contribute to the formation of ground-level ozone,and other photochemical reaction products.

nitrogen oxides (NO_x) are often mentioned in discussions of nitrogen-based air pollution as a reference to both nitric oxide (NO) and nitrogen dioxide (NO_2). In addition to particulates and sulfur dioxide, NO_x is one of the major combustion-related pollutants. They can be oxidized to nitric acid in the air which can react to form to ammonium nitrate which is a fine particulate.

opportunity cost is the value of the best alternative to a given choice, or the value of resources in their next best use. In regard to time, the opportunity cost of time spent on one activity is the value of the best alternative activity that the person might engage in at that time.

ozone at the ground level can arise from the reaction of its precursors nitrogen dioxide, and volatile organic compounds, in the presence of sunlight. It can also result from transport from the stratosphere. Its precursors have both natural and anthropogenic sources. When it arises above background levels it is regarded as a form of air pollution. It is to be distinguished from stratospheric ozone, which has the same chemical formula, but is found 10 to 40 km high in the Earth's atmosphere and protects people from harmful radiation from the sun. Background tropospheric ozone is thought to have increased by a factor of 2-3 in the last 100 years as a result of increasing human emissions of NO_x and VOCs.

ozone precursors are chemicals such as hydrocarbons and oxides of nitrogen, occurring either naturally or as a result of human activities, which contribute to the formation of ozone, a major component of smog.

particulate matter (PM) is a form of air pollution that includes soot, dust, dirt and secondary acidic and organic aerosols. Common terminology uses PM_{10} to refer to all particles less than 10 µm in aerodynamic diameter, and $PM_{2.5}$ to refer to particles less than 2.5 µm in aerodynamic diameter. Coarse PM contains primarily materials derived from the earth's crust, such as soil and minerals. Fine PM, usually the result of anthropogenic activities, contains sulphate, nitrate, ammonium, metals, elemental carbon, and hundreds of different organic compounds.

primary particles are emitted directly into the atmosphere from anthropogenic sources (e.g. combustion generated fine particles, or coarse particles that result from crushing, grinding and erosion).

physical-based models are three-dimensional grid models with meteorological transport processes, complex photochemistry and emissions of gases and PM from natural and anthropogenic sources, designed to calculate the concentrations of chemically reactive pollutants in the atmosphere. They can be run for different spatial and temporal scales from urban to global. The state of the art models simulate the meteorological, physical and chemical processes that affect pollution concentrations. Older models often calculate ozone and PM separately, and used meteorology from other models. The most sophisticated models can be run in a nested fashion incorporating physical, chemical and meteorological processes on several different scales concurrently.

present value is the value today of a sum to be paid or collected in the future to buy a good or service.

QALYs are a composite measure of the number of years of life gained or lost by a particular decision, but weighted according to the expected quality of life during those years, and to this added measures of the improvement in quality of life (say from reduced morbidity). Years of poor health are weighted as a fraction of years of good health. QALYs provide a metric of preferences over alternative health states that allows one to determine if procedure A is more effective at meeting a chosen standard than procedure B.

rollback approaches assume changes in ambient concentrations of air pollution concentrations are directly proportional to changes in precursor emissions.

Residual Discharge Information System (RDIS) is a microcomputer-based software package that allows for the compilation, maintenance and reporting of air emissions data, by regions, provinces and for Canada. The system is designed to store information from all major Canadian emission sources, of man-made and natural origin. When source data on specific pollutants is not available emission discharge factors are used to estimate the emissions. These factors indicate the rate at which a contaminant is released into the environment as the result of a given activity.

secondary particles are formed through chemical reactions involving gases such as sulphur dioxide (SO_2), nitrogen oxides (NO_x), volatile organic compounds (VOCs) and ammonia (NH_3) and other particles and gases in the atmosphere.

sensitivity analysis examines the effect of parameter uncertainty by modifying the parameter value of a single uncertain variable. A series of plausible alternative values for the parameters are introduced into the mathematical model and the risk estimate is recalculated using the substituted parameter values.

socio-economic analysis (SEA) includes a wide variety of social and economic analysis methods, of which cost-benefit analysis (CBA) is one example.

source apportionment or source attribution allows for the identification (quantitatively and qualitatively) of contributing sources to support the development of atmospheric models and air quality management strategies.

source-receptor modeling starts with observed particle concentrations at a receptor (i.e. at a monitoring site) and seeks to apportion the observed concentrations between several source types based on knowledge of the compositions of the source and receptor materials.

stakeholders are citizens, environmentalists, businesses, and government representatives that have a stake or concern about how air quality is managed.

stratosphere is the layer of the Earth's atmosphere above the troposphere and below the mesosphere. It extends between 10 and 50 km above the Earth's surface and contains the ozone layer in its lower portion. The stratospheric layer mixes relatively slowly; pollutants that enter it may remain for long periods of time.

sulfur dioxide (SO_2) is a strong smelling, colorless gas that is formed by the combustion of fossil fuels. Power plants, which may use coal or oil high in sulfur content, can be major sources of SO_2. SO_2 and other sulfur oxides contribute to the problem of acid deposition.

sulfur oxides include pungent, colorless gas (sulfur dioxide), sulfates and fine particles formed primarily by the combustion of sulfur-containing fossil fuels, especially coal and oil.

tax interaction effect can occur when environmental policies, such as emission taxes or permits, and the conventional tax system interact. This effect is the cost of the overall reduction in employment and investment caused by environmental policies, which exacerbate the distortionary effects of pre-existing taxes on labor and capital.

terpenes are a naturally occurring organic compound, of the general empirical formula, $C_{10}H_{16}$, biologically built from a naturally occurring "monomer" called isoprene, C_5H_8, which is found as a volatile oil in plants. They are reactive organic compounds, whose temperature sensitive emissions are implicated in the production of ozone.

Total Suspended Particulate (TSP) is a gravimetric measure of particles of solid or liquid matter **--** such as soot, dust, aerosols, fumes, and mist where upper size limit varies from approximately 20 to 50 microns in size, depending on wind speed and distance from the source.

troposphere is the layer of the Earth's atmosphere nearest to the surface of the Earth. The troposphere extends outward about 5 miles at the poles and about 16 km at the equator.

Urban Airshed Model (UAM) is a three-dimensional photochemical grid model with meteorological transport processes, designed to calculate the concentrations of both inert and chemically reactive pollutants in the atmosphere. It simulates the physical and chemical processes that affect pollution concentrations.

value of a statistical life (VSL) method estimates the dollar value of a given reduction in risk, in reference to an individual's willingness to pay to reduce that risk.

visibility is a measurement of the ability to see and identify objects at different distances. Visibility reduction from air pollution is often due to the presence of sulfur and nitrogen oxides, as well as particulate matter.

Volatile Organic Compounds (VOCs) are carbon-containing compounds that evaporate into the air (with a few exceptions) with both natural and anthropogenic sources. VOCs contribute to the formation of smog and/or may themselves be toxic. VOCs often have an odor, and some examples include gasoline, alcohol, and the solvents used in paints.

Willingness To Pay (WTP) is one form of economic value associated with a change in quality or quantity of a good or service. WTP is a theoretical measure of the value an individual places on the good or service, or in the case of health effects, reflects the value of avoiding an adverse health effect based on an individual's willingness-to-pay (WTP) for risk reduction. By summing many individuals WTP to avoid small increases in risk over a large sample, the value of a statistical premature death avoided can be inferred. This valuation is expressed as is dollars per mortality avoided or value of a statistical life (VSL) even though the actual valuation represents small changes on mortality risk experience by a large number of people. The VSL method estimates the dollar value of a given reduction in risk, in reference to an individual's WTP to reduce that risk. WTP is often based on wage-risk studies, which derive WTP values from estimates of the additional compensation demanded in the labour market for riskier jobs, or from contingent valuation (CV) studies which directly solicit WTP information from personal interviews.

with-without principle In order to concentrate on the benefits and costs of the issue at hand, CBA should measure the projected benefits and costs with the change (defined as one or more specific policy 'options'), compared to the benefit and costs without the change. With/without analyses require the definition of a time path or baseline regulatory structure (including baseline expectations for technical change impacts on costs of emissions compliance, ambient air monitoring and other expected impacts).

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Executive Summary

Canada-Wide Standards (CWS) for particulate matter (PM) and ozone may be the most ambitious environmental standards ever proposed in Canada. They have attracted considerable attention and debate. This report addresses the validity of the socio-economic modeling aspects of the Canada-Wide Standards development process.

Socio-economic considerations are addressed in one of eight principles underlying the development and attainment of CWS, according to a CWS sub agreement signed by the Canadian Council of Ministers of the Environment (CCME). Principle 3.1.7 states that

> *"measures to attain agreed-upon Canada-Wide Environmental Standards will be determined in a sustainable development context, recognizing environmental and* ***socio-economic*** *considerations"*.

A CCME Framework for Socio-economic Analyses in Setting Environmental Standards (CCME, 1998) describes procedures and information requirements for socio-economic assessments of potential or proposed environmental standards. This Framework states that while it may not be possible or necessary to carry out all of the analytical steps because of time, data or resource constraints, a partial assessment can produce information that is useful for policy deliberations. The Framework also notes that socio-economic findings are not intended to be prescriptive concerning decisions about environmental standards because other input factors such as toxicity, epidemiology, ecological consequences and geographical distribution of effects and other equity considerations are also necessary and important to an informed choice with respect to standard setting. Socio-economic considerations are also specified under Government of Canada Regulatory Policy requiring federal regulatory authorities to demonstrate that the benefits of regulatory requirements are greater than their costs. Regulatory authorities must "*ensure that the benefits outweigh the costs to Canadians, their governments and businesses. In particular, when managing risks on behalf of Canadians, regulatory authorities must ensure that the limited resources available to government are used where they do the most good*" (Government of Canada, 1999). This implies not only that benefits should be greater than costs, but that benefits minus costs, or net benefits are to be maximized, which means an attempt should be made to make standards efficient.

The objective of the Expert Panel process was to provide an independent, expert review and critique of the socio-economic (SEA) analyses – in this case a cost-benefit analysis (CBA) – conducted in developing the Canada-Wide Standards on PM and ozone. Through a review of the models and associated data and assumptions used in the analyses, the Panel was asked to produce a report to address the following questions:

a. What are the strengths, merits, limitations, gaps and the degree of uncertainties of the proposed approaches, models, and their inputs and outputs?

b. By what means could the models and analytical approaches be improved, so as to minimize uncertainties and maximize the relevance, reliability and utility of outputs?

c. What other approaches and/or tools could be used to conduct these analyses?

The benefits and costs associated with Canada-Wide Standards for PM and ozone are highly uncertain and controversial. Uncertainties are associated with each step in the analysis of benefits and costs - including the link between emission reductions and ambient air quality, the extent to which human health and the environment are affected by changes in ambient levels of pollutants, the economic values (as measures of preferences) associated with improvements in environmental and human health, accuracy of the emissions inventory and projections of what this inventory and other factors will be in a future baseline, and the scope and magnitude of economic costs associated with emission reductions, both to industry and to society.

With uncertainties so pervasive, analysts are required to make many choices and assumptions in estimating costs and benefits. For example, while it is clear that the epidemiological association between PM and excess mortality is consistent and robust, there are many remaining gaps in current understanding of the relative toxicity of PM components and gaseous co-pollutants and the magnitude of potential life-shortening effects. These uncertainties introduce possible biases into the estimation of PM-related health benefits.

CONCLUSIONS.

The Panel draws the following major conclusions from its review of the CBA undertaken for the development of CWS for PM and ozone and from the academic and policy literature relevant to this topic. In drawing these conclusions, the Panel views the use of a structured approach to the examination of costs and benefits as a positive development in Canadian regulatory policy analysis.

1. **The CWS Socio-Economic Analysis (SEA) was in fact limited to a cost-benefit analysis (CBA).**

Because the CWS implementation of the SEA process was judged by the Panel to be limited to a CBA it was reviewed as such. CBA is just one of many available decision support tools. The requirements of CBA in the CWS process depends on whether the purpose is to select an ambient air quality standard or to guide and evaluate the implementation process. The extent to which the results of the CBA can inform the CWS decision process is limited for various reasons including the following:

- Provinces are required to establish implementation plans to ensure that CWS will be met. Therefore, when doing a prospective CBA, the measures to be implemented (i.e., revised emission control regulations) are necessarily undetermined, as are the compliance levels for the revised control regulations that will be achieved in various provinces.

- The distributional impacts of both costs and benefits have not been assessed.

The Panel acknowledges that the CWS Development Committee for PM and Ozone describes the cost estimation as *"preliminary and, in some instances, a cursory analysis used to provide a macro level order of magnitude perspective on the costs associated with the various optional levels for PM and ozone CWSs"* and notes that caution should be exercised in their interpretation (Canada-Wide Standards Development Committee for

PM and Ozone, 1999). Deficiencies in Canadian data and modeling capabilities and limited time and resources restricted the scope of the analysis that could be undertaken for the CWS process.

2. **Credible CBA should be conducted to support the development of Canada-Wide Standards.**

The Panel recommends that CBA be used to inform decision-makers about the projected costs and anticipated benefits of CWS. CBA needs to be designed to distinguish between the costs and benefits of meeting alternative PM and ozone standards within the limits of current science. There are potential overlaps in the estimation of costs and benefits for PM and ozone because emission control strategies will impact both PM and ozone levels and it is not clear which components of the air pollution mix are responsible for the various health effects. These uncertainties in the CBA need to be clearly communicated. At its best, CBA provides the decision-maker with a systematic identification, estimation, and measure of uncertainty of monetary values for the relevant costs and benefits of interest to decision-makers and stakeholders. To be fully informative, the CBA results provided to stakeholders and decision-makers need to adequately analyze and explain the major sources of uncertainty in the inputs of the CBA model projections, and their likely effect on model outputs.

3. **In view of the importance of the proposed regulatory decision, the CBA performed for the CWS for PM and ozone is deficient in relation to the state-of-the-art for CBA.**

If the CWS CBA was intended to provide an adequate basis for balancing costs and benefits and for influencing where the CWS should be set, this CBA was not up to the task. If the objective was strictly to confirm that costs were not exorbitant for CWS that were deemed to be both technically feasible and associated with some substantial benefits, then this CBA provided contributions towards those judgments.

When judged against the elements of process and structure of CBA required for credibility as indicated in conclusions 4 and 5 below, the Panel finds that the CWS CBA does not satisfy these requirements and does not meet a reasonable level of quality for a CBA to support a decision of this import. While the CWS CBA has some value as a scoping analysis and provides a limited degree of guidance for decision-makers, it requires substantial improvement to meet the criteria for credibility.

4. **The Process for using CBA in CWS needs improvement.**

A Discussion Paper on Particulate Matter (PM) and Ozone Canada-Wide Standard Scenarios for Consultation prepared by the Canada-Wide Standards Development Committee for PM and Ozone (May 1999) states that "*in selecting PM and ozone CWS level scenarios for stakeholder considerations, an attempt was made to balance the anticipated benefits of improved air quality with the technological feasibility and costs of achieving those improvements*". But, it is apparent that the standard CBA procedure of comparing incremental benefits for tighter standards with incremental costs was not done for the CWS CBA process. Timelines for the analysis were exceedingly short, and the CBA effort appeared to be underfunded, resulting in short-cuts that substantially reduced the credibility of the analysis. Reporting and communicating the CBA results was also ineffective, particularly in terms of conveying a clear understanding of what was done and why it was done as it was.

5. **The Panel has identified several CBA elements of primary concern that require attention in order to ensure the credibility of CWS CBA.**

The elements of primary concern for assessing credibility of CBA are the following:

- accuracy of emissions inventory data
- accuracy of cost estimates
- use of state-of-the-art air quality models
- sufficiency of air quality monitoring
- use of reasonable baseline assumptions for regulatory regime
- inclusion of well-documented environmental conditions

- inclusion of demographics in the CBA
- adequate consideration of economic growth
- selection of dose-response functions based on current weight of evidence
- selection of valuation functions based on current weight of evidence
- explicit expressions of uncertainty (measurement, model and statistical)
- compatibility of scenarios with the form of standard (8 hour and 24 hour averaging times)
- inclusion of distributional analysis of costs and benefits (identification of sensitive subgroups, affected sectors)
- internal consistency of analyses (linking costs with benefits consistently)
- discussion of non-quantifiable endpoints
- explanation for the choices of benefits and costs included

Performing CBA that meets these requirements will involve substantial investment on a continuing basis. Critics of the adequacy of the CWS CBA should support the generation of a knowledge base adequate to perform credible CBA. Generation of that knowledge base will require substantial investment of money, infrastructure and expertise.

The Panel has outlined its view on the limitations of the CBA undertaken for the development of CWS for PM and ozone. Given those concerns, the Panel provides the following two conclusions on the measures of benefits and costs as calculated within the CWS CBA.

6. **As in all CBA the estimates of benefits and costs are uncertain.**

Emerging analyses (in particular, of the tax interaction effect and the value of a statistical life) suggest that the costs associated with reducing emissions may be underestimated and the human health benefits overestimated. However, there are additional uncertainties that temper the impact of these emerging studies on the CBA. Most notably, the cost analysis performed in the CWS process is based on engineering estimates (resulting in overestimated costs) and the benefit measures do not include ecosystem effects (potentially large, but highly uncertain). The direction of the bias in net benefits depends on the weight placed on these factors.

7. **The overall direction of the errors in benefits estimation is undetermined.**

The premature deaths reduced and broader range of health effects avoided by reducing PM ambient levels to the CWS PM standard are likely underestimated in the CWS analysis. However, the dollar value estimates for mortality reductions (based on value of statistical life, VSL) are very likely overestimated. The overall effect of these potential biases on the benefits realized from emission reductions is not clear from the current evidence.

RECOMMENDATIONS

Based on these conclusions, the Panel makes the following recommendations.

1. **Capacity Building**

Given the identified deficiencies in the CWS CBA, the Panel recommends the following:

- Canada should build a capability for conducting CBA for CWS by improving emissions inventories, air quality modeling capability, air quality monitoring networks, socio-economic modeling, human health data gathering and developing economic analyses of health-environment interactions. This capacity building will require long term financial support to build the infrastructure as well as government and industry commitment to making these improvements.

- Data and models should undergo continuing development and refinement with reporting and documentation at periodic intervals that are integrated within the timeframe for decision-making. This includes particularly the Air Quality Valuation Model (AQVM) used to estimate the health benefits of air quality improvements. As detailed later in this report, the AQVM may already be out of date in its choice of dose-response functions for estimating mortality risk reductions and for valuation of this health endpoint. The decision to update AQVM to include more recent work depends on the criteria of study inclusion, e.g. degree of peer review, number of confirmatory studies, etc. The Panel recommends that such criteria used to develop the AQVM be reviewed and updated, as necessary.

- Improve Canadian capacity for air quality modeling. Collaborations with other North American agencies and research groups should be encouraged and supported with long term funding.

- Inclusion of risk-risk tradeoffs (estimation of damages associated with risks from pollutants that increase as a result of the pollutants of interest being reduced, e.g., UV-B radiation increasing as a result of ozone concentrations being decreased).

- All CBA model specifications and input values (e.g. risk coefficients, health event valuations) used for the purposes of regulatory decision-making should be fully transparent and readily accessible to all interested stakeholders and researchers.

- An explicit procedural and consultative framework should be developed for CBA to inform the decision process. Informed decisions require dialogue and consultation between decision-makers, stakeholders and CBA analysts in an open, transparent process. The decision process should require consideration of results of CBA along with other inputs to the decision.

- Funding should be allocated, and roles and responsibilities within the CWS process should be defined - - including an external expert advisory body to review approaches, progress, etc.

- Formal guidelines for considering evidence and making and communicating decisions should be developed.

2. Communication

Improved two-way communication concerning the assumptions, limitations and uncertainties associated with the methods and results of CBA is needed between analysts and policy makers and between policy makers and the public. Clear communication of the conceptual underpinnings and limitations of valuation techniques and the interpretation of the results of cost and benefit studies is needed to correct prevailing misconceptions about the conduct and interpretation of these studies.

3. Cost-Benefit Analysis and Other Types of Socioeconomic Analyses

Socio-economic analysis (SEA) includes a wide variety of social and economic analysis methods, of which cost-benefit analysis (CBA) is one example and is typically the foundation for other socio-economic analyses. However, the Panel recommends that CBA be conducted separately from broader socio-economic analyses, including plant closures, unemployment, regional economic impacts, competitiveness, or inflation for broad-based rules. Such analyses often ignore labor and capital mobility and are not commensurate with values used in CBA and therefore, when provided alongside CBA estimates, the results of broader SEA models may give rise to double counting of benefits or costs. If effects are expected to be borne disproportionately by only a few sectors, these types of analyses are useful, but they should be presented as contributions to the assessment of the distribution of impacts. A more promising, but more resource intensive, approach is the expansion of the CBA to a general equilibrium analysis to capture the costs of the tax interaction effect (see Section 7.4). As for competitiveness analysis, shifts to imports may have positive environmental effects that would need to be taken into account and in this sense, a broader SEA or general equilibrium framework would be useful.

4. Cost-benefit Framework for Analysis of Environmental Quality Regulations

The Panel endorses the use of a cost-benefit framework for the analysis of environmental regulation while recognizing the empirical limitations of CBA. The Panel recommends:

- Continued development of methods for accurate assessment of costs and benefits, including methods for the analysis of general equilibrium (including tax interaction) effects and international trade impacts of regulatory change.

- Continued development of and communication regarding alternative decision-making frameworks, including multi-attribute methods, to be used as methods to "triangulate" with traditional CBA.

- Investments in human capital in the area of CBA of environmental regulation so that policy makers and the Canadian public can be confident that cost and benefit measures accurately reflect Canadian values and public preferences as well as Canadian institutional arrangements.

References

Canada-Wide Standards Development Committee for PM and Ozone. 1999. Discussion Paper on Particulate Matter (PM) and Ozone. Canada-Wide Standard Scenarios for Consultation. 99-05-05.

Canadian Council of Ministers of the Environment. 1998. Framework for the Application of Socio-economic Analyses in Setting Environmental Standards. Economic Integration Task Group. Version 7/9/98.

Government of Canada. 1999. Government of Canada Regulatory Policy. Privy Council Office.

1 Introduction

1.1 Panel Terms of Reference

The objective of the Expert Panel process is to provide an independent, expert review and critique of the socio-economic analyses conducted in developing the Canada-Wide Standards (CWS) on ambient particulate matter (PM) and ozone. Through a review of the models and associated data and assumptions used in the analyses, as well as the relevant academic and policy literature, the Panel has produced a report to address the following questions posed in the Panel's Terms of Reference:

a. What are the strengths, merits, limitations, gaps and the degree of uncertainties of the proposed approaches, models, and their inputs and outputs?

b. By what means could the models and analytical approaches be improved, so as to minimize uncertainties and maximize the relevance, reliability and utility of outputs?

c. What other approaches and/or tools could be used to conduct these analyses?

1.2 Complexity of the Problem

"All Models are wrong, but some are useful"

Box (1979)

The Panel recognizes the difficulty and complexity of the task of evaluating socio-economic factors arising from Canada-Wide Standards for particulate matter and ozone. Our critical remarks of the government's analyses in support of these standards should be taken in this context. Some of the dimensions of this complexity include:

- facing multiple dimensions of time, space, character
- the necessity to rely on estimates for most inputs and fundamental parameters, rather than use direct and relatively certain measurements

Even mundane financial analysis involving future predictions cannot be absolute, so it is not surprising that answers to the complex questions in the CWS process are not going to have fully satisfying answers, either. A question as mundane as: *Should an individual use any available cash to pay down their mortgage or invest in a Registered Retirement Savings Plan?* does not have a generalizable answer. Even for a specific individual, the answers that can be generated will depend on a number of forecasting assumptions about uncertain factors like future inflation, interest rates, investment rates of return and tax rates. Once we must tackle something that involves non-monetary costs and costs that are dependent on hypothetical scenarios as well as physical effects that are both uncertain in real terms and in terms of the means for valuing them, we enter a realm of considerably greater complexity. This means that it is easy to find fault with efforts to assess costs and benefits of complex scenarios and there will always be scope for differing perspectives on the choices and assumptions made.

Simple criticism from the Panel would be hollow and would certainly not be helpful unless we can offer viable alternatives. We cannot expect a simple, precise and accurate answer to the analysis for a complex forecast based on enormous uncertainties, such as are involved in the CWS process. Yet, implementing policies that seek to achieve major social benefits at substantial societal costs without a reasonable idea of the range of magnitude of either the costs or the benefits is not responsible public policy.

We must recognize these realities and the further reality that there is no right answer or right way to do the analysis called for in the CWS process. We can identify errors, important omissions and qualifications about how this analysis should be interpreted. We need to explore the advantages and disadvantages of alternative ways of seeking answers. The bottom line is that this type of modeling can and should be only one input to the decision-making process. The inherent limitations of this or any other attempt to forecast future reality mean that we should not allow the answers to any such modeling exercise to dictate ultimate decisions without exercising substantial judgment in the process.

1.3 Organization of the Report

The estimation of costs and benefits associated with proposed air quality standards involves a series of linked steps each with its own conceptual foundation, analytical approaches, assumptions and uncertainties. Briefly, these steps include: i) modeling changes in ambient air quality resulting from reductions in pollutant emissions; ii) estimating avoided health effects; iii) estimating avoided non-health ecological and other welfare impacts; iv) estimating costs of emission reduction; v) economic valuation of avoided health and non-health effects; and vi) balancing costs and benefits. The report discusses the Panel's assessment of the strengths, limitations, gaps and uncertainties associated with each of these steps of the CBA. A summary table of the Panel's assessment of the key limitations, relative uncertainties and recommendations is provided at the end of each chapter. The details of the cost-benefit analysis as described in the CWS documentation are provided, as well as the Panel's interpretation of the analyses. The report provides an overview of the conceptual foundations of CBA and describes other approaches to broadening the scope of CBA. The report is organized as follows:

- Chapter 2 presents an overview of the conceptual underpinnings of CBA as a tool for policy analysis, including its purpose, limitations and methods. Several key components that are common to most credible CBAs are discussed including the with-without principle, damage function approach, measurement of benefits and costs, summary measures of benefits and costs, choice of discount rate and treatment of uncertainty.

- Chapter 3 provides the documentation of the methodology and results of the analyses of costs and benefits that were undertaken for the PM and ozone CWS development process.

- Chapter 4 reviews the Canadian emission inventory and assesses the linear approach used in the CWS CBA to link rollbacks in pollutant emissions with corresponding changes in ambient air quality. The applicability and shortcomings of the linear assumptions used to connect rollbacks in emissions with putative air quality changes is reviewed. The use of physical-based modeling as an alternative to the linear approach used in the CWS CBA is presented.

- Chapter 5 discusses the approach used to quantify avoided cases of premature mortality and morbidity associated with reductions in ambient PM and ozone levels.

The uncertainties underlying the epidemiological studies selected to derive the estimation of avoided health effects associated with improvements in ambient air quality are identified and an alternative approach for estimating avoided mortality is proposed.

- Chapter 6 discusses the assumptions and uncertainties associated with the estimation of avoided non-health environmental impacts in the CWS process as well as those that were not assessed but within the scope of evaluation using existing models. The chapter discusses overall impact of omitted endpoints on the CWS benefits estimate.

- Chapter 7 discusses the conceptual foundations of cost estimation and provides an assessment of the assumptions and limitations of the approach used to estimate costs in the CWS process. Approaches to broadening the scope of the cost analysis are presented.

- Chapter 8 discusses common approaches to valuation of improvements in health and non-health endpoints arising from improvements in air quality and assesses uncertainties in the CWS approach. The implications of recent valuation literature for estimation of mortality benefits are discussed.

- Chapter 9 compares cost-benefit analysis to other methods that have been proposed for assessing evidence for regulatory decision-making, including cost effectiveness analysis and multi-attribute analysis.

- Chapter 10 provides conclusions arising from the Panel's review.

- Chapter 11 provides recommendations arising from the Panel's review.

- Appendix A presents frequently raised concerns with cost-benefit analysis.

- Appendix B summarizes the monetary values assigned to morbidity effects in the benefits assessment model (AQVM) and describes the studies from which they are derived.

- Appendix C indicates concerns raised by stakeholders in written submissions to the Panel and identifies the location of specific responses within the Panel report.

- Appendix D presents a summary table of the Panel's assessment of the CWS approach to CBA including the assumptions, uncertainties and recommendations for alternative approaches.

- Appendix E presents the Terms of Reference for the Expert Panel's task.

1.4 References

Box, G. E.P. 1979. In: Lamer, R.L. and G.N. Wilkinson (Eds.) *Robustness in the Strategies of Scientific Model Building.* Academic Press, New York, NY.

2 Purpose, Capability and Limitations of Socio-Economic Analysis / Cost-Benefit Analysis

Our society has many pressing needs but limited resources with which to address these needs. Demands for environmental quality, health care, employment, education and other services are ever-present. Attaining the appropriate balance between meeting these demands and expending scarce resources is challenging. One way of examining the balance issue is to do cost-benefit analysis. The Canada-Wide Standards approach to determining PM and ozone standards is an example of such a resource allocation process in that trade-offs between the costs of regulation are being weighed against various potential health and environmental benefits.

In economics, cost-benefit analysis (CBA) is a tool that is employed to help assess public policy options by examining the anticipated benefits and costs of the various policy options. The Canada-Wide Standards process has been referred to as "Socio-Economic Analysis." There are many definitions of "Socio-Economic Analysis" ranging from social impact assessment to economic impact assessment.[1] In this report we have focused on the narrower concept of CBA because the Canada-Wide Standards (CWS) approach was largely limited to CBA; a mainstream practice that is highly developed and widely discussed and analyzed in the economics literature.

CBA is an attempt to rigorously define, organize, measure and compare the various benefits and costs arising from a policy change or from a project. As such, CBA is a framework and a tool that is useful in policy analysis. In this section, the conceptual underpinnings of CBA, as well as some of the challenges will be outlined. Chapter 3 addresses the specifics of what was done in the case of the CWS exercise.

[1] See US EPA. Sept. 2000. Guidelines for Preparing Economic Analyses. 240-R-00-003.

2.1 Social Welfare and Cost-Benefit Analysis

The focus of economic analysis is on the efficient use of scarce resources. In other words, economic analysis is seeking an answer to the question: *How can society's resources be put to their best use?* In the present context, economic analysis – CBA – examines the social impact of proposed changes in regulatory standards to examine whether the specific options under consideration are likely to result in a net increase in social welfare, and, if so, by how much. Thus, CBA is a mechanism used to estimate net improvement or reduction in overall social welfare. While there are many concepts of social welfare (e.g. Hargreaves-Heap et al., 1992) in economic analysis social welfare is defined as the sum of individual welfare or well-being. For a specific policy change (or project) each individual is examined to determine if he or she is better off or worse off with the change under analysis versus without that change. These considerations of *better off* or *worse off* include market impacts (e.g. changes in profits for industrial plant owners under a revised air quality standard, changes in consumer satisfaction) as well as non-market impacts (e.g. changes in health states or changes in recreational quality because of improved air quality).

Determining if an individual is *better off* and by how much is a challenging endeavour. CBA employs measures of economic value to reflect whether a person is better off or worse off and by how much. These measures of economic value are based on trade-offs that individuals commonly make or choices that individuals accept that describe their personal preferences. Economic values are determined as the amount that an individual would be willing to pay in exchange for some good or service (or conversely, the amount that they would be willing to accept in compensation to give up some good or service). Often, economic value can be derived from observations of market choice behaviour – an individual is observed to pay $5 for a good and thus it is assumed that the value of that good is at least $5 for that individual. However, economic value can also be observed for choices that are outside of the marketplace. Individuals are observed to make choices to reduce their health risks. This provides information on the value to them of health risk improvements. The key element in CBA is its foundation on individual preferences, and the use of monetary values determined by individual trade-offs in the measurement of individual and social values.

Economists focus on measuring these individual level welfare measures and reporting them in monetary terms because this allows for comparison across benefit and cost categories, allows for assessments of the net benefits (benefit minus costs), and facilitates the process of aggregating the individual benefits and costs into social benefit cost measures.

Aggregating individual benefits and costs in CBA typically weights all individuals equally ($1 of benefit to A = $1 benefit to B). CBA does not evaluate the *distribution* of costs and benefits across various individuals and groups, except to the extent that the analyst can examine who benefits and who loses. CBA focuses on economic efficiency (maximizing social welfare) and treats all individuals as equal with respect to their personal allocation of costs incurred and benefits received. While some theorists (Slesnick, 1999) have described methods that would explicitly weight members of society (e.g., placing higher weights on the poorer members of society in an attempt to increased the benefits of these members relative to richer members of society) this is typically not the approach employed in CBA. However, CBA can be used to describe the incidence of the projected benefits and costs and thus provide useful equity information into policy analysis.

CBA involves the aggregation of individual expected monetary values into measures of overall social welfare. CBA follows the logic of the *compensation principle* which states that: *a policy or project that creates benefits such that the beneficiaries could compensate the losers and still be better off than before the change is social welfare enhancing.* There has been considerable debate in the economics profession regarding the compensation principle; nevertheless, this is the most practical approach for the assessment of social benefits and costs. Boadway and Bruce (1984) and Just, Hueth and Schmitz (1982) discuss the compensation principle and related issues in CBA.

CBA has been employed in the area of environmental policy for five decades. The actual process of conducting a CBA should provide for a transparent analysis and presentation of the range of expected benefits and costs, the monetary values placed on various benefit and cost categories, the estimated incidence of the impacts, the underlying assumptions made in calculating benefits and costs, and various other methodological issues. This transparent accounting system is intended to provide a great deal of information to policy makers. It also serves to point out the deficiencies or weaknesses in data or measurement associated with the issue. A well-constructed CBA will also provide some indication of the variance of the estimated benefits and costs and the uncertainty associated with these measures. Most economists recognize that the information included in CBA is not perfect, but the discipline and rigour of measuring and categorizing this information can provide a great deal of information on the trade-offs involved in the policy or project being considered (Kopp, Krupnick and Toman, 1997).

2.2 Key Components of Cost-Benefit Analysis

There are several key components that are common to most credible CBAs. These include the *with/without principle*, the damage function approach, accounting stance, measurement of costs and benefits, techniques for measuring benefits, the discount rate, summary measures of benefits and costs, and treatment of uncertainty.

2.2.1 The With/Without Principle

CBA always examines a *change*. A proposed change may be a change in policy (a regulatory change) or it may be the implementation of a project (a dam, hydroelectric plant, etc.). In order to concentrate on the benefits and costs of the issue at hand, CBA should measure the projected benefits and costs *with* the change (defined as one or more specific policy 'options'), compared to the benefit and costs *without* the change. For example, in the context of a regulatory CBA, *with* a new, more strict emission standard (the policy option), for example, increased costs to the emitters would result, but benefits may be generated from improved environmental quality. Without the proposed change (the baseline condition), the industry will still have a regulatory standard to meet, and costs will still be incurred to meet the existing standard. CBA should deal with these net differences in costs and benefits between *with* and *without* conditions.

While the *with/without* principle seems simple, its implications for analysis can be quite complex. For example, imagine that the existing regulatory standard is based on ambient air quality. *Without* any changes in regulation, industry might nonetheless incur increased costs for future emission controls because increased economic activity would result in greater overall emissions leading to the challenge of maintaining air quality at levels prescribed by the existing standard. These increases in costs must be taken into account in the *without* (or baseline) case in order to make a fair comparison to the *with* case of regulatory change. Another example involves the use of *phased* regulation. If the current regulatory policy involves a systematic reduction in emissions as part of the existing policy (e.g. reduction of emissions by x% required in year 2010) then this increase in costs must be included in the *without* component of the analysis so that this cost is not confounded with the costs of the new regulatory policy. A comparison of *with/without* is not the same as *before/after*. *With/without* analyses require the definition of a time path or baseline regulatory structure (including baseline expectations for technical change impacts on costs of emissions compliance, ambient air monitoring and other expected impacts).

Baselines must consider a number of changes over time, including changes in: regulations, environmental conditions, demographics and economic conditions. The importance of baselines is elaborated in Morgenstern (2000).

2.2.2 The Damage Function Approach

The damage function approach relies on developing a causal chain that links reductions of pollution emissions to changes in environmental quality (usually ambient air or water quality), which in turn causes changes in human health status or materials (buildings, houses, etc.), or agricultural/forest productivity. Each *pollutant* is linked to one or more *endpoints* (health effects, materials damage, etc.). These bio-physical impacts are then examined for their economic impact by applying economic models to assess the change in *utility* or *welfare* benefits arising from the projected changes in health, material, agriculture/forest productivity or other endpoints. Assuming that source apportionment for various industrial sectors and societal activities is possible, the predicted benefits from emissions changes are then aggregated over time, space, and the number of individuals affected, to arrive at the aggregate benefit. Ideally the economic valuation component should be integrated with the health and environmental effects, and not be a separate model that simply provides *per case* measures of value. Further discussion of the damage function approach is provided in Krupnick, Rowe and Lang (1997).

2.2.3 Accounting Stance: Scope of the CBA

The accounting stance defines the scope of the proposal being considered and sets out the boundaries for the assessment of costs and benefits. These boundaries can be geographical (*is the analysis local, regional, national, or international?*), temporal (*what are the starting and ending dates for the analysis? What time step will be analyzed, e.g. every year, every five years, just the end year?*), and sectoral (*which sectors of the economy are directly or indirectly affected?*).

Consider the geographical boundary question as an example. The accounting stance can be a significant factor in CBA involving site-specific projects because environmental costs often accrue in a local area while social benefits are spread over a larger domain (e.g. hydroelectric dams). In the case of air quality assessment the accounting stance may also play a significant role if, for example, industry is concentrated in certain regions while affected populations are dispersed widely across the country. If the accounting stance focuses only on the region that contains the industrial activity, then the benefits arising from air quality changes outside of the region will not be considered. However, the question remains as to how broad a net should be cast in order to capture all of the relevant benefits and costs. In the context of the CWS discussion, the accounting stance for Canada's air quality is national (i.e. interprovincial), thereby reducing many of the concerns regarding accounting stance effects on the CBA. Other complications arise with a national focus, such as the uneven interprovincial distribution of benefits and costs. A discussion of accounting stance impacts on CBA is provided by Howe (1971).

2.2.4 Measurement of Costs and Benefits

As described above, CBA relies on monetary measures of costs and benefits. These measures of costs and benefits are based on economic theory of 'the firm' and 'the consumer' (see e.g. Dasgupta and Pearce, 1978). In the context of air quality changes, benefits are the changes in the welfare (well-being) of each individual affected by the change. For CBA purposes, benefits are typically measured as an individual's maximum willingness to pay for the environmental quality change, or alternatively the amount of money that the individual would be willing to forgo to enable the change in environmental quality. Note that these conceptual measures of welfare can be measured by observing trade-offs that individuals make in the market place, or by structuring experiments to examine how individuals would make trade-offs in hypothetical situations presented to them by investigators. While the concept of welfare is relatively straightforward, the measurement of this welfare change, based on market data or other information, is often a challenging task.

The theoretically appropriate measures of costs are the impact on social welfare or the opportunity cost (cost of opportunities forgone) of the change in regulation. For example, if a new regulation requires reduced emissions, direct compliance costs of installing new emission reduction equipment are often used as measures of the cost of the policy change. However, if the firm can employ alternative inputs, the costs of meeting the regulation may be lower than the costs of installing new equipment. Furthermore, the output prices that a firm can charge to its customers may increase as a result of the regulation and the resulting reduction of products supplied, again reducing the monetary impact of the regulation on the firm. The firm's behaviour in light of the new regulation and their choice of cost-minimizing strategies under this new regulation, compared to the firm's behaviour without the new regulation, indicates the opportunity cost of the regulation[2].

There are three broad approaches for estimation of costs: (1) Direct costs, that include no behavioural changes in the industry and only consider direct costs of implementing

[2] The theoretically appropriate measure of opportunity cost is the change in *producer surplus*. Producer surplus is defined as the area above the firm's supply or marginal cost curve but below price. As such it is a measure of the gains the producer has for selling at prices higher than marginal cost. Regulatory policies typically shift the marginal cost curve by increasing costs of production. However, since the firm can respond to regulatory change in many ways, not only through direct implementation of emissions reduction technology, the cost of implementing emissions reduction technology is thought of as an upper bound on the cost estimate.

emission reduction technology; (2) Partial equilibrium costs, that include behavioural or market changes within the sector or industry being directly affected; and (3) General equilibrium costs, that examine behavioural or market impacts on the affected industry and all industries linked to it. Direct costs are often referred to as *engineering cost estimates,* since these are typically based on the costs of revising the production system to address the policy or regulatory change. Typically, direct cost estimates are assumed to overstate the true social cost of regulatory change because they ignore behavioural changes that can generate cost savings. Furthermore, direct costs ignore the market impacts of regulatory changes that at times may result in higher product prices and thereby affect the returns to the firm from the product market. While partial and general equilibrium approaches are more theoretically appealing, they also require considerably more data and analysis and thus raise a variety of complicating issues in the analysis. A second dimension of cost analysis is the degree to which costs are assessed in a static context or in a dynamic / intertemporal setting. If costs are examined in a static framework, dynamic factors such as capital investment are ignored. Furthermore, the role of technological change, research and innovation is ignored in static analyses. Considerably more detail on the issues associated with measuring costs are provided in Chapter 7, which deals specifically with the cost analysis for the Canada-Wide Standards development process.

2.2.5 Techniques for Measuring Benefits

While economic theory describes benefit measures as *willingness to pay* or *willingness to accept*[3], the actual measurement of this amount introduces various challenges. Freeman (1993) summarizes the methods used to value the improvements in environmental amenities, including those relevant to air quality issues. Table 1 summarizes the information from Freeman (1993, Table 14-1, p. 487). Note that some of the techniques

[3] While the benefit measures are referred to as “willingness to pay” or “willingness to accept”, most benefit measurement techniques do not actually directly ask individuals to reveal this amount. Individuals may reveal their willingness to pay or willingness to accept through market transactions or other forms of behaviour. There is considerable confusion in the popular literature about the concept of “willingness to pay” (a theoretically appropriate welfare measure) and the method of benefit estimation, contingent valuation, that actually asks individuals what they would be willing to pay.

employed are consistent with economic theory and actually attempt to measure welfare, while other techniques (identified in Table 1) are not consistent with economic theory and are approximations (usually lower bound approximations) to economic welfare or benefit measures.

Table 1 and Table 2 refer to a variety of approaches for valuation. *Direct valuation* techniques determine the monetary value of the good or service (or a change in the quality of the good or service) directly from the observations collected. For example, in a perfectly competitive market, the price of the good reveals the individual's or firm's marginal willingness to expend some of their limited financial resources to pay for that good in preference to other goods. *Contingent valuation* is a technique that also attempts to directly elicit willingness to pay although in this case the elicitation involves the administration of a highly structured hypothetical question (or set of questions) that identifies how much an individual would be willing to pay for a good or service. *Indirect valuation* approaches develop estimates of the monetary value of the good or service by observing related buy–and–sell markets, or by inferring results from observations that an individual would pay at least a certain amount. *Referenda* for example, when involving actual monetary versus service tradeoffs, can be used to indicate individual monetary values. If an individual votes for an option that requires expenditures of $X, that individual could be said to be willing to pay at least $X for that option. *Hedonic property* and wage models are methods that *decompose* the price of market goods (property and labor) into components that include environmental amenities, health risks, and other elements. For example, these methods might examine how much of a wage premium an individual would have to be paid to work in a higher risk occupation. These methods are indirect approaches since the monetary value of the change in environmental quality or health risk is not elicited directly by questioning individuals, rather it is indirectly determined from the market for property or labor. *Travel cost models* and *random utility models* are models commonly used in evaluating the value of recreation activities and the impact of changes in environmental quality on recreation value. Again, these methods are indirect as they examine the value of a typically un-priced (or administratively priced) good – outdoor recreation – through market purchases of other goods required for travel to the site for the activity.

Table 1 Environmental Quality Changes and Valuation Techniques

ENVIRONMENTAL SERVICE FLOW	TECHNIQUE OR MODEL EMPLOYED
Impacts on Human Health	
Mortality risk	Revealed Preference Hedonic Wage Averting Behaviour Stated Preference Contingent Valuation Contingent Behaviour Stated Choice / Conjoint *Other Methods* *Human Capital (foregone earnings)* *Quality Adjusted Life Year / Cost of Illness*
Morbidity	Revealed Preference Averting Behaviour Stated Preference Contingent Valuation Contingent Behaviour Stated Choice / Conjoint *Other Methods* *Cost of Illness (lost earnings, medical costs, etc.)*
Impacts on Visibility / Amenity	
Property Values	Revealed Preference Hedonic Property Values Averting Behaviour Stated Preference Contingent Valuation Contingent Behaviour Stated Choice / Conjoint
Impacts on Ecological Function / Services	
Recreation	Revealed Preference Travel Cost Models / Random Utility Models Stated Preference Contingent Valuation Contingent Behaviour Stated Choice / Conjoint *Other Methods* *Unit day values*
Agricultural and Forestry Impacts	Revealed Preference Changes in Producer and Consumer Surplus Averting Behaviour Hedonic Property Values Stated Preference Contingent Valuation Contingent Behaviour Stated Choice / Conjoint
Damages to Materials (soiling, deterioration, etc.)	Revealed Preference Changes in Producer and Consumer Surplus Averting Behaviour Hedonic Property Values Stated Preference Contingent Valuation Contingent Behaviou Stated Choice / Conjoint *Other Methods* *Replacement costs*
Passive Use Values	Stated Preference Contingent Valuation Contingent Behaviour Stated Choice / Conjoint

Based on Freeman (1993, Table 14-1, p.487). Methods in *italics* are generally not consistent with economic theory.

Table 2 Valuation Techniques

	Revealed Preference (Observed Behaviour)	Stated Preference (Hypothetical)
Direct Valuation	Competitive Market	Contingent Valuation (open ended) Contingent Valuation (bidding games)
Indirect Valuation	Travel Cost / Random Utility Models Hedonic Property Values Hedonic Wage Models Referenda	Referendum Contingent Valuation Discrete Choice Contingent Valuation Contingent Behaviour Stated Choice / Conjoint

Based on Freeman (1993) and Mitchell and Carson (1989).

Hypothetical indirect methods include contingent behaviour methods that ask individuals structured questions to identify how their behaviour might change if prices or quality were to change. For example, individuals may be asked how many recreation trips they would make if environmental quality (perhaps fishing catch rates) was enhanced at all recreation sites within their region. Referendum contingent valuation is a hypothetical referendum that asks respondents to vote on alternatives in a hypothetical referendum where the alternatives include trade-offs between environmental quality or health quality and money. For example, individuals may be asked how they would vote on a program that would cost $X to improve long range visibility in their neighborhood. If they voted yes to the program they would be indicating that they were willing to pay at least $X for the program. Discrete choice contingent valuation similarly asks individuals about their choice of alternatives involving trade-offs, however, it does not necessarily involve a referendum setting. Choice experiments or conjoint analysis also ask individuals to make choices from hypothetical options (referendum options, behavioural choices like a choice of fishing sites, etc.) but these choices are characterized by attributes that determine the major reasons for the choices made. This allows the valuation of the attributes of the choice alternative, as well as the alternative itself.

Other methods that are not consistent with economic theory are often employed as approximations to true welfare measures. In the mortality category, for example, human capital or lost productivity/wages methods have been employed as estimates of the foregone economic output associated with premature death. However, foregone earnings are not consistent with economic theory as a welfare measure, and this measure significantly understates the benefits of improved environmental quality (Braden and Kolstad, 1991; Freeman, 1993). Similarly, Cost of Illness (COI) is often used as a measure of the economic cost associated with morbidity, but these measures are at best lower bound estimates of the true economic cost of ill health (Braden and Kolstad, 1991, Freeman, 1993).

Each one of the valuation techniques discussed above involves considerable technical skill and attention to detail to be credible. Further details on the methods can be found in the following sources: General Overviews (Freeman, 1993; Braden and Kolstad, 1991), Travel Cost and Random Utility Models (Freeman, 1993; Braden and Kolstad, 1991), Stated Choice / Conjoint (Adamowicz et al., 1999; Adamowicz, 2000), Contingent Valuation (Mitchell and Carson, 1989; Carson, 2000) and Hedonic Price methods (Braden and Kolstad, 1991).

2.2.6 Comparison of Benefits and Costs Over Time. The Discount Rate

The economic approach to estimating the dollar values of costs to be expended in future and benefits to be accrued over time is termed *discounting*. When estimates of future benefits and costs are discounted, their anticipated monetary values at future points in time are converted into present-day dollar amounts, by adjusting downwards the value of projected benefits and costs by a few percentage points per year (the discount rate). This is done to reflect the opportunity costs of non-productive capital expenditures and the social rates of time preference – meaning that a deferred expenditure is better than an immediate expenditure of the same dollar amount, and a near-term benefit is preferable to a long-term payoff (Kopp, Krupnick and Toman, 1997). With all regulatory impacts converted to consumption equivalents, analysts can discount streams of benefits and costs at a discount rate that reflects consumption tradeoffs across a defined span of time. Uncertainty regarding an “exact” rate of discount (using either shadow price of capital or social time preference approaches) illustrates the need for sensitivity analysis – one must introduce a series of plausible discount rates into the CBA calculations to gauge how benefits and costs would change with alternative discount-rate hypotheses. This approach is also necessary for careful assessment of the extent to which the burdens of regulation fall on consumption or investment.[4]

2.2.7 Summary Measures of Benefits and Costs

The appropriate outcome of a CBA is usually expressed as the Net Present Value (discounted benefits minus discounted costs). Alternative summary measures may be expressed as an estimated benefit-cost ratio (the discounted present value of benefits divided by the discounted present value of costs) or an Internal Rate of Return (the size that the discount rate needs to be for projected net benefits to equal zero.) As we want to identify actions that have the largest net benefit for society and a high benefit-cost ratio is not necessarily consistent with large net benefits, Net Present Value is the preferred summary measure.

A well constructed CBA will include information about the relationship between variation in these summary welfare measures as model outputs and the underlying variation in the benefit and cost measures that constitute the model inputs. The impact on the summary outcome measures of uncertainties in the critical input measures of costs and benefits (including discount rates) must also be determined and reported (Howe, 1971; Dasgupta & Pearce, 1978).

2.2.8 The Treatment of Uncertainty

In principle, the appropriate approach to handling uncertainties involves comparing estimates of the total present discounted benefit distribution and the discounted cost distribution to yield a net benefit distribution associated with the given scenario. This distribution, evaluated according to some decision rules and compared with net benefit distributions from other scenarios, permits an efficient scenario to be identified, at least within the confines of the analysis.

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[4] A special but important issue arises here when regulatory impacts have an intergenerational time scale (e.g., costs borne today but benefits received by the next generation). Even a consumption-based discount rate could reduce future impacts to trivial levels over a long time frame. In terms of discount rate policy, it could be argued that intergenerational effects deserve "special" treatment reflecting societal tradeoffs across generational income distributions. One simple example is the argument that a discount rate reflecting the long-term rate of economic growth can reflect an "equal" treatment of the generations, reflecting the presumed greater economic ability of future generations to shoulder burdens relative to ourselves.

The standard approach in CBA for comparing distributions of net benefits is defined under expected-utility theory, in which each potential state of the world generates a particular expected net benefit, and the utility of these net benefits is weighted by their likelihood of occurrence and then summed. The structure of this utility function in part reflects the attitude of members of the society toward risk.

In recent years, this approach to describing valuation of uncertain outcomes has been criticized. Critics argue that individuals ignore or systematically estimate risks inaccurately, especially low-probability, high-consequence events; that individuals' valuation of risky situations is influenced by their frames of reference; and that perceptions of risky outcomes are affected by concerns about future regret as well as expected utility. Camerer and Kunreuther (1989) provide an extensive review of these and other issues. However, these criticisms are by no means universally accepted, and alternatives to expected utility theory also have not won widespread acceptance. For the time being, CBA will continue to be based on calculation of net benefits, with adjustments for the cost of risk-bearing, while research continues.

The analysis of uncertainty can be conducted within this framework using Monte Carlo simulation techniques. These techniques involve characterizing *statistical* uncertainties in the input data, equation parameters, and other features of the analysis with estimated probability distribution functions (PDFs). Monte Carlo simulation uses a random sample of each of these PDF distributions in multiple repetitions of the designated calculations (in what are called realizations) to form probability distributions of the output variables of interest (e.g. net benefit). These distributions reflect the statistical uncertainties within and between the appropriate stages of the analysis. This simulation approach does not address the inherent *model* uncertainty, i.e. whether the equations used in the model simulations accurately reflect the reality of the system that is being simulated.

In practice, the full representation of uncertainties is often ignored in favour of more *ad hoc* approaches, such as the representation of some output variables by their expected values and of others by *low*, *middle*, and *high* values (say, by the values representing the 95% confidence interval around some expected value). These are then paired with their corresponding values from the next stage of the analysis. The result is a set of *low*, *middle*, and *high* values for the final output distribution (say, the benefits of a waste cleanup) that do not correspond to any particular confidence interval. However, this simplified approach violates several important rules of statistical computation, and it can often produce misleading information about the nature and degree of uncertainty in the CBA results.

2.3 Cost Effectiveness Analysis and Economic Impact Analysis

Cost-benefit analysis (CBA) and cost-effectiveness analysis (CEA), are economic techniques that produce information intended to improve the quality of public policy decisions (Kopp, Krupnick and Toman, 1997). Conceptually, then, CBA could be used to rank policy options on the basis of their improvements or reductions in well-being. For example, on the basis of such improvements, one could rank three air-quality policies that are related to urban ozone and that offer various ambient ozone standards to be attained, various reductions in illnesses related to ozone exposure, and various costs of attaining those standards.

CEA is a particular form of CBA. In the example of air quality above, a decision-maker would use CEA to choose among various options to attain a chosen standard. CEA does not imply choosing the policy with the smallest dollar price tag (although many people believe that it does). Strictly speaking, CEA chooses the policy that achieves the specified goal with the smallest loss in social well-being. The smallest welfare loss might not be associated with the smallest dollar cost. CBA and CEA are often described as economic impact analysis techniques but distinctions are necessary. Economic impact analysis, strictly speaking, is yet another form of analysis that focuses on the impacts on employment, wage rates, price changes and other changes in the economic system that arise when policies change. Economic impact analysis often refers to assessment via input-output models or the use of output, income and employment multipliers. Economic impact analysis examines how the economic system will change, however, only some of these changes are relevant for CBA. For example, CBA does not typically include creation of employment as a benefit. This is because CBA tends to focus on the primary benefits and costs (the direct costs and benefits arising from the change) and not the indirect costs and benefits. Counting both would result in double-counting. Thus, while creation of employment may occur, in the case of a fully employed economy the employment creation in one sector will be offset by other potential indirect costs - like increases in wage rates for all sectors, increases in services required and a variety of other issues (see Section 9.3 for further discussion of CEA).

2.4 Limitations of Cost-Benefit Analysis

CBA has several key theoretical or conceptual assumptions and limitations, including the reliance on individual measures of monetary value as the cornerstone of CBA and the use of the compensation principle as the indication of socially beneficial projects.

A number of concerns about CBA are commonly raised. They include the following:

(i) The environment is a public good that is not exchanged in markets and therefore defies economic valuation. Thus, the use of CBA to evaluate environmental policies is inappropriate.

(ii) Environmental protection is often desirable for reasons that cannot be quantified--social, spiritual, and psychological values that defy monetization.

(iii) CBA does not take the “rights” of future generations into account.

(iv) Economic benefit measures are hypothetical measures of benefits and are not actual benefits that can be measured in terms of savings in health case costs or other “real” benefits.

These issues and the economist response to them are elaborated in Appendix A.

Furthermore, there have been many criticisms of the concentration on efficiency rather than equity. CBA is typically employed in analysis that considers benefits and costs over time and measures are taken to make benefits and costs in future time periods commensurate with benefits and costs today (discounting). As discussed in Section 2.2.6 the choice of discount rate can significantly affect the outcome of the CBA and has been the focus of much debate. Many of these conceptual issues are debated in Kopp, Krupnick and Toman (1997).

In addition to theoretical issues there are many measurement challenges in CBA. While the monetary values of market goods are relatively straightforward to measure, the monetary values of various non-market goods are more difficult to assess. Some would argue that there are many goods and services for which no measurable monetary value exists. However, in the past few decades there has been a substantial increase in the literature on the valuation of non-market goods and services and an explosion of empirical estimates of non-market values (see Freeman, 1993). While this area of the literature has increased dramatically, there is still debate about how non-market values arising from health effects, for example, are measured. This issue is addressed in Chapter 8.

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3 Panel's Interpretation of Cost-Benefit Analysis for PM and Ozone Canada-Wide Standards

3.1 Introduction

Canada-Wide Standards (CWS) for PM and ozone were ratified by the Canadian Council of Ministers of the Environment (CCME) in June 2000[5]. Socio-economic considerations are one of eight principles underlying the development and attainment of CWS, according to a CWS sub agreement signed by the Canadian Council of Ministers of the Environment (CCME). Principle 3.1.7 states that

> *"measures to attain agreed-upon Canada-Wide Environmental Standards will be determined in a sustainable development context, recognizing environmental and socio-economic considerations"* .

A CCME document *Framework for Socio-economic Analyses in Setting Environmental Standards* (CCME, 1998), describes procedures and information requirements for socioeconomic assessments of potential or proposed environmental standards. The Framework involves five steps each with key tasks and information requirements to define: 1) the environmental problem and adverse effects; 2) the sources and trends of problem activities and/or pollutant releases; 3) potential technical methods to reduce releases and achieve standards and their costs; 4) beneficial economic and environmental consequences of potential standards; and 5) evaluation techniques and decision criteria for selecting a standard. This Framework states that while it may not be possible or necessary to carry out all of the analytical steps because of time, data or resource constraints, a partial assessment can produce information that is useful for policy deliberations. The Frameworks notes that *quantitative uncertainties in quantitative estimates of benefits and costs must be analyzed and implications communicated to all those involved in the development of standards*.

[5] A CWS for $PM_{2.5}$ of 30 ug/m^3, 24 hour averaging time, by year 2010. A CWS for ozone of 65 ppb, 8 hour averaging time, by year 2010.

Although the Framework emphasizes socio-economic assessment methods and monetary values as indicators of economic and social importance, the views and perspectives of individuals, groups and organizations are recognized as important considerations in the decision-making process. The Framework also notes that socio-economic findings are not intended to be prescriptive concerning decisions about environmental standards because other input factors such as toxicity, epidemiological findings, ecological consequences and geographical distribution of effects are also necessary and important to an informed choice with respect to standard setting.

Socio-economic considerations are also specified under Government of Canada Regulatory Policy (Nov. 1999) requiring federal regulatory authorities to demonstrate that the benefits of regulatory requirements are greater than their costs. When regulations address health, social, economic or environmental risks, it must also be demonstrated that regulatory effort is being expended where it will do the most good.

This chapter provides an overview of the methodology and results of the analyses of costs and benefits that were undertaken as part of the CWS development process. The following reports prepared for the CWS Development Committee for PM and Ozone provide further details of the CWS methodology and results:

i) *Compendium of Benefits Information* – 99-08-17

ii) *Compendium of Cost Information* – Aug. 6, 1999 and

iii) *Emission Control Cost Study for Sources of NOx, VOCs, PM_{10}, $PM_{2.5}$ and SO_2 Emissions: Methodology Report,* prepared by Stratus Consulting Inc. Dec. 3, 1999.

Preliminary results of these analyses were presented at the final National Stakeholder Consultation Workshop held on May 26-28, 1999 in Calgary, Alberta. A *Discussion Paper on Particulate Matter (PM) and Ozone Canada Wide Standard Scenarios for Consultation,* prepared by the CWS Development Committee (CWS DC, 1999) for distribution at the workshop provided an overview of the approach and results of the analyses.

3.2 General Approach

The Discussion Paper on Particulate Matter (PM) and Ozone Canada-Wide Standard Scenarios for Consultation prepared for the final National Stakeholder Consultation Workshop indicates that *"in selecting PM and ozone CWS level scenarios for stakeholder consideration, an attempt was made to balance the anticipated benefits of improved air quality with the technological feasibility and costs of achieving those improvements (CWS DC, 1999 p. 15)"*. The following step-wise approach to the cost-benefit analysis was described:

Step 1 – Identify the optional ambient targets (range of candidate CWS levels)
Step 2 – Estimate the required ambient reductions to reach the targets
Step 3 – Estimate the corresponding avoided impacts (benefits)
Step 4 – Estimate the required reductions in precursor pollutant emissions
Step 5 – Assess the technological feasibility and estimate the associated emission reduction costs
Step 6 – Compare the avoided costs (benefits) and the anticipated costs of improved air quality

The methodology is described by the CWS Development Committee as "*a preliminary and, in some cases, cursory analysis used to provide a macro level order of magnitude perspective on the costs associated with the various optional levels for PM and ozone CWSs and combinations of optional levels"*. The discussion paper notes that considerable additional work will be required to improve the information base in order to enable refinement of cost and benefit estimates and to allow the detailed design of implementation plans and specific sectoral strategies in various regions across Canada.

The methodologies and results of each of the above steps are described below. The Panel's assessment of the strengths, limitations and uncertainties of the approaches taken in each step is presented in Chapters 4 to 8.

3.2.1 Step 1: Identify the optional ambient targets (range of candidate CWS levels)

The range of ambient air quality scenarios for socio-economic analysis was determined by the CWS Development Committee based on scenarios agreed to at the October 1998 National Multi-Stakeholder Consultation Workshop. The air quality scenarios considered for the benefits assessment were as follows:

PM_{10}	40, 60 and 80 µg/m^3 (24 hour)
$PM_{2.5}$	20, 30 and 40 µg/m^3 (24 hour)
Ozone	50 to 70 ppb (8 hour)

3.2.2 Step 2: Estimate the ambient level reductions required to reach targets

The AIR QUALITY VALUATION MODEL (AQVM) was used to estimate the health and environmental benefits associated with reduced ambient levels of PM and ozone. The AQVM requires the user to define an ambient air pollution change scenario by specifying either an absolute or a percentage change from baseline ambient concentrations.

To estimate the annual change in ambient concentrations as input into AQVM, the following air quality information was required: 1) determination of baseline ambient air concentrations 2) estimation of natural background ambient air concentration level above which benefits would occur 3) a method for adjusting baseline ambient data to simulate attainment of the CWS scenarios:

i) Baseline Ambient Air Quality

$\underline{PM_{10}}$

Baseline one year distributions of PM_{10} air quality data for 37 Canadian cities (census metropolitan areas (CMAs) and census agglomerations (CAs)) were determined using three years (1994-1996) of manual data (collected primarily on a one-in six day schedule) for 25 Canadian cities and towns and one or more years of continuous (Tapered Element

Oscillating Microbalance or TEOM) monitoring data from 1994-1997 for 12 cities and towns.

<u>$PM_{2.5}$</u>

Baseline one year distributions of $PM_{2.5}$ air quality data for 14 Canadian cities (CMA/CA'S) were determined using three years (1994-96) of manual data (collected primarily on a one-in-six day schedule). Continuous (TEOM) $PM_{2.5}$ monitoring data was only available for 2 stations.

<u>Ozone</u>

Baseline ozone data for 36 cities and towns for three years (1994-1996) was determined using hourly data from May to September at 119 monitoring stations across Canada.

ii) Background levels

The annual average natural background concentration was subtracted from the predicted concentration distribution based on application of a rollback algorithm. This was to ensure that the estimated avoided impacts did not include attribution of benefits to reductions below natural non-anthropogenic background levels.

An average value of 5 $\mu g/m^3$ was selected as the estimated daily background concentration for PM_{10} and 2.5 $\mu g/m^3$ was selected as the background concentration for daily $PM_{2.5}$ data. An average value of 40 ppb was selected as the estimated hourly background concentration for ozone.

iii) Ambient Concentration Changes

<u>PM</u>

To calculate community-wide air quality concentration changes consistent with the expected form of the CWS, the following guidelines were used:

1. All CMA/CAs for which community-oriented monitoring data were available were included in the analysis. Rural (or background) sites were not included.
2. For CMA/CAs that have both manual (mostly 1 in 6 day sampling) and continuous monitors, the continuous (or daily) monitor results were used.

3. For CMA/CAs that have multiple years of data, the annual concentration changes for each year were averaged over the number of years for which data were available
4. For CMA/CAs that have multiple monitor stations, the annual concentration changes averaged for each station were averaged with the results from other stations, even if they were for a different number of years.
5. For CMA/CAs with no continuous monitors but with more than one type of manual monitor or multiple manual monitors, all annualized concentration changes from each of the monitors were averaged.

A 'proportional linear rollback' approach was used to simulate attainment of the range of optional CWS levels by adjusting the current (baseline) air quality data for concentrations exceeding an estimated background level. For each CWS scenario, the ratio between the target level and the 3rd highest maximum concentration at a given site was used to scale back the baseline data for that site. The sum of the rolled back data, averaged over 365 days, was subtracted from the baseline annual mean (less background) to determine the annual concentration change for each of the scenarios.

Annual Change = Baseline Conc. – Rollback Conc.

Baseline Conc. =	(daily concentration – background concentration (BG))÷365
Rollback Conc. =	(daily concentration X Rollback Reduction) – BG) ÷ 365
Rollback Reduction =	{1-(3rd Max-CWS)/3rd Max)} = CWS/3rd Max

The 3rd highest maximum reading was used to provide a more robust representation (i.e. less likelihood of outliers giving a false or misleading result) of the level of reduction required to reduce the peak concentrations and the associated distribution of concentrations.

Ozone

The following guidelines were used to calculate community-wide air quality concentration changes consistent with the expected form of the CWS:

1. All CMA/CAs for which monitoring data were available were included in this analysis. Sites outside CMA/CA's were not included.
2. The most recently available three years (1994-1996) of data were used to calculate the seasonal concentration changes. For stations that did not have three years of data, the average of the available data was used.

3. For CMA/CAs that had multiple monitor stations, the station with the highest average ozone concentrations was used.

A curvilinear "rollback approach" was used to model the decline in concentration frequency for rollback of hourly average ozone concentrations. The algorithm used was based on results from a trends analysis on the distribution of Canadian hourly ozone data. For sites that experienced a downward trend, the greatest decline in the frequency of hourly average concentrations was experienced within the high-level (>90 ppb) concentration ranges and the decline approached zero in the low concentration ranges (30-40 ppb). The decline in frequency was used as a surrogate for the decline in concentration and a linear percent-change algorithm was developed to rollback the hourly values for each scenario.

Percentage reduction required in Maximum Ozone:

$$Rmax = ((O_{3Max}-R_n)/O_{3Max}) * 100\% \qquad (1)$$

where

R_n=Rollback Target 50, 60 ppb etc.
O_{3Max}= Maximum Ozone

Reductions scaled from Maximum to Threshold:

$$R_s= (1-((O_{3Max}-0_{3h})/ (O_{3Max} – T_h)))*R_{max} \qquad (2)$$

$$0_{3hR}=O_{3h}*(1-R_s) \qquad (3)$$

Note: if $O_{3h}<T_h$ then $R_s=0$; if $0_{3h}>0_{3Max}$ then $R_S=R_{MAX}$

Where:

T_h=Threshold (no reduction below this level, set to 40 ppb)
O_{3h}= Measured hourly ozone value (ppb)
O_{3hR} = Adjusted hourly ozone value (ppb)

This algorithm was applied directly in the case of sites with maximum ozone concentrations of 100 ppb or less. For sites with higher maxima the algorithm was applied twice, with the first application used to roll maxima to 100 ppb and the second to further reduce the maxima to the scenario value.

3.2.3 Step 3: Estimate the corresponding avoided impacts (benefits)

3.2.3.1 AQVM: The Benefits Assessment Model

The AIR QUALITY VALUATION MODEL (AQVM) was used to generate estimates of the absolute numbers of avoided health events for alternative reductions in ambient concentrations of PM_{10}, $PM_{2.5}$ and ozone, and to provide estimates of the monetary value of these avoided health impacts. The AQVM requires the user to define an ambient air pollution change scenario by specifying either an absolute or a percentage change in baseline ambient concentrations. The AQVM contains a baseline air quality database for Canada derived from available ambient monitoring data and a population database from the 1996 Census for all the census divisions (CDs) and for the CMAs. The AQVM contains default concentrations-response functions for health outcomes derived from key epidemiological studies as well as the monetary value estimates for various types of human health and environmental endpoints. The human health and environmental impacts included in the benefits analyses for the CWS process are identified in Table 3.

Table 3: Human Health and Environmental Effects Included in the Estimation of Ambient PM and Ozone Reduction Benefits for CWS

AQVM Benefit Category	Included	Excluded
Mortality	X	
Morbidity		
Chronic bronchitis cases	X	
Respiratory hospital admissions	X	
Cardiac hospital admissions	X	
Emergency room visits	X	
Asthma symptom days	X	
Restricted activity days	X	
Acute respiratory symptom days	X	
Child bronchitis	X	
Production/consumption Crops		X
Forests		X
Fisheries		X
Economic Assets Materials (corrosion, soiling)	X	
Property values		X
Environmental Assets Use		X
Recreation		X
Visibility Aesthetics		X
Passive Use (nonuse) and/or Total Values for other impacts to vegetation, wildlife, and other ecologic resources		X

3.2.3.2 Concentration Response Relationships for Human Health Effects

Concentration-response functions allow the estimation of the change in the frequency predicted occurrence of each health effect that would be expected as a result of changes in ambient pollution. Concentration-response functions used in the AQVM were drawn from epidemiological literature according to three criteria:

i) studies that recognized and attempted to minimize the effects of confounding variables such as seasonality and weather are preferred,

ii) studies were selected that examined exposure to levels of air pollution relevant to the Canadian context, particularly those from North America and Western Europe,

iii) studies that addressed clinical outcomes or changes in behaviour that would best lend themselves to economic valuation were included.

Specific concentration-response functions were selected from the studies according to a “weight of the evidence” approach. Central estimates generally reflect the mean or midpoint results from selected studies. Low and high estimates reflect the reasonable range of credible results, not the absolute range of highest and lowest values. The concentration response functions utilized in the AQVM for $PM_{2.5}$, PM_{10} and ozone are summarized in Table 4, Table 5, and Table 6.

Table 4: Concentration – response relationships utilized in AQVM for $PM_{2.5}$

Health Event Category	Concentration-Response Parameter (Probability Weighting Applied)
Annual mortality risk per 1 μg/m^3 change in annual average $PM_{2.5}$ concentration *Sources: Pope et al. (1995); Schwartz et al. (1996)*	Low 0.87 x 10 $^{-5}$ (22%) Central 2.14 x 10 $^{-5}$ (67%) High 4.82 x 10^{-5} (11%)
Chronic bronchitis (CB) annual risk per 1 μg/m^3 change in annual average $PM_{2.5}$ concentration *Source: Abbey et al. (1995)*	For population 25 years and older: Low: 4.13 x 10^{-5} (25%) Central 8.27 x 10^{-5}(50%) High 12.4 x 10 $^{-5}$ (25%)
Respiratory hospital admissions (RHA) daily risk factors per 1 μg/m^3 change in daily average $PM_{2.5}$ concentration. *Source Burnett et al. (1995)*	Low 1.00 x 10^{-8} (25%) Central 1.21 x 10^{-8} (50%) High 1.42 x 10^{-8} (25%)
Cardiac hospital admissions (CHA) daily risk per 1 μg/m^3 change in daily average $PM_{2.5}$ concentration *Source: Burnett et al. (1995)*	Low 0.79 x 10^{-8} (25%) Central 1.02 x 10^{-8} (50%) High 1.26 x 10^{-8} (25%)
Net emergency room visits (ERV) daily risk factors per 1 ug/m^3 change in daily average $PM_{2.5}$ concentration. *Source: Stieb et al. (1995)*	Low 4.62 x 10^{-8} (25%) Central 5.61 x 10^{-8} (50%) High 6.61 x 10^{-8} (25%)
Asthma symptom day (ASD) daily risk factors given a 1 μg/m^3 change in daily average $PM_{2.5}$ concentration. *Sources: Whittemore and Korn (1980); Ostro et al. (1991)*	For population with asthma (6% of population) Low 1.62 x 10^{-4} (33%) Central 2.64 x 10^{-4} (34%) High 3.65 x 10^{-4} (33%)
Restricted activity day (RAD) daily risk factors given a 1 μg/m^3 change in daily average $PM_{2.5}$ concentration. *Sources: Ostro (1987); Ostro and Rothschild (1989)*	For nonasthmatic population (94% of population) 20 years and older Low 1.31 x 10^{-4} (25%) Central 2.50 x 10^{-4} (50%) High 3.95 x 10^{-4} (25%)
Net day with acute respiratory symptom (ARS) daily risk factors given a 1 μg/m^3 change in daily average $PM_{2.5}$ concentration. *Source: Krupnick et al. (1990)*	For nonasthmatic population (94% of population) Low 1.25 x 10^{-4} (25%) Central 2.79 x 10^{-4} (50%) High 4.14 x 10^{-4} (25%)
Child acute bronchitis (B) annual risk factors given a 1 μg/m^3 change in annual average $PM_{2.5}$ concentration: *Source: Dockery et al. (1996)*	For population under age 20: Low 0.62 x 10^{-3} (25%) Central 1.65 x 10^{-3} (50%) High 2.69 x 10^{-3} (25%)

Source: Human Health and Environmental Benefits of Achieving Alternate CWS for Inhalable Particulates ($PM_{2.5}$, PM_{10}) and Ground Level Ozone. Final Report. Prepared by Paul De Civita, Environment Canada, David Stieb, Health Canada, Lauraine Chestnut, David Mills, Robert Rowe, Stratus Consulting. July 25, 1999. In Compendium of Benefits 99-08-17.

Table 5: Concentration-response relationships utilized in AQVM for PM_{10}

Health Event Category	Concentration-Response Parameter (Probability Weighting Applied))
Annual mortality risk factors given a 1 μg/m^3 change in annual average PM_{10} concentration *Sources: Schwartz et al. (1996), Pope et al. (1995)*	Low 4.4 x 10^{-6} (22%) Central 12.1 x 10^{-6} (67%) High 28.2 x 10^{-6} (11%)
Chronic bronchitis (CB) annual risk factors given a change in 1 μg/m^3 annual average PM_{10}concentration *Source: Abbey et al. (1993).*	For population 25 years and over: Low 3.0 x 10^{-5} (25%) Central 6.1 x 10^{-5} (50%) High 9.3 x 10^{-5} (25%)
Respiratory hospital admissions (RHAs) daily risk factors given a 1 μg/m^3 change in daily PM_{10}concentrations *Sources: Burnett et al. (1995), Pope (1991)*	Low 0.64 x 10^{-8} (33%) Central 0.78 x 10^{-8} (50%) High 3.26 x 10^{-8} (17%)
Cardiac hospital admissions (CHAs) daily risk factors given a 1 μg/m^3 change in daily PM_{10}concentration *Source: Burnett et al. (1995)*	Low 5.0 x 10^{-9} (25%) Central 6.6 x 10^{-9} (50%) High 8.2 x 10^{-9} (25%)
Net emergency room visits (ERVs) daily risk factors given a 1 μg/m^3 change in daily PM_{10}concentration *Source: Stieb et al. (1995)*	Low 2.96 x 10^{-8} (25%) Central 3.66 x 10^{-8} (50%) High 14.3 x 10^{-8} (25%)
Asthma symptom days (ASDs) daily risk factors given a 1 μg/m^3 change in daily PM_{10}concentration *Sources: Whittemore and Korn (1980), Ostro et al. (1991)*	For population with asthma (6% of population) Low 1.62 x 10^{-4} (33%) Central 1.72 x 10^{-4} (34%) High 1.82 x 10^{-4} (33%)
Restricted activity days (RADs) daily risk factors given a 1 μg/m^3 change in daily PM_{10}concentration *Sources: Ostro (1987), Ostro and Rothschild (1989)*	For nonasthmatic population (94% of population) 20 years and older: Low 0.8 x 10^{-4} (33.3%) Central 1.6 x 10^{-4} (33.4%) High 2.5 x 10^{-4} (33.3%)
Net days with acute respiratory symptoms (ARSs) daily risk factors given a 1 μg/m^3 change in daily PM_{10} concentration *Sources: Krupnick et al. (1990)*	For nonasthmatic population (94% of population) Low 1.62 x 10^{-4} (25%) Central 3.44 x 10^{-4} (50%) High 5.18 x 10^{-4} (25%)
Children with acute bronchitis (B) annual risk factors given a 1 μg/m^3 change in annual average PM_{10} concentration *Source: Dockery et al. (1996)*	For population under age 20: Low: 0.57 x 10^{-3} (25%) Central: 1.42 x 10^{-3} (50%) High 2.27 x 10^{-3} (25%)

Source: Human Health and Environmental Benefits of Achieving Alternate CWS for Inhalable Particulates ($PM_{2.5}$, PM_{10}) and Ground Level Ozone. Final Report. Prepared by Paul De Civita, Environment Canada, David Stieb, Health Canada, Lauraine Chestnut, David Mills, Robert Rowe, Stratus Consulting. July 25, 1999. In Compendium of Benefits 99-08-17.

Table 6: Concentration-response relationships utilized in AQVM for Ozone

Health Event Category	Concentration-Response Parameter (Probability Weighting Applied)
Daily mortality risk factors given a 1 ppb change in daily high-hour ozone concentration *Source: Science Assessment Document*	Low 0.37 x 10^{-9} (33%) Central 16.3 x 10^{-9} (34%) High 27.4 x 10^{-9} (33%)
Respiratory hospital admissions (RHAs) daily risk factors given a 1 ppb change in daily high-hour ozone concentration *Source: Burnett et al. (1997)*	Low 0.6 x 10^{-8} (25%) Central 1.1 x 10^{-8} (50%) High 1.6 x 10^{-8} (25%)
Net emergency room visits (ERVs) daily risk factor given a 1 ppb change in daily high-hour ozone concentration *Sources: Stieb et al. (1995): Burnett et al. (1997)*	Low 2.6 x 10^{-8} (25%) Central 4.7 x 10^{-8} (50%) High 6.9 x 10^{-8} (25%)
Asthma symptom days (ASDs) daily risk factor given a 1 ppb change in daily high-hour ozone concentration *Sources: Whittemore and Korn (1980), Stock et al. (1988).*	For population with asthma (6% of population): Low 1.06 x 10^{-4} (33%) Central 1.88 x 10^{-4} (50%) High 5.20 x 10^{-4} (17%)
Minor restricted activity days (MRADs) daily risk factors given a 1 ppb change in daily high-hour ozone concentration *Source: Ostro and Rothschild (1989)*	For nonasthmatic population (94% of population) Low 1.93 x 10^{-5} (25%) Central: 4.67 x 10^{-5} (50%) High 7.40 x 10^{-5} (25%)
Net days with acute respiratory symptoms (ARSs) daily risk factors given a 1 ppb change in daily high-hour ozone concentration *Source: Krupnick et al. (1990)*	For nonasthmatic population (94% of population): Low 5.07 x 10^{-5} (25%) Central 9.03 x 10^{-5} (50%) High 13.0 x 10^{-5} (25%)

Source: Human Health and Environmental Benefits of Achieving Alternate CWS for Inhalable Particulates ($PM_{2.5}$, PM_{10}) and Ground Level Ozone. Final Report. Prepared by Paul De Civita, Environment Canada, David Stieb, Health Canada, Lauraine Chestnut, David Mills, Robert Rowe, Stratus Consulting. July 25, 1999. In Compendium of Benefits 99-08-17.

3.2.3.3 Valuation of Morbidity Risks

The monetary values for morbidity effects used in AQVM and the studies from which they are derived for adult chronic bronchitis, respiratory hospital admissions, cardiac hospital admission, emergency room visits, child bronchitis, restricted activity days, asthma symptom days, minor restricted activity days, and acute respiratory symptom days are provided in Appendix B. The studies use willingness to pay and cost of illness measures to assign a dollar value to avoided incidences of each effect. These approaches are discussed in Chapter 8 .

3.2.3.4 *Valuation of Mortality Risks*

The AQVM adopts midpoint and range estimates for a *Value of a Statistical Life* (VSL) based on values used in Rowe et al. (1995)[6] which are similar to the values selected by Cropper and Freeman (1991)[7] based on their review of the literature (see Table 7). The AQVM methodology report indicates that it is important to note that they are based on WTP of the individual for reducing his or her risk of death by a small amount, not on the value of a human. Value of statistical life (VSL) estimates are determined by dividing estimates of average amounts that individuals are willing to pay for a given small reduction in the probability of death by this risk. The VSL estimates available from the literature are based primarily on samples of working age adults. A few of the contingent valuation studies in this literature included individuals of retirement age, but this age is not well represented in the mean VSL values. These VSL estimates are therefore applied only to the under 65-year-old population. Approaches for estimating VSL are discussed in more detail in Chapter 8.

The AQVM used an adjustment to the VSL for those 65 and older of about 75% of the average VSL for adults under 65. An age weighted average VSL for this analysis is calculated on the assumption that 85% of the particulate related deaths are experienced among people 65 and over. The results in Table 6 are default VSL estimates applied to the predicted changes in premature deaths for mortality risk changes associated with changes in particulate matter (including sulphates) and ozone air pollution.

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[6] Rowe, R. D., et al. 1995. The New York Electricity Externality Study. Dobbs Ferry, New York: Oceana Publications.

[7] Cropper and Freemen selected six VSL studies of 21 as “best” for use in policy analysis. Four are wage risk studies and two are contingent valuation studies. The wage-risk estimates range from $3 million to $9 million (1996 CDN dollars), and the contingent valuation estimates range from $4 million to $5 million (1996 CDN dollars). The arithmetic mean of all six selected VSL estimates is about $5 million (1996 $ CDN).

Table 7 Selected Monetary Values for Mortality Risks in AQVM 3.0

Population Group	Selected VSL Estimates (1996 C$ million)		
	Low	Central	High
65 years old	$2.3	$3.9	$7.8
< 65 years old	$3.1	$5.2	$10.4
Age-weighted average VSL*	$2.4	$4.1	$8.2
Probability associated with the estimates for uncertainty analysis	33%	50%**	17%

* Assuming 85% of deaths are individuals aged 65 and over
** The weight selected for the central estimate is 50%, because the underlying WTP estimates are predominantly in the $3 to $6 million range. The high estimate is represented by fewer studies and a somewhat skewed distribution in the available WTP estimates. These weights result in a weighted mean value that approximates the selected central estimate.
Source: Human Health and Environmental Benefits of Achieving Alternate CWS for Inhalable Particulates ($PM_{2.5}$, PM_{10}) and Ground Level Ozone. Final Report. Prepared by Paul De Civita, Environment Canada, David Stieb, Health Canada, Lauraine Chestnut, David Mills, Robert Rowe, Stratus Consulting. July 25, 1999. In Compendium of Benefits 99-08-17.

The selection of probability weights for low, central and high estimates is judgmental because there are several uncertainties in using these estimates in this analysis for which no quantitative information is available. The selected weights reflect the uncertainty in the underlying WTP estimates for small changes in risks of accidental death for working-age adults, but do not fully reflect the uncertainty in using WTP estimates in AQVM 3.0 as, at the time, no studies actually estimated WTP for older adults and those in poor health, who are most at risk.

3.2.3.5 Valuation of Non-health Environmental Benefits

AQVM 3.0 has the capability to assess a number of environmental endpoints including visibility aesthetics benefits, materials benefits from reduced exposure to particulate matter and sulphur, agricultural benefits (corn, soybeans, wheat, tobacco) from reduced exposure to ozone; and recreational fishing benefits resulting from changes in precursor emissions of SO_2, NO_x and VOCs. Household material soiling related to PM was the only non-health endpoint included in the analysis of benefits associated with various CWS for PM and ozone. Five studies (four based on household cleaning costs and one willingness to pay) are used to derive the low, central and high valuation estimates of $1.75, $3.50 and $8.75 per household per year per $\mu g/m^3$ of PM.

3.2.3.6 Results

Avoided health effects and their monetary benefits were estimated for the year 2015 and over a thirty year period (2005 to 2035) assuming the same reduction in pollutant concentration each year. The Compendium of Benefits document provides detailed results tables by province, by CMA and summed across all CMAs for the following ambient levels – for $PM_{2.5}$: 2.5, 20, 30, 40 µg/m^3 [8]; for PM_{10}: 5, 25, 40, 60 and 80 µg/m^3 [9] and for ozone: 60, 70 and 80 ppb [10]. The present value monetary benefits use alternative annual discount rates of 2%, 5% and 7.5% and incorporate a base year of 1996.

Tables 8 to 10 provide central estimates of the present value benefits of achieving alternative reductions in ambient concentrations of $PM_{2.5}$, PM_{10} and ozone in the year 2015 (1996$, discount rate 5% and base year 1996) as presented in Table 11 of the Compendium of Benefits document. The percentage of total benefits associated with each health and environmental endpoint has been included by the Panel to illustrate the distribution of health benefits among the various categories. Table 8 indicates that the total estimated present value of health benefits associated with meeting the current CWS for $PM_{2.5}$ (30 µg/m^3) is $2.1 billion (central estimate). The largest benefits category for both PM and ozone reductions is avoided mortality, representing 79% of the total estimated benefits and valued at $1.6 billion dollars for meeting the current $PM_{2.5}$ standard of 30 µg/m^3. Avoided chronic bronchitis is the next largest benefit category (13% of total benefits or $284 million) followed by restricted activity days (4.2% of total benefits or $89 million). Household material soiling, the only environmental benefit endpoint included in the assessment is valued at $25.8 million or 1.2% of total benefits for the current $PM_{2.5}$ CWS of 30 ug/m^3 and 2% of total benefits for PM_{10} reductions. For ozone reductions, avoided mortality is the dominant health benefit, estimated at approximately $388 million or 95% of total present value benefits in 2015 (5% discount rate) to achieve a CWS of 60 ppb (8-hour).

[8] Tables 1A – 4H of Section 2 of Compendium of Benefits Information
[9] Tables 5A – 9H of Section 2 of Compendium of Benefits Information
[10] Tables 10A – 12H of Section 2 of Compendium of Benefits Information

Table 11 summarizes the provincial benefits estimates as presented in Tables 1F – 12F of the Compendium of Benefits document. For all ambient levels of $PM_{2.5}$, PM_{10} and ozone, the largest share of benefits are expected in Ontario and Quebec. For the current $PM_{2.5}$ CWS of 30 μg/m^3, 70% of total benefits for Canada are estimated to occur in Ontario and 30% of total benefits are estimated to occur in Quebec. For PM_{10} reductions, greater benefits are expected in Quebec (72% of total) than in Ontario (22%) for achieving ambient levels of 60 μg/m^3. For ozone, Ontario is estimated to incur the largest proportion of avoided costs at 72% of total benefits for 70 ppb and 68% of total benefits for 60 ppb, followed by Quebec at 17% for both levels.

Discussion of results of the benefits analysis in the Compendium of Benefits document was limited to identifying the following three general trends: 1) benefits are greater for the more stringent of the proposed scenarios. For example, total estimated benefits increased by $1.2 billion with $PM_{2.5}$ reduction from the current CWS of 30 μg/m^3 (1.5 billion) to 20 μg/m^3 (2.7 billion); 2) Health endpoints responsible for the greatest portion of total benefits are mortality, chronic bronchitis, and restricted activity days (in that order); and 3) Benefits are greatest in the largest cities and the cities with the highest baseline PM_{10} concentrations – Montreal, Toronto and Vancouver.

Table 8: Present Value Benefits of achieving alternative reductions in ambient concentrations of $PM_{2.5}$ Year 2015 – Census Metropolitan Area (CMAs) in the year 2015 (1996$ thousands, % of total benefits (in parentheses), discount rate 5%, and base year 1996)

$PM_{2.5}$ µg/m^3-24-h	Total	Mortality	Chronic Bronchitis	Respiratory Hospital Admissions	Cardiac Hospital Admissions	Emergency Department Visits	Asthma Symptom Days	Restricted Activity Days	Acute Respiratory Symptoms	Child Bronchitis	Household Material Soiling
2.5	$7,273,432	$5,780,723 (79%)	$973,296 (13%)	$1,920 (.02%)	$2,060 (.03%)	$769 (.01%)	$17,522 (.24%)	$305,046 (4.2%)	$94,602 (1.3%)	$8,781 (.12%)	$88,691 (1.2%)
20	$4,170,047	$3,314,359 (79%)	$558,009 (13%)	$1,101 (.02%)	$1,181 (.03%)	$441 (.01%)	$10,046 (.24%)	$174,782 (4.2%)	$54,240 (1.3%)	$5,044 (.12%)	$50,851 (1.2%)
30	$2,121,763	$1,685,858 (79%)	$284,385 (13%)	$560 (.02%)	$601 (.03%)	$224 (.01%)	$5,110 (.24%)	$89,014 (4.2%)	$27,589 (1.3%)	$2,557 (.12%)	$25,865 (1.2%)
40	$665,613	$528,942 (79%)	$89,151 (13%)	$176 (0.2%)	$189 (0.3%)	$70 (0.1%)	$1,603 (.24%)	$27,909 (4.2%)	$8,656 (1.3 %)	804 (.12%)	$8,115 (1.2%)

Source: Compendium of Benefits Information p. 17

**Note that the present value benefits are central estimates. High and low estimates are provided in the Compendium of Benefits document to reflect uncertainties in concentration-response relationships and economic values.

Table 9: Present Value Benefits of achieving alternative reductions in ambient concentrations of PM_{10} Year 2015 – Central Metropolitan Area (CMAs) in the year 2015 (1996$ thousands, % total benefits (in parentheses), discount rate 5%, and base year 1996)

PM_{10} µg/m^{3-} 24-h	Total	Mortality	Chronic Bronchitis	Respiratory Hospital Admissions	Cardiac Hospital Admissions	Emergency Department Visits	Asthma Symptom Days	Restricted Activity Days	Acute Respiratory Symptoms	Child Bronchitis	Household Material Soiling
5	$9,663,230	$7,165,515 (74%)	$571,697 (5.9%)	$2,714 (.08%	$2,923 (.03%)	$1,100 (.01%)	$25,027 (.26%)	$427,464 (4.4%)	$255,711 (2.6%)	$16,626 (0.17%)	$194,435 (2%)
25	$7,411,147	$5,495,597 (74%)	$205,410 (5.9%)	$2,082 (.08%)	$2,242 (.03%)	$844 (.01%)	$19,194 (.26%)	$327,791 (4.4%)	$196,117 (2.6%)	$12,757 (0.17%)	$14,9122 (2%)
40	$4,155,919	$3,081,086 (74%)	$676,598 (5.9%)	$1167 (.08%)	$1,257 (.03%)	$473 (.01%)	$10,761 (.26%)	$183,880 (4.4%)	$109,952 (2.6%)	$7,141 (0.17%)	$83,604 (2%)
60	$1,314,126	$973,225 (74%)	$214,899 (5.9%)	$369 (.08%)	$397 (.03%)	$149 (.01%)	$3,399 (.26%)	$58,320 (4.4%)	$34,731 (2.6%)	$2,229 (0.17%)	$26,408 (2%)
80	$169,162	$125,547 (74%)	$27,428 (5.9%)	$48 (.08%)	$51 (.04%)	$19 (.01%)	$438 (.26%)	$7,448 (4.4%)	$4,480 (2.6%)	$296 (0.17%)	$3,407 (2%)

Source: Compendium of Benefits Information p. 17
**Note that the present value benefits are central estimates. High and low estimates are provided in the Compendium of Benefits document to reflect uncertainties in concentration-response relationships and economic values.

Table 10: Present Value Benefits of achieving alternative reductions in ambient concentrations of Ozone Year 2015 – Central Metropolitan Area (CMAs) in the year 2015 (1996$ thousands, % total benefits (in parentheses), discount rate 5%, and base year 1996)

Ozone ppb 8-hr	Total	Mortality	Chronic Bronchitis	Respiratory Hospital Admissions	Cardiac Hospital Admissions	Emergency Department Visits	Asthma Symptom Days	Restricted Activity Days	Acute Respiratory Symptoms	Child Bronchitis	Household Material Soiling
60	$407,582	$388,183 (95%)	N/A	422 (.10%)	N/A	156 (.04%)	$3,014 (.74%)	$8,414 (2.06%)	$7,396 (1.81%)	N/A	N/A
70	$285,163	$271,589 (95%)	N/A	295 (.10%)	N/A	109 (.04%)	$2,109 (.74%)	$5,887 (2.06))	$5,174 (1.81%)	N/A	N/A
80	$167,266	$159,303 (95%)	N/A	173 (.10%)	N/A	64 (.04))	$1,237 (.74%)	$3,453 (2.06)	$3,035 (1.81%)	N/A	N/A

Source: Compendium of Benefits Information p. 17
**Note that the present value benefits are central estimates. High and low estimates are provided in the Compendium of Benefits document to reflect uncertainties in concentration-response relationships and economic values.

Table 11. Total Present Value Benefits of Achieving Optional PM and Ozone Levels in the year 2015[11] (% of total benefits) 5 percent discount rate 1996 ($ Millions)

LEVEL	ON	QUE	ALTA	MAN	SASK	NS	NB	BC	NFD	CAN
$PM_{2.5}$ µg/m^3 24-hr 2.5	4,006 (55.1)	1744 (24.0)	538 (7.4)	176 (2.4)	-	91 (1.2)	33 (0.5)	684 (9.4)	-	7,272
20	2,678 (64.2)	1,148 (27.5)	172 (4.1)	64 (1.5)	-	52 (1.3)	18 (.5)	37 (.9)	-	4,169
30	1,477 (69.6)	632 (29.8)	-	-	-	9 (.4)	3 (.2)	-	-	2,121
40	501 (75.3)	164 (24.7)	-	-	-	-		-	-	665
PM_{10} µg/m^3 24-hr 5	4,567 (47.3)	3006 (31.1)	597 (6.2)	291 (3.0)	147 (1.5)	81 (.8)	24 (.3)	948 (9.8)	-	9,661
25	3,478 (46.9)	2,568 (34.7)	427 (5.8)	239 (3.2)	103 (1.4)	54 (.7)	17 (.2)	524 (7.1)	-	7,410
40	1,812 (43.6)	1,839 (44.2)	186 (4.5)	145 (3.5)	33 (.8)	3 (.1)	6 (.2)	130 (3.1)	-	4,154
60	284 (21.6)	939 (71.5)	14 (1.1)	36 (2.7)	-	-	-	41 (3.1)	-	1,314
80	9 (5.3)	137 (81.1)	-	11 (6.5)	-	-	-	12 (7.1)	-	169
Ozone ppb 8h 60	275 (67.8)	71 (17.5)	34 (8.4)	.3 (.1)	. 3 (.1)	2 (.5)	.6 (.2)	22 (5.4)	-	405.4
70	205 (72.0)	49 (17.2)	16 (5.6)	-	-	1 (.4)	.4 (.1)	13 (4.6)	.1 (.1)	284.5
80	135 (81.6)	27 (16.4)	-	-	-	.2 (.1)	.2 (.1)	3 (1.8)	-	165.4

[11] Numbers are central estimates. Upper and lower bound estimates are provided in the Compendium of Benefits document Tables 1F – 12 F.

3.2.4 Step 4: Estimation of Emission Reduction Requirements

1. 1994-1996 ambient PM_{10} and $PM_{2.5}$ data for large urban centres were examined to determine the percent reductions in ambient levels that would be required to achieve different optional levels of PM_{10}/$PM_{2.5}$ and ozone CWS (see Table 12 and Table 13) in each province/territory. The percent reductions for PM CWS options were based on which of PM_{10} and $PM_{2.5}$ is causing the greater exceedance of the optional CWS level in different jurisdictions, as indicated in Table 12 .

2. A number of assumptions were made to approximate the reductions in precursor pollutant emissions that would be required in each of the provinces/territories to achieve the optional ambient targets identified in Step 1. The assumptions are as follows[12]:

For ambient PM_{10} and $PM_{2.5:}$

- A 1:1 ratio of percent emission reduction to percent ambient level reduction of emissions of $PM_{2.5}$ and SO_2
- Where PM_{10} is the pollutant causing the greater exceedance of the optional CWS level (between PM_{10} and $PM_{2.5}$) (e.g. in the prairie provinces), emission reductions of NO_x, VOC are determined solely by the reductions needed for ozone
- Where $PM_{2.5}$ is causing the greater exceedance between PM_{10} and $PM_{2.5}$, a 1:0.75 ratio (approximately; percentages rounded) of percent emission reduction to percent ambient level reduction for emissions of NO_x and VOCs. The 0.75 number is based on advice from scientists that the ambient response is probably somewhere in the 0.5 – 1.0 range, the average being 0.75.

For ambient ozone:

- A 1.5:1 ratio of percent emission reduction to percent ambient level reduction for emissions of NO_x and VOCs (noted to compare with Ontario's conclusion that a 45% reduction in NO_x and VOC emissions will close the gap considerably but not quite achieve 80ppb, maximum 1-hour and with the conclusion in the 1990 CCME NO_x/VOC Management Plan that a 50-75% reduction in NO_x and VOC emissions would be required to achieve 82 ppb in the Windsor-Quebec City corridor)

[12] The assumptions are based upon the analyses performed for the Atmospheric Science Expert Panel for the Sulfur in Gasoline and Diesel Fuels Program, the Environment Canada review of proposed U.S. Environmental Protection Agency (EPA) NOx Rules for 22 eastern U.S. states and scientific work for the NOx/VOC program and the Canada-Wide Acid Rain Strategy for Post 2000.

Table 12: PM Reductions Needed in Urban Centres to Achieve Optional PM CWS Levels

CWS LEVEL		Prov/Terr. Ambient Level Reductions Needed (%) – Urban					
PM_{10}	$PM_{2.5}$						
00	50	Ont (<10) $PM_{2.5}$					
90	45	Ont (10-30) $PM_{2.5}$					
80	40	***(<10)**** Ont (30-50) $PM_{2.5}$	Que (<10) $PM_{2.5}$	Man/Alta (<10) PM_{10}			
70	35	***(10-30)**** Ont (30-50) $PM_{2.5}$	Que (10-30) $PM_{2.5}$	Man/Alta (10-30) PM_{10}	NB (<10) $PM_{2.5}$		
60	30	***(30-50)**** Ont (50-70) $PM_{2.5}$	Que (30-50) $PM_{2.5}$	Man/Alta (10-30) PM_{10}	NB (10-30) $PM_{2.5}$	BC(<10) PM_{10}	NS (<10) $PM_{2.5}$
50	25	***(30-50)**** Ont (50-70) $PM_{2.5}/PM_{10}$	Que (30-50) $PM_{2.5}/PM_{10}$	Man/Alta (30-50) PM_{10}	NB (30-50) $PM_{2.5}/PM_{10}$	BC (30-50) PM_{10}	NS (10-30) $PM_{2.5}$
40	20	***(50-70)**** Ont (>70) $PM_{2.5}$	Que (50-70) $PM_{2.5}/PM_{10}$	Man/Alta (50-70) PM_{10}	NB (50-70) $PM_{2.5}/PM_{10}$	BC (30-50) PM_{10}	NS (30-50) $PM_{2.5}$

( ) Ont. without Windsor and Hamilton

Source: CWS Development Committee for PM and Ozone. May 1999. Discussion Paper on PM and Ozone. CWS Scenarios for Consultation. Appendix F.

Table 13: Ozone Reductions Needed in Urban Centres to Achieve Optional Ozone CWS Levels

CWS LEVEL (ppb-8hr)	Prov/Terr Level Reductions Needed (%) – Urban					
80	Ont (10-20)	Que (0-10)				
75	Ont (10-20)	Que (10-20)	NS (0-10)			
70	Ont (20-30)	Que (10-20)	NS (10-20)	NB/Nfld/Alta (0-10)		
65	Ont (20-30)	Que (10-20)	NS (10-20)	NB/Nfld/Alta (10-20)	BC (0-10)	
60	Ont (30-40)	Que (20-30)	NS (20-30)	NB/Nfld/Alta (10-20)	BC (10-20)	Man (0-10)
55	Ont (30-40)	Que (30-40)	NS (30-40)	NB/Nfld/Alta (20-30)	BC (20-30)	Man (10-20)

Source: CWS Development Committee for PM and Ozone. May 1999. Discussion Paper on PM and Ozone. CWS Scenarios for Consultation. Appendix F.

3.2.5 Step 5 – Estimation of the Emission Reduction Costs

3.2.5.1 Overview of Emission Control Cost Study

Stratus Consulting was contracted to apply U.S. EPA emission control cost and control efficiency data to estimate the costs of reducing PM and ozone precursor emissions in Canada. The approach was intended to provide preliminary cost estimates for across-the board percentage emission reductions from the 1995 emission inventory numbers for 38 sectors emitting the top 95% of emissions of $PM_{10,}$ $PM_{2.5}$, SO_2, NO_x and VOCs within each province/territory. A scaling process was used to estimate costs at the 25%, 50% and 75% total emission reduction level for each Standard Industrial Classification (SIC) code. The controls are largely technology based and do not include other measures such as fuel switching. The methodology is described in Section 3.2.5.2.

Cost curves were plotted for each province/territory to identify cost effectiveness break points (% emission reductions above which costs begin to escalate rapidly). These curves were plotted by pollutant-specific emission levels and for the five pollutant emissions in aggregate.

Cost estimates for reducing each of the five target pollutants by the percentages determined in Step 4 were extracted from the provincial/territorial cost tables generated by Stratus Consulting. The Development Committee (CWS DC 1999) notes that the cost estimates can be considered to provide an order of magnitude perspective only, because they are based upon ballpark emission reduction estimates. Also noted is the inherent assumption that in near-border regions influenced by the transboundary flow of pollutants from the U.S., comparable reductions in emissions from U.S. sources would also have to occur to achieve CWS target levels. Total cost for Canada was determined by summing the provincial/territorial costs for those optional levels. Some costs were reduced to remove double counting, particularly for PM_{10} and $PM_{2.5}$ where some of the sources and technologies for control were the same. This was very approximate (e.g. scanning of PM_{10} and $PM_{2.5}$ cost data to get a sense of amount of overlap, then reducing one of PM_{10} or $PM_{2.5}$ control costs (the non-controlling pollutant) by an approximate percentage, such as 50 or 75%).

From the range of optional cost estimates (see Table 14), the incremental and total costs involved in reaching each successive optional CWS level for PM and ozone was determined (see Table 15 and Table 16).

3.2.5.2 Methodology for Stratus Consulting Emission Control Cost Study

Approximately 300,000 data points were used to analyze three control options for each of five pollutants emitted by 150 industries, in all Canadian provinces and territories. The methodology involved summarizing and merging existing U.S. EPA and Environment Canada datasets described below, and applying assumptions and industry expertise to the results. To analyze the costs of various emission reduction levels across industries, it was first necessary to obtain U.S. EPA cost and effectiveness data for each control option. These data were then combined with Canadian emissions inventory data from

Environment Canada's 1995 Residual Discharge Information System (RDIS) to estimate the cost to each industry to achieve emission reduction goals, and these estimates were aggregated to the provincial and national levels to estimate total emission reduction cost.

3.2.5.3 Results

The *Compendium of Cost Information* provides detailed tables of reduction costs and emission abatement cost curves for each pollutant by province and nationally. Table 14, Table 15 and Table 16 below provide the estimated costs to achieve the optional PM and ozone CWS. To provide some perspective on what the costs of the various options mean relative to the Canadian economy, the costs estimates were presented as a percentage of provincial/territorial and national 1995 GDP. The results indicated that CWS more stringent than 50/25/65 $PM_{10}/PM_{2.5}/O_3$ exceeded the range for minimal impacts on productivity, competitiveness, employment and economic growth for some provinces (CWS DC, 1999 Table G-9).

Table 14: Estimated Costs of Achieving Optional Combinations of PM and Ozone CWS Levels ($M/yr)

PM_{10}	$PM_{2.5}$	O_3	ONT.	QUE.	ALTA.	MAN.	N.S.	N.B.	B.C.	CANADA
70	35	70	770	105	72	0	10	5	0	962
70	35	65	1400	370	200	0	28	13	32	2,043
60	30	65	1630	470	300	21	28	13	32	2,494
50	25	65	2280	580	440	87	28	24	32	3,471
50	25	60	6200	900	670	122	57	33	120	8,102
50	20	55	>12400	3940	850	1650	120	60	240	>19,260
40	20	55	>>14000	4300	1240	2010	220	110	250	>>22,130

>indicates costs would be greater than the numbers shown since cost estimates could not be made for the full emission reduction levels required to achieve that particular CWS option
>>indicates costs would be much greater than the numbers indicated

Source: CWS Development Committee for PM and Ozone. May 1999. Discussion Paper on PM and Ozone. CWS Scenarios for Consultation. Table 5.3 p. 21.

Table 15: Estimated Incremental and Total Costs of Achieving Optional PM CWS Levels ($M/yr)

$PM_{10}/PM_{2.5}$ 24 hr,98th perc.	ONT.	QUE.	ALTA.	MAN.	N.S.	N.B.	B.C.	CANADA
70/35	170	0	0	0	0	0	0	170
70/35 60/30	**230**	**100**	**100**	**21**	**0**	**0**	**0**	**451**
60/30	400	100	100	21	0	0	0	620
60/30 50/25	**650**	**110**	**140**	**66**	**0**	**11**	**0**	**977**
50/25	1,050	210	240	87	0	11	0	1,600
50/25 40/20	**>1,600**	**360**	**390**	**360**	**100**	**61**	**10**	**>2,870**
40/20	>>2,650	570	630	447	100	61	10	>>4,470

>indicates costs would be greater than the numbers shown since cost estimates could not be made for the full emission reduction levels required to achieve that particular CWS option
>>indicates costs would be much greater than the numbers indicated

Source: CWS Development Committee for PM and Ozone. May 1999. Discussion Paper on PM and Ozone. CWS Scenarios for Consultation. Table 5.4 p. 21.

Table 16: Estimated Incremental and Total Costs of Achieving Optional Ozone CWS Levels ($M/yr)

8 hr,4th highest ozone level	ONT.	QUE.	ALTA.	MAN.	N.S.	N.B.	B.C.	CANADA
70	600	105	72	0	10	5	0	790
70 65	**630**	**265**	**128**	**0**	**18**	**8**	**32**	**1,071**
65	1,230	370	200	0	28	13	32	1,871
65 60	**3,920**	**320**	**230**	**35**	**29**	**9**	**88**	**4,631**
60	5,150	690	430	35	57	22	120	6,502
60 55	**>6,200**	**3,040**	**180**	**1,528**	**63**	**27**	**120**	**>11,158**
55	>>11,350	3730	610	1,563	120	49	240	17,660

>indicates costs would be greater than the numbers shown since cost estimates could not be made for the full emission reduction levels required to achieve that particular CWS option
>>indicates costs would be much greater than the numbers indicated
Source: CWS Development Committee for PM and Ozone. May 1999. Discussion Paper on PM and Ozone. CWS Scenarios for Consultation. Table 5.5 p. 22.

3.2.6 Step 6: Comparison of Costs and Benefits

While the comparison of avoided costs (benefits) and anticipated costs of achieving emission reduction is identified as a separate step in the overall methodology for selecting PM and ozone CWS level scenarios for stakeholder consideration, the Panel notes that a direct monetary comparison does not appear as a formal part of the analysis according to the review of the documentation and discussions with Environment Canada officials. Technical feasibility and costs were indicated in the Discussion Paper on PM and ozone CWS Scenarios for Consultation as the rationale for selecting lower bound CWS levels for PM and ozone, while upper bounds were determined in part by health protection considerations, citing epidemiological support for the assumption of no threshold for health effects. This lack of direct comparison of monetized benefits and costs may have been partly due to time and resource constraints as the benefits analyses were still in progress at the time of the May 1999 National stakeholder consultation meeting when PM and ozone standard scenarios were presented for stakeholder consideration.

3.2.7 Consideration of Baseline

The Compendium of Costs document indicates that it may be appropriate to weed out "anyway costs" – costs that will be incurred regardless of the CWS program (e.g. mobile source controls) – and the portion of the costs that should be attributed to other programs for common measures (e.g. climate change, acid rain). The Panel notes that the CWS CBA does not attempt to define or quantify PM and ozone reductions that are likely to occur anyway under current or forthcoming regulations governing air emissions and quality. While linkages with other Canadian and U.S. air quality management initiatives are recognized by the CWS Development Committee, the associated reductions in PM and ozone precursor emissions were not factored into the estimates of benefits and costs. Consideration of the impact of the related air quality management programs described in Table 17 would improve the basis for estimating incremental benefits and costs of CWS.

Table 17: Current and Proposed Canadian Air Quality Management Initiatives

AIR QUALITY INITIATIVE	DESCRIPTION
CANADA-U.S. OZONE ANNEX UNDER THE 1991 CANADA-U.S. AIR QUALITY AGREEMENT	Negotiations are ongoing, both countries aim to reach a signed agreement by the end of 2000. Canada is pledging that emission controls on power plants in central Canada will meet or exceed the U.S. NO_x requirement for fossil fuel power plants. U.S. requirements – NO_x state implementation plans, are expected to result in U.S. power plants meeting a NO_x emission rate that is approximately three times more stringent that the current regulated rate in the U.S.
SULPHUR IN GASOLINE AND DIESEL FUEL	Two stage phase in of low sulphur gasoline. Jan. 1 2002 – limit to avg. value of 150 ppm with a never to be exceeded max. of 200 ppm. Jan. 1, 2005 level will be limited to an average value of 20 ppm with never to be exceed max. of 80 ppm. Further reductions in sulphur content of diesel fuel to 15 ppm were proposed in May 2000.
CANADA-WIDE ACID RAIN STRATEGY FOR POST-2000	Signed in Oct. 1998. Negotiations are ongoing. SO2 emission reduction target of 50% by 2015. Ontario and other eastern provinces will establish targets and schedules for emission reduction. Further SO2 emission reduction commitment from the U.S. will be pursued by Fed govt. with support from provinces and territories.
VEHICLE EMISSIONS AND FUELS	Standards will be phased in from model year 2004 to 2009. Goal is to meet or exceed U.S. standards. New standard will result in a 77% reduction on average in smog causing emissions such as nitrous oxides for new passenger vehicles and 95% for light duty trucks including SUVs.
CANADIAN AND U.S. CLIMATE CHANGE PROGRAMS	Canada target of 6% reduction in greenhouse gas emission reduction by 2008-2012 U.S. target of 7% reduction in GHG emissions by 2008-2012. Specific measures not identified
EASTERN U.S. OZONE REDUCTION PROGRAM	Overall NO_x emission reductions in the range of 50% by 2007
U.S. PM REDUCTION PROGRAM	Most reductions in 2008-2012, full implementation in 2015-2017

All of the foregoing measures will reduce the costs and the benefits that are attributable to CWS for PM and ozone.

3.3 References

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4 Emission Inventories and Air Quality Changes from Emissions Reductions

Ultimately, in order to achieve projected benefits, whether they be ecological in nature or human health related, it will be necessary to attain specified levels of ambient air quality. Ambient air quality is determined by the emission of a complex mixture of species into the atmosphere, both in gaseous and particulate form, their transport from the source region by the winds, possible transformation by chemical reactions within the atmospheric envelope, and their final loss from the atmosphere by scavenging by clouds, rainfall and deposition onto the surface. In addition, several species such as ozone and some types of particles are not emitted directly but are formed by reactions from gaseous precursor species.

Emissions and meteorology affect air quality on a variety of scales from continental, right down to local impacts of an industrial stack around the corner. Since at mid-latitudes the winds are westerly (from the west), on the continental scale, Canadian air quality can be affected by long range transport of particles and ozone from Asia. In a similar manner, Canada exports air pollution to Europe. On the regional (provincial) scale, air quality can principally be affected by either local emissions, long range transport, or a mixture of the two. For the case of local sources, the Greater Vancouver Regional District (GVRD) is a prime example of local emissions being trapped within the confines of the Fraser Valley and being recirculated to give very high ozone and particulate values. Nova Scotia also suffers from high levels of ozone from time to time as a result of long range transport of ozone from the New York region or from the Windsor Quebec corridor. The latter region, which suffers the most frequently from ozone exceedances, is affected by long range transport from the States that lie to the SW such as Michigan, and Ohio and occurs largely during periods of stable high pressure over eastern North America. During these types of meteorological events the polluted air is recirculated (but on a larger scale than occurs for Vancouver) and the emissions can build up over a period of several days. However, concurrently the emissions from the heavily populated regions of SW Ontario are also important contributors to the degradation of air quality in this region of Canada.

A critical link in the cost-benefit analysis chain, is the one that relates expected reductions in gas phase and particulate emissions with concomitant changes in air quality, i.e. the changes in the atmospheric abundances of the emitted species. A critical question is thus *"How good is our knowledge of the emission inventories?"* For clearly if we cannot specify current emission inventories with any degree of confidence how can we know that reductions applied will have the desired impact? An important related question is *"How reliable are the relationships between reductions in emissions and improvements in air quality that have been used for CWS?"* In this chapter, we will assess the current status of the emission inventory for Canada and the methods used in the CWS study to connect reductions in emission inventories with corresponding ambient air quality. We will also discuss uncertainties and possible improvements to this part of the CBA process.

4.1 Emission Inventories

An emissions inventory for each species of interest is a fundamental requirement for comprehensive air quality modeling and CBA. Unfortunately, this is one of the areas where there is much uncertainty, both in total amount and in the spatial and temporal distribution of emissions. Table 18 gives estimates of the yearly emissions in kilotonnes, for total particulate matter[13] (TPM), $PM_{2.5}$, PM_{10}-$PM_{2.5}$ which is the amount of particulate with diameters between 2.5 and 10 microns, SO_x, NO_x, VOCs and CO for Canada, for 1995, prepared by Environment Canada (EC, 2000). For aerosols or particulate matter (PM), the estimates in Table 18 refer to direct emissions of PM. $PM_{2.5}$ and PM_{10} emissions are subsets of TPM. In addition to direct emissions of PM from sources such as fuel combustion, PM can also be formed as a result of reactions in the atmosphere such as gas-to-particle conversion by precursor species. Both SO_x and NO_x are precursor

[13] We shall use the term total particulate matter or TPM to refer to all forms of particle matter entering the atmosphere. This material comes in a variety of sizes, shapes and compositions and it is often characterized by an effective radius or diameter (effective since the particles need not be spherical.). Thus $PM_{2.5}$ refers to that part of TPM which has an effective aerodynamic diameter less that 2.5 microns and is often described as the "fine" part of the distribution while larger particles are referred to the "coarse fraction". PM_{10} refers to that part of the TPM distribution which has diameters less than 10 microns and thus includes the $PM_{2.5}$ part of the distribution. We shall use the term particulate matter or PM when we do not need to discriminate between the various parts of the distribution.

emissions for PM and this PM formation is referred to as a secondary source of PM. For the $PM_{2.5}$ range, secondary sources can represent a substantial fraction of the total $PM_{2.5}$ source. NO_x and VOCs are precursor emissions for ozone formation and VOCs can also be precursors for PM composed largely of organic carbon. CO has been included since it plays an important role in ozone formation.

Table 18 Annual Canadian Emissions Estimates for 1995

	AIR POLLUTANT IN KT (% OF TOTAL)						
SOURCE	TPM	PM_{10}-$PM_{2.5}$	$PM_{2.5}$	SO_x	NO_x	VOC	CO
Industrial Source	621(4)	115(3)	172(11)	1,950(73)	620(25)	941(26)	2,177(13)
Non-industrial fuel combustion	225(1)	22(1)	157(10)	566(21)	333(14)	407(11)	1,079(6)
Transportation	97(1)	13(0)	83(5)	136(5)	1,290(52)	734(21)	6,708(35)
Miscellaneous	21(0)	5(0)	9(1)	2(0)	1(0)	550(15)	14(0)
Open Sources	14,717(94)	3,696(96)	1,097(72)	1(0)	216(9)	937(26)	7,103(41)
National Total	15,684	3,852	1,519	2,654	2,464	3,575	17,128

Notes: 1) PM_{10}-$PM_{2.5}$ represents the size fraction that lies between 2.5 and 10 microns.
2)Non-industrial fuel combustion includes electrical power generation.
3)These figures do not include the source of emissions from biogenics which have been estimated at about 14 MT (Deslauriers, 1996).
4)Incineration is less than 1% for all species and has not been included.

Source: Environment Canada www2.ec.gc.ca/pdb/ape/cape_home_e.cfm

Emissions inventories are prepared using a number of different methods. Measurements from sources can be made for large emitters such as stacks, although such measurements are very expensive and therefore are rarely done. Also they provide only a snapshot of the emissions from the stack or industrial facility. Estimates of emission factors for similar activities can be used and estimates can be provided from industrial and government agencies. Engineering estimates based on detailed knowledge of the emission source are often used. The Canadian Residual Discharge Information System (RDIS, 2000) is such a data source.

In Table 18, "Industrial Sources" indicates industries such as the wood industry, iron and steel industry, mining and smelting, and pulp and paper. "Non-Industrial fuel combustion" refers to commercial and residential fuel combustion, residential wood burning and electric power generation. “Transportation” includes gasoline and diesel vehicles, railroads, aircraft and tire wear. “Miscellaneous” includes emissions from pesticides and fertilizer application, general solvent use and structural fires. The “Open Source” includes emissions from agricultural sources, construction operations, dust from paved and unpaved roads, landfill sites, and forest fires. Municipal and industrial incineration appear to produce less than 1% of the emissions and have not been included in the Table. Of course, while not important in a national or regional sense they could be quite important locally. Altogether the inventory contains estimates from more than 60 industrial and non-industrial activities and more than 4,600 facilities have been assessed. An important omission from Table 18 is that of NH_3 which is important for secondary aerosol formation.[14]

Biogenic emissions originate from trees and agriculture and consist of VOCs such as isoprene and terpenes which are very reactive in the atmosphere. Some of the reaction products resulting from the degradation of biogenics can act as sources of PM, or react with NO to form ozone. Some of the biogenics are of low vapour pressure and thus may condense and act as growth sites for PM. These emissions are not included in Table 18. However, from the 1990 emission inventory (Deslauriers, 1996) biogenic emissions from across Canada are estimated to be about 14 MT. NO_x is also produced naturally from forests and agricultural land but there are no Canada-wide estimates of NO_x from these sources although it has been included in regional modeling (e.g. Plummer, 1999).

Table 18 shows that the estimate of direct production of particulate mass in Canada from industrial and non-industrial sources totals, typical of about 1995, is about one megatonne (MT). However, the major source of PM is from open sources which include agricultural tilling and wind erosion (1.8MT), construction (2.4MT), paved roads (2.5MT), unpaved roads (6.8MT) and forest fires (0.8MT) with a total of 14.7MT. Much of this material is quite “large”, being generated from the surface by wind. Because it is “heavy” the large section of the PM falls out quite rapidly. Nevertheless, about 33% is PM_{10} which has a

[14] Estimates of NH_3 emissions have recently been obtained for Regional studies in SW Ontario, but are not generally available for Canada wide studies.

longer residence time in the atmosphere than the larger settleable (PM>10) fraction of PM and can be transported further from the source. $PM_{2.5}$ has an even longer residence time than PM_{10} and about 1.1MT is produced from open sources of which 53% is from forest fires. For PM in the size range of a fraction of a micron the deposition and rainout lifetimes are much longer than for other sizes. Thus a considerable part of the fine fraction or $PM_{2.5}$ will be subject to long range transport.

The major sources of the fine fraction, $PM_{2.5}$, listed in Table 18 come from combustion. On a country-wide scale the minor contributors to direct $PM_{2.5}$ emissions are about 11% from the industrial sources, 5% from transportation, 10% from non-industrial fuel combustion and <1% from miscellaneous sources. The largest percentage fine particle source, 72%, comes from open sources of which 38% is from forest fires. The total is about 1.5MT of $PM_{2.5.}$ Since forest fires usually occur during dry periods in the summer there will be periods when the contribution to the fine fraction will be much larger than 38%.

As noted above these figures do not include secondary sources of $PM_{2.5}$ and, in particular, biogenic sources both of which could be considerable. The contribution of secondary sources to the $PM_{2.5}$ fraction is quite variable but estimates from measurements and source apportionment, indicate that it can be as large as 50% .

SO_2, NO_x and VOCs can all be regarded as precursor species for secondary PM formation. Annual Canada-wide emissions are about 2.6 MT for SO_2, and 2.5MT for NO_x, most of which are from industrial sources, non-industrial fuel combustion and transportation. Annual VOC emissions are about 3.6 MT. One important source of VOCs omitted from the 1990 inventory is forest fires (0.9MT). Important sources of VOCs are industrial sources (26%), transportation (21%), miscellaneous solvent use (15%), forest fires (open sources) (26%) and non-industrial fuel use (11%). These sources do not include biogenic emissions of NO_x and VOCs which can be large. Natural emissions of SO_2 or precursors are small over land areas.

Table 18 provides a picture of annual Canada-wide emissions. An important aspect, from the perspective of assessing impact, is the spatial heterogeneity of the emissions. This is particularly important for regional and urban scales. Figure 1 through Figure 4 show 1995 PM, NO_x, VOC and biogenic emissions (Environment Canada, 2000) with resolution of about a few hundred km. Some of the emissions are from point sources such as stacks whereas other emissions, such as transportation can be thought of as line emissions, while open source emissions and biogenics are widely distributed. For example, biogenic emissions of VOCs may represent about 14MT (see comment 3 in Table 18) and dominate the other sources listed in Table 18. The various species that make up the biogenics are emitted from heavily forested regions, often well away from large NO_x sources which might lead to a generation of ozone. Nevertheless, they could, via chemical reactions, be oxidized to compounds which could form $PM_{2.5}$ which may be transported to other regions of Canada. Uncertainty in the estimation of the biogenic emissions from Canada's extensive forests and their expected seasonal variability could translate into uncertainty in the estimation of background $PM_{2.5}$ in various regions of Canada. As might be expected most of the NO_x emissions are near urban and or industrial areas.

Except for the GVRD, emissions estimates are rarely prepared with a sufficiently fine spatial resolution on an on-going basis to be useful for physical-based modeling (see below). Emission inventories required for physical-based modeling have to be prepared specially using information such as population densities and fuel use to redistribute the emissions with sufficiently high spatial and temporal resolution. Using the currently available Canadian software this is an expensive exercise. In addition, physical-based models have the capacity to operate at 1 km horizontal resolution but they are often forced to use emissions estimates prepared at 20 km horizontal resolution, thus degrading the information that might be obtained from modeling studies. Thus, development of software that can provide, not only total Canada-wide emissions, but emissions with improved spatial and temporal resolution, is an important area in need of more work.

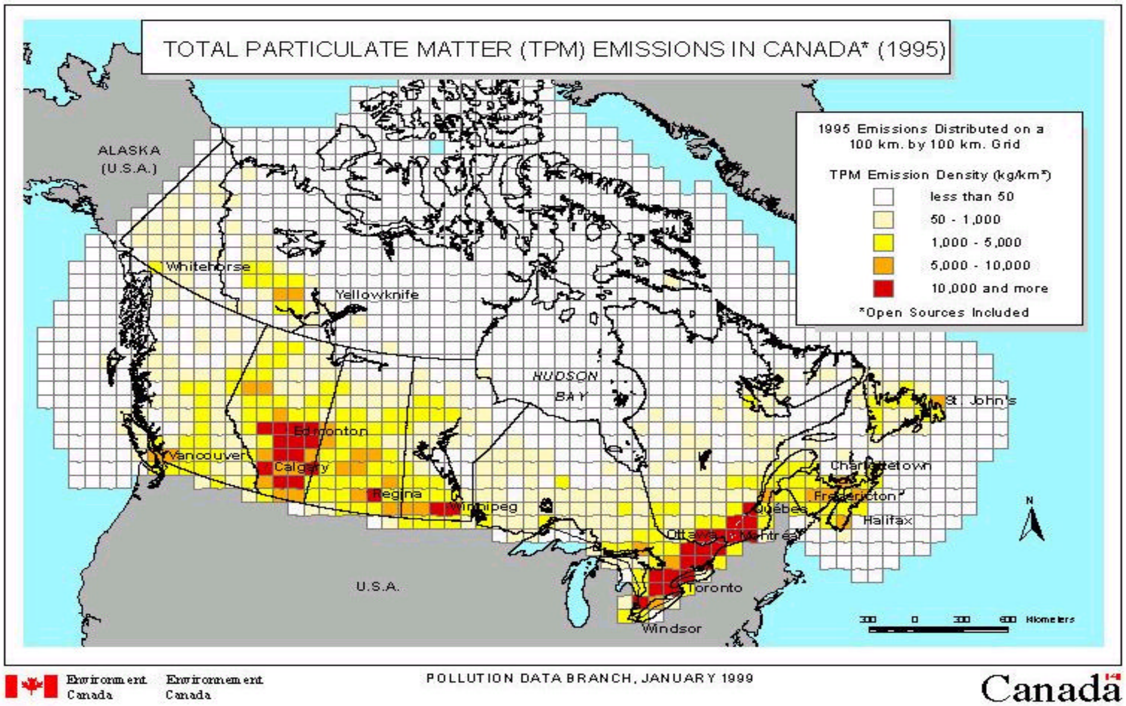

Figure 1 Total Particulate Matter Emissions in Canada for 1995. (Source: Environment Canada www2.ec.gc.ca/pdb/ape/cape_home_e.cfm).

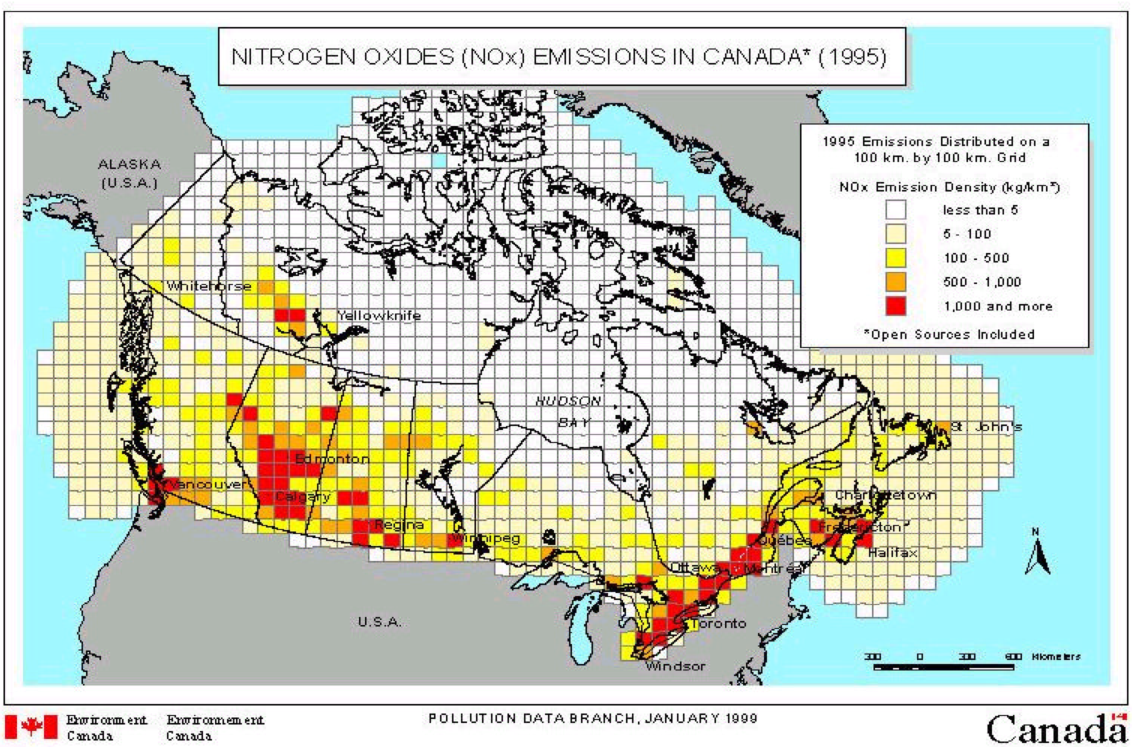

Figure 2 : Nitrogen Oxide (NOx) Emissions in Canada for 1995. (Source: Environment Canada www2.ec.gc.ca/pdb/ape/cape_home_e.cfm)

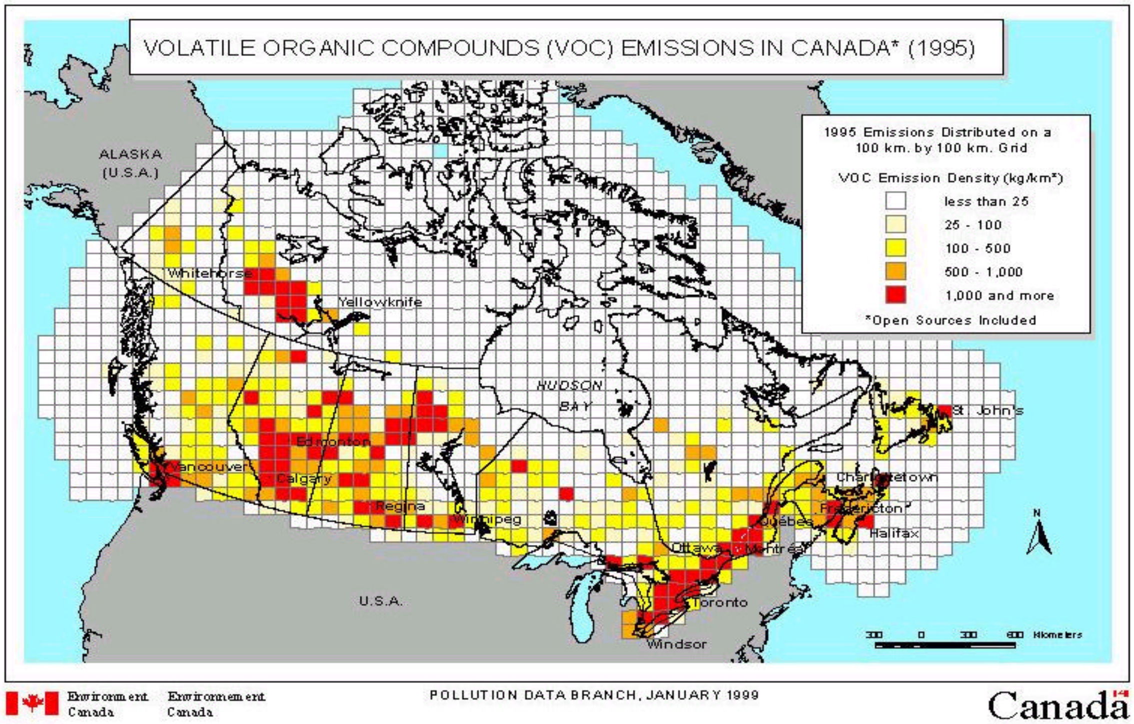

Figure 3 : Volatile Organic Compound (VOC) Emissions in Canada for 1995. (Source: Environment Canada www2.ec.gc.ca/pdb/ape/cape_home_e.cfm)

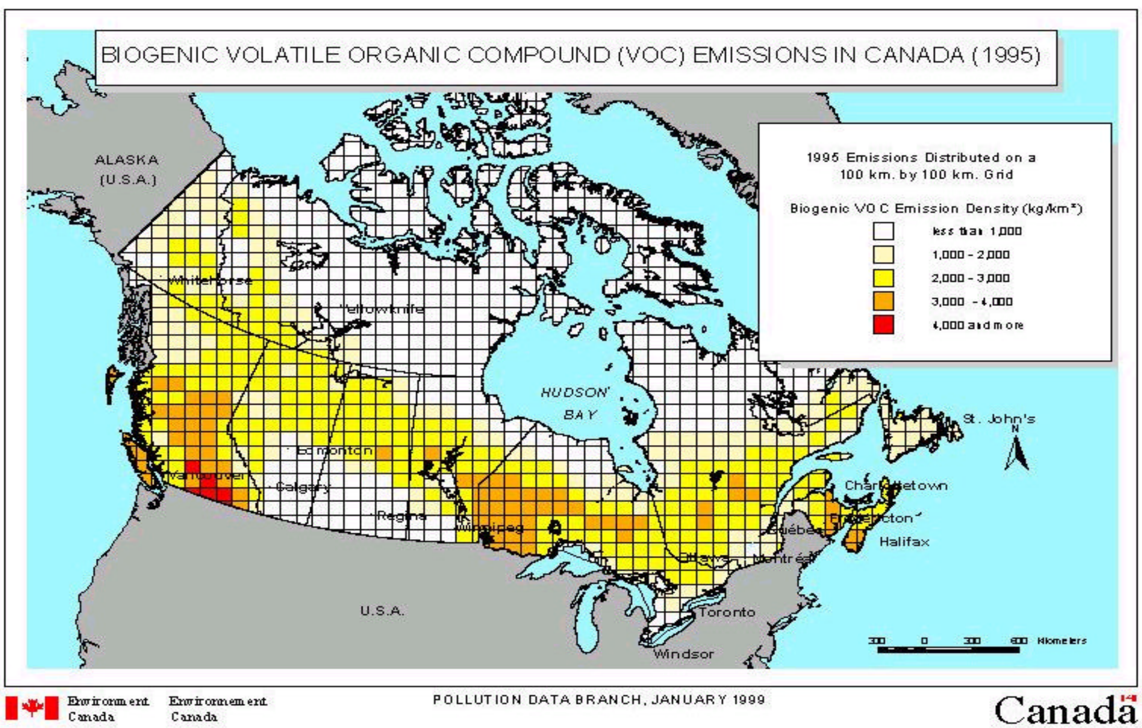

Figure 4 : Biogenic VOC Emissions in Canada for 1995. (Source: Environment Canada www2.ec.gc.ca/pdb/ape/cape_home_e.cfm)

4.1.1 Reduction of Ambient levels of Pollution: Emissions Modification

There are several methods that can be used to estimate the modifications that must be made to the estimation of emission fields in order to obtain compliance with CWS. Perhaps the best method, discussed in detail below, is to use complex 3D physical-based modeling along the lines that has been attempted already for the acid rain problem in Canada (Venkatram et al., 1988) and the U.S. (Change et al., 1987).

For the U.S. EPA cost-benefit study on the impacts of the Clean Air Act (U.S. EPA, 1999) physical-based modeling was used (see Section 4.4). This study was a requirement of section 812 of the Clean Air Act Amendment and assessed the potential impact of controlling emissions according to the Act for the period 1990 to 2010. Physical-based modeling was used to assess, on a point by point basis, the changes in ozone, $PM_{2.5}$ and PM_{10} with and without regulations. The ratio of these changes were then applied to the measurement sites used for assessing health benefits. For NO, NO_2, SO_2 and CO a simple scaling based on the change in local emissions was used to scale the field measurements.

Aside from the physical realism, physical-based modeling also allows for the examination of the impact of targeted reduction in emissions in a systematic fashion. In order to capture the variability associated with the natural atmosphere, climatologies of model runs must be assembled and this requires a substantial commitment of human resources. Quite often such models have only been used to address specific incidents that might last from a few days to perhaps as long as a week. However, in order to capture the variability associated with the real atmosphere, models should be run over periods of several months. In the past it has required extensive human resources to assemble all the meteorological data, run and analyse the gigabytes of data. However, this has changed and in particular, the use of on-line models offers a more self-consistent approach (e.g., Bouchet et al., 1999, Kasibhatla and Chamiedes, 2000).

An alternative method of linking reductions in ambient air quality concentrations and emissions reduction is to adopt a statistical approach. This is the method applied in the CWS study. Given a set of CWS atmospheric concentration limits for PM (TPM, PM_{10}

and $PM_{2.5}$) and ozone that must be met, there are two steps to assess what levels of reduction of emissions are necessary. First, a method or algorithm must be devised to estimate by how much current ambient species levels must be decreased in order to comply with the CWS. This, of course, requires an adequate knowledge of current levels of PM and ozone. (The current status of measurements of ozone is much better than that for PM.) Next, given the required reduction in ambient levels of PM and ozone, the decrease in emissions required to achieve these levels must be estimated. This statistical method does not readily allow itself to be targeted to reductions in emissions such as the transportation or industrial sectors, although it does take account of the climatological aspects of the problem by using air quality data over several years with different patterns. Some of the details of the various methods applied by the CWS study to estimate reduction in ambient species level with a decrease in emissions are described in more detail below. But first we describe some of the inherent problems that occur when trying to estimate the response of an inherently non-linear system by external estimation rather than by internal scaling.

4.2 Non-linear behaviour in atmospheric air quality

The difficulty in evaluating the relationship between emission changes and atmospheric concentration changes varies from species to species and can depend on whether the species is emitted directly such as NO_x, CO, SO_2 and direct-PM or, like ozone and secondary-PM, results from the interaction of precursors. The existence of a statistical relationship will depend on the complexity of the chemical production and loss of the species and its atmospheric lifetime. If the atmospheric lifetime is short then it is more likely that there will be a linear relationship between emissions and species abundance at the measurement location. This is not the case for ozone or secondary PM. Longer lived species are subject to long range transport and thus local species abundances reflect emissions and chemistry from quite distant locations. For example, the formation of ozone in the background troposphere is a result of the interaction between NO_x, VOCs and sunlight with a contribution from transport from the stratosphere: ozone is destroyed by UV sunlight and by deposition to (destruction on) surfaces. The lifetime of ozone can be several months and long range transport plays an important role in maintaining its distribution. However, as mentioned above, during extreme pollution events in locations such as the GVRD, ozone is primarily generated locally.

Figure 5 illustrates the nature of the non-linear problem. It shows the results of ozone generated in a box model moving over Atlanta with various amounts of NO_x and VOCs emitted into the box from within the city. The dot gives the estimated NOx and VOC emissions in Atlanta. If this diagram is used to investigate responses to changes in emissions for Atlanta we note the following. For fixed VOC emissions, it can be seen that *decreases* in NO_x from the standard one (i.e. movement in the vertical direction) elicit rather small changes in ozone while large *increases* in NO_x actually lead to a decrease in ozone! Some of this ozone would be hidden as titrated NO_2 but much of it would be tied up in products such as nitric acid. On the other hand, decreasing VOC emissions (horizontal movement along the axis) leads to measurable decrease in ozone, whereas increasing VOCs in this case by a large amount would not have much effect. Such a situation is said to be VOC-limited. If Atlanta lay to the right hand side of the ozone "ridge" then the situation would be reversed and the ozone concentrations would be much more sensitive to NO_x emissions and the situation would be identified as NO_x-limited. For many cities, ozone generation appears to be VOC-limited while generally the surrounding rural regions are NOx limited. We stress that this is a simplistic model and the conditions will be different for each city. For example, Atlanta may be rather more characteristic of the southern U.S. since biogenic emissions from trees, which are unavoidable, form a large part of the VOC emission budget during the summer months. More complex models also exhibit the types of responses described above.

Another example of complex chemical behaviour is that of the formation of fine particulate matter by secondary processes. Secondary $PM_{2.5}$ can be formed by gas-to-particle conversion processes such as, ozone or OH reacting with heavier VOCs emitted from anthropogenic or natural sources, oxidation of SO_2 to SO_4 and oxidation of NO to NO_3. The acid forms of SO_4 and NO_3 can then be neutralised by reaction with NH_3 forming particles. This non-direct relationship between the formation of species in the atmosphere and emissions complicates the assessment of expected change in atmospheric pollutant levels due to emission changes.

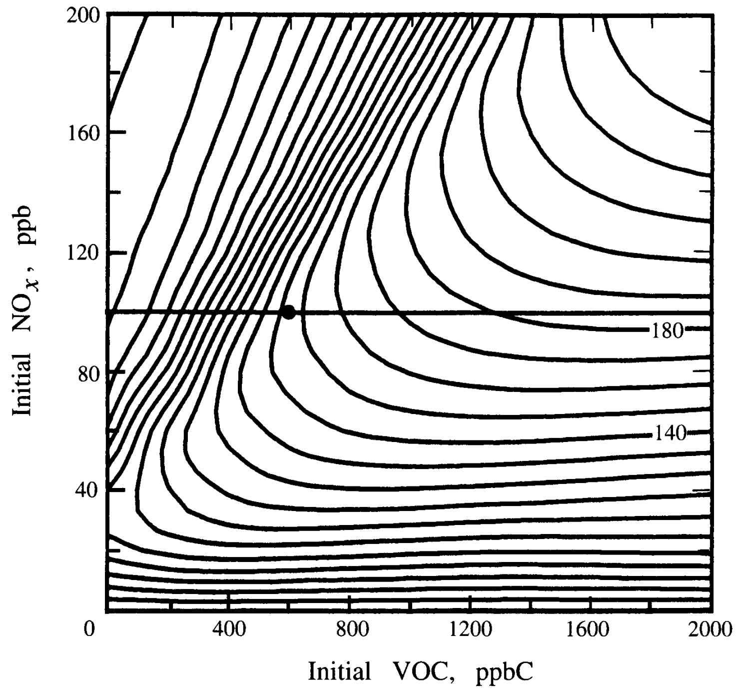

Figure 5: Ozone Contour Plot (ppbv) from Seinfeld and Pandis (1998)

This plot shows levels of ozone generated over Atlanta for various levels of NO_x and VOCs using a trajectory chemical box model. The dot gives the results for emissions thought to be typical of Atlanta. The ozone "ridge" running from bottom right to top left is typical of such diagrams and highlights the non-linear nature of ozone generation for a simple situation. Thus if VOCs are reduced by 50% say, without reducing NO_x, the ozone will decrease from 145 ppbv to about 110 ppbv or about 24%. However, doubling VOCs from 600 to 1200 ppbC would only lead to a 24% increase. Alternatively, if NO_x is reduced by 50%, the VOCs remaining constant, ozone only decreases by about 9%. If the NO_x were doubled ozone would actually decrease by 66% to about 50 ppbv. The location of Atlanta on the diagram illustrates VOC-limited ozone production.

On a more speculative level, a similar situation may exist with the oxidation of SO_2 to SO_4 and reductions in NO_x emissions within the urban environment, even though on a regional scale the changes in sulphate deposition are linearly related to reduction in the source emissions. However, in urban areas where most of the impacted population live, a reduction in NO_x will alter the urban chemistry modifying ozone, which in turn will affect OH and H_2O_2 generation both of which play important roles in the oxidation of SO_2 to SO_4. Also within the urban environment, reduction in NO_x will change OH levels which locally will affect the oxidation rate of SO_2 to SO_4 and thus the local production of SO_4 particulate which occupies the fine particle range. A similar effect may obtain for the oxidation of SO_2 to SO_4 within cloud droplets where it may be limited by the availability of H_2O_2.

Similar effects may have taken place in the numerical experiment reported by Meng et al. (1997) and some of their results are shown in Table 20. Their models included complex gas phase and aerosol chemistry and they found that decreases in NO_x and VOC emissions which affect ozone did not lead to a proportionate decrease in secondary PM.

Table 20 shows that the Riverside region is similar to that of Atlanta in that it is VOC-limited so that with NO_x fixed, a 50% reduction in VOC leads to a 34% reduction in ozone. However, there is a concomitant 25% increase in PM-nitrate and a 20% increase in $PM_{2.5}$ mass, most likely associated with an increase in OH due to decreased VOCs. If the same VOC reduction scenarios are repeated for a simultaneous 25% reduction in NO_x, there are only small changes in ozone, nitrate and $PM_{2.5}$ mass with changing VOCs although the amounts have decreased by 5-10% compared to the 0% reduction case. It is unlikely that these results could have been predicted from a statistical model and, even though the Meng et al. model has limitations, the results serve as a warning regarding simplistic scaling to assess the effects of changing emissions.

Table 20 Simulated Maximum 1-hour average concentrations for Riverside California, on 28th August 1987 for various combinations of VOC and NO_x reduction from base estimated 1987 basin wide emissions (after Meng et al., 1997).

	VOC REDUCTION			
NO_x Reduction	Chemical Species	0%	25%	50%
0%	Ozone (ppb)	180	146	119
	$PM_{2.5}$ NO_3(μgm^{-3})	97	119	121
	$PM_{2.5}$ mass (μgm^{-3})	146	173	175
25%	Ozone (ppb)	175	172	170
	$PM_{2.5}$ NO_3(μgm^{-3})	87	87	89
	$PM_{2.5}$ mass (μgm^{-3})	133	134	137
50%	Ozone (ppb)	168	150	135
	$PM_{2.5}$ NO_3(μgm^{-3})	76	69	71
	$PM_{2.5}$ mass (μgm^{-3})	120	133	124

Not all atmospheric chemical relationships are non-linear. As noted in Section 4.5, linear statistical relationships between emissions and some species atmospheric concentrations are expected. For example, as noted above, certain classes of emission such as larger (heavier) PM are lost from the atmosphere relatively rapidly by deposition or rainout.

Although we may generally obtain a linear source-receptor relationship on a regional scale such may not be obtained on an urban scale, where the majority of the adverse impacts will be realized. Another important aspect related to non-linearity on a local level concerns long range transport for aerosols in the fine particle mode. We emphasize the removal of these fine particulates by wash out and deposition, especially in the 0.1 to 1.0 micron size range is slower than for other sizes so that they may be transported much longer distances away from the source region. Thus the size distribution of the aerosol may be altered by long range transport.

4.3 Scaling ambient levels to conform to CWS

Atmospheric concentrations of PM and ozone vary diurnally and seasonally and the normalized number of times a particular concentration occurs is called the distribution frequency. There is concern that extreme events rather than average air quality may drive health effects (see Chapter 5). Knowledge of the distribution frequency of atmospheric species concentrations is important in assessing what changes in air quality will have to be made to comply with CWS. In the CWS study, the required reduction in PM emissions was obtained by simply scaling the current PM concentrations by the CWS level for a particular scenario, divided by the 3rd highest maximum at that site[15]. The evaluation of the reduction factor for ozone was slightly more complex. An attempt was made to incorporate modeling results and other analysis. These studies indicated that the reduction factor for "moderate" levels of ozone, associated with emission changes, was smaller than for higher levels. Thus for ozone, the frequency of an observation was scaled linearly above a threshold of 90 ppbv. For measurements between this threshold, and a lower threshold taken to be the nominal background level of 40 ppbv, the rollback was applied linearly, being zero at 40 ppbv to the maximum value at the high threshold. This is described in more detail in Section 3.2.2 of this report.

[15] The 3rd highest maximum is chosen in an attempt to ameliorate the effects of outliers, i.e. anomalous extreme events.

4.3.1 Emissions Rollbacks

The scaling factors from the above analysis for PM and ozone from the CWS study were applied to emissions as described in the "Estimation of benefits and costs of achieving optimal levels" study (CWS Development Committee for PM and Ozone, May 1999). For ambient PM_{10} and $PM_{2.5}$ it was assumed that there is a 1:1 linear relationship between reduction in emissions and change in atmospheric concentrations, i.e. a 1% reduction in emissions will result in a 1% reduction in PM_{10}, $PM_{2.5}$ and SO_2. The rationale for this scaling was quoted as being based on work done for the "Sulphur Report" (discussed below) for which the anticipated changes would be small. The anticipated changes required to comply with CWS are much larger and are likely to lie outside the linear limits of the approximation.

For NO_x and VOCs the reductions are determined somewhat differently depending on whether or not they may be related to PM_{10} or $PM_{2.5}$ in some fashion. Presumably the rationale applied was that as combustion is a source of not only NO_x and VOCs but also of the fine fraction, $PM_{2.5}$, then adjustment of one will impact the others. However, combustion is not a major source of PM_{10} so that adjustment of PM_{10} should be independent of NO_x and VOC. Thus if PM_{10} (rather than $PM_{2.5}$) is the particle species contributing more to exceeding the CWS, such as might occur on the prairies, NO_x and VOC emission reductions are constrained by the requirements necessary to reduce ozone. However, if $PM_{2.5}$ is the pollutant causing the exceedence (rather than PM_{10}), then a 4:3 ratio of emission reduction of $PM_{2.5}$ to NO_x and VOC emission reduction is applied. Thus a 1% reduction in $PM_{2.5}$ required to meet the CWS would require a 0.75% reduction in NO_x and VOC: it is not clear whether or not this reduction is in addition to that required (see next paragraph) to meet CWS ozone standards. The 4:3 ratio used is quoted as being from the advice of scientists, presumably from the "Sulphur: Atmospheric Report" (ASEP, 1997). The study by Meng et al. (1997) discussed above clearly shows that the above ratios are gross over-simplifications which largely arise from treating the ozone changes separately from the PM changes. There is a clear case for comprehensive modeling, in which the combined system of PM, VOCs and NO_x is treated together (see Section 4.4).

For ambient ozone, the CWS analysis assumed that a 3:2 ratio will apply to NO_x and VOCs, i.e. a 100% reduction in both NO_x and VOCs will result in a 67% reduction in excess ozone, i.e. the difference between the ambient level and the assumed ozone background of about 40 ppbv. This reduction assumes that NO_x and VOCs will be scaled by similar amounts which may not be appropriate for certain cities where a reduction in VOCs alone may be more appropriate as discussed for Figure 2 above.

Clearly the question of whether or not linearity can be applied to the anticipated changes in emissions to deduce air quality changes is an important one. In the case of the sulphur emission from fuel, the assumption of a linear response of the system is more reasonable since the putative changes in the air quality were quite small (in the few percent range). However, in the CWS study under review the changes envisaged are much larger. Consequently, it is likely that the simple algorithms utilized to extrapolate the effects are much less robust. In fact, based on the discussion in the preceding section, much of the rationale used by CWS was faulty. The assumption that a local change in emissions will result in a local change in gas phase and PM species is reasonable for *some* of them, such as CO and perhaps also PM_{10}, with the caveats noted above. Unfortunately, neither secondary-$PM_{2.5}$ nor ozone fit into this “reasonable” category and the uncertainty is quite high. In fact, the direction of the change could possibly be incorrect in some cases.

4.4 Physical-based modeling

As was noted above, a more internally consistent method of assessing the effect of decreased emissions is to use detailed physical-based three dimensional source-receptor modeling. In this case, the model simulates all important physical and chemical processes involved, ranging from the emissions being released into the atmosphere, their transport, possible transformations, secondary creation of gas and PM species, and losses from the atmosphere by deposition and rainout. Typical Canadian models that could be applied for ozone changes are the CHRONOS model (Pudykiewicz et al., 1997), MC2-AQ model (Plummer, 1999, Kaminski et al., 2000) and the Regional Climate Model (RCM) (Bouchet et al., 1999). CHRONOS is a chemical transport model which uses meteorology supplied by the Canadian mesoscale model MC2, while MC2-AQ has the oxidant chemistry on-line with the meteorological framework. The RCM is a relatively low resolution model designed to investigate regional climate. Currently there are no working Canadian models with both oxidant chemistry and aerosol formation and chemistry although several are in the development stage. Some of the above models are quite similar to those used in the EPA cost-benefit study (U.S. EPA, 1999).

There were 3 main models used for the EPA study (U.S. EPA, 1999). The UAM (Urban Airshed Model) (SAI, 1990) is an ozone air quality model that was used to predict ozone changes based on modified emissions, while the RADM/RTP (Regional Acid Deposition Model with a Regional Particulate Matter module) model (Denis et al., 1993; Binkowski and Shankar, 1995 and references therein) was one of the models used for investigating aerosol impacts. The RADM/RTP model was used at low horizontal resolution which degrades the accuracy with which the impacts can be assessed. The third model was the Regulatory Modeling System for Aerosols and Acid Deposition (REMSAD) which is based on the UAM-V model.

None of the above models are comprehensive or integrated. By that it is meant that they can treat both ozone and PM chemistry and processes. A relatively new model that has both oxidant chemistry and an improved aerosol model is Models3/Community Multiscale Air Quality (CMAQ) model (CMAQ, 2000), but this is still in the throes of being validated for aerosols. Integrated models of this nature are clearly a more mature way to attack the problem of estimating questions of changing emissions and air quality. However, the air quality modeling community are moving to even more comprehensive modeling concepts. The models are described as on-line models, which means that meteorology and air chemistry are combined and evolve simultaneously so that feedbacks (e.g., NCAR, 2000) are included. A Canadian on-line model which does not yet have PM chemistry is MC2-AQ. We note that the US National Academy of Science reports on PM research priorities have also, in addition to their primary focus on health effects, endorsed improved physical-based modeling (NRC, 1998, 1999, 2001).

The use of physical-based models is limited by several factors. One critical limitation is uncertainty in the size distribution, composition and amount of input emissions (NRC, 2001). These are probably not determined locally to better than a factor of two for gaseous emissions, while for particulate emissions, both their mass and size distribution are much more uncertain than equivalent parameters for gas phase emissions. Table 18 shows that from time to time, $PM_{2.5}$ is likely to be dominated by long range transport from forest fires. Although local emissions might be quite uncertain, Canada-wide emissions of SO_2 and NO_x are probably accurate to within 20-30% since total fuel use provides an integrating constraint.

Another important factor is knowledge of the gas phase and heterogeneous chemistry. Most air quality models today do a reasonable job of reproducing ozone levels for episodes, but there remain disconcerting problems such as poor representation of the location and magnitude of ozone maxima which may be due to remaining uncertainties in the underlying chemistry (and related to particle formation and heterogeneous chemistry), and/or to uncertainties in emissions or poor representation of the meteorology.[16] The level of knowledge for emission and formation of aerosols is much more uncertain than for gas phase chemistry and in some respects this is a more complex problem.

Another limitation of the models is the species resolution of the gas phase chemistry. Every day thousands of different hydrocarbons of differing reactivities are emitted into the atmosphere, but the details of their chemical breakdown are not well characterized. In models, this is handled by "lumping" the hydrocarbons with similar properties together as a single species, such as light alkanes, heavy alkanes, light alkenes, heavy alkenes etc. and assuming similar reaction properties for their breakdown products. Likewise the enormous complexity of the aerosols is unlikely to be captured by their limited representation by a few modes or types within a model. At one level, aerosol modeling within the 3D context is at a much more rudimentary stage than that used for oxidant chemistry. Nevertheless, models are improving all the time as more complete physical and chemical representations of the real atmosphere are included. They do permit emission reductions to be assessed in an internally consistent manner and permit analysis of targeted reductions for specific sectors. Their use also allows the investigation of the effects of changes in meteorology from year to year, and the inclusion of the effects of a changing climate in which predictions will have to be made.

Because they include detailed physical processes and feedbacks, physical-based models can be used for diagnosis of atmospheric conditions. One important diagnostic capability is in the generation of source-receptor relationships that can connect source emissions from different downwind sites (possibly a thousand or more kilometres away) with the receptor site (see for example Stratus (2000)). However, development of these relationships with a fully developed model can be still quite computationally expensive since many emission and climatological scenarios have to be investigated to explore a useful range of parameter space. Consequently, less complete models are often used for this type of study.

4.5 CWS methodology

This section describes the rationale for the emissions adjustment used in the CWS study, briefly outlined in Sections 3.2.2, 3.2.4 and 4.3. According to the CWS study documentation, the underlying assumptions used to assess the atmospheric changes were based on the Atmospheric Science Expert Panel (ASEP) section of the Sulphur in Gasoline and Diesel Fuels report (ASEP, 1997). Thus we have had to assess much of the rollback and scaling methods from the ASEP report.

Ratio method: The main approach used for gaseous emissions was based on the use of a long lived species such as CO which is emitted within an urban area and transported out of the region without reacting (the lifetime of CO ranges from a few months at the equator to 8-9 months at winter high latitudes). Thus, the difference between the local/urban CO mixing ratio and the background mixing ratio is proportional to the emission rate (molecules cm^{-2} s^{-1}) and this constant of proportionality is given by $\tau/(H*n)$

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[16] Although, there is a growing suspicion that these types of air quality model can represent the climatology of ozone more robustly than specific incidents (Bouchet et al., 1999; Kasibhatla and Chameides, 2000)

where H is the height of the planetary boundary layer[17] into which the emissions are emitted, n is the total air density and τ is a characteristic time for transport out of the urban region. If sources are similarly distributed then the constant should be similar. Using NO_x as a source gas, the Atmospheric Science Expert Panel found a factor of 2 discrepancy between emissions using CO. Part of this difference may be attributed to the fact that NO_x has a lifetime less than a day within an urban region. However, it should still be longer than a typical transport time. Thus there is a strong suggestion that the CO or NO_x emissions may be in error by about a factor of two and this must be taken into account when assessing uncertainty of local emissions.

Dispersion and source-receptor modeling: In the Sulphur report (ASEP, 1997) several other types of modeling were used to support the emission to air quality concentration estimates. Dispersion (Gaussian plume) modeling was used to assess air quality changes due to vehicular emissions. In addition, source-receptor statistical modeling, whereby similarities between the chemical characteristics of the emitted aerosol and the aerosol composition measured in the atmosphere was used to assess the source strength. This is potentially quite a useful technique. However, as was pointed out in the Sulphur Report, for this to be credible really requires a composition analysis of the potential sources in order to correctly assess their contribution to the air concentrations. This composition analysis is often lacking for many Canadian sources.

One aspect of this type of chemically speciated based source-receptor analysis is that it is implicitly assumed that the emissions of species all occur relatively nearby. In terms of regional pollution episodes, a large fraction of the source may come from outside the populated region of interest. Thus attempts to rollback emissions from local sources, as assumed in the CWS analysis, may not achieve target ambient levels as long range transport of pollutants from distant sources may cause PM and ozone exceedances. To

[17] The planetary boundary layer (PBL), also called the atmospheric boundary layer and over the ocean the marine boundary layer, is the atmospheric region next to the surface in which the air is rapidly mixed vertically during the daytime to a (generic) height of about 1 km. At night, over land, the thickness of the layer shrinks to about 100 m due to the lack of input of solar energy to drive the mixing. During the night most emissions of pollutants into the PBL are trapped unless they are from stacks or have sufficient buoyant energy to “break through” the PBL.

better account for long range emission sources, physical-based modeling should be utilized.

Box modeling: To estimate the impact of changing NO_x and VOC emissions on ozone levels, the results of previous Eulerian model studies for acid rain and ozone generation were utilized. These were older studies using relatively low horizontal resolution physical-based modeling, for which the models provided a state-of-the-art (at that time) assessment of changes in ozone for different emission reduction scenarios. These studies provide guidance regarding connections between emission rollbacks and ambient concentrations, but we note that many of the non-linear effects described above, occur particularly in the urban environment which requires high horizontal resolution studies. Thus there will be locations, mostly near major NO_x sources, where ozone may actually increase if NO_x (and VOC) emissions are cut back because of titration of the ozone to NO_2. In fact, this type of effect could more readily be assessed by looking at what is commonly called “odd oxygen” which is the sum of ozone and NO_2: although the toxic effects of NO_2 are different from those of ozone. Away from the suburbs and in surrounding countryside the ozone is expected to decrease with decreasing NO_x and VOC emissions.

As noted above, a large fraction of the aerosols are in the fine fraction ($PM_{2.5}$) and of these a large proportion can be due to secondary processes such as the oxidation of SO_2 to sulphate or oxidation of VOCs, and these secondary aerosols can also change their composition by reaction with other species such as NH_3 or by adsorption. Assessing their change due to emission reduction is very difficult. The principal method used in the CWS analysis to derive the conditions applied was Lagrangian photochemical box modeling. Although the box model is quite detailed, it does not take account of long range transport effects as the box is applied locally to the regime of the urban area. Also, the aerosol models that were applied are quite limited in terms of representing the actual physical situation.

4.6 Future possibilities

In the last decade there has been an enormous improvement in computer capabilities, which make the use of physical-based models more realistic for applications such as the CWS analysis. However, one limitation is the accuracy with which models reproduce actual measurements. As noted above, the situation appears reasonable for ozone in the sense that models are doing an adequate job of representing reality with the caveats mentioned above regarding phase and magnitude differences for ozone maxima. An important additional caveat, is that few other species have been measured with the same thoroughness or success as ozone. For aerosols the situation is worse as the measurement database is not as extensive, particularly in Canada. Thus one must treat the predictions from the aerosol Eulerian models with some caution. Yet at the same time they do represent a powerful tool if used circumspectly, in combination with statistically based methods.

4.7 Other considerations

Baseline: One of the aspects that has not been consistently addressed with the AQVM part of the CWS study is that of emissions baseline. Particularly as CWS looks to the future, atmospheric emission changes are expected to be driven by other external factors. We note that implementation of the sulphur in gasoline legislation requiring reductions across Canada to 150 ppm by 2002 and 30 ppm by the end of 2004 will impact emissions and thus the atmospheric concentrations forming the baseline. There will also be a response to the Kyoto Protocol to reduce CO_2 emissions which will likely impact other pollutants. Thus there are several issues that will affect ozone and particulate levels in Canada but which have not been addressed by the CWS study report, presumably due to a lack of resources.

Changing Stratosphere/Ozone: There is a well documented connection between UVB radiation and both melanoma and non-melanoma skin cancer. During the last two decades stratospheric (i.e. upper atmospheric) ozone has been decreasing due to the effects of anthropogenic chlorine (i.e. CFCs) and, as a result, the solar UVB levels at the surface have been increasing (e.g., Wardle et al., 1997). However, the high levels of surface ozone and aerosols have been ameliorating this impact of the increased UVB in the following manner. The increased tropospheric ozone and particulate levels that occur during the summer, act to reduce the UVB radiation transmitted by depleted stratospheric ozone: the aerosols act to scatter the UVB radiation and in some cases they also absorb it. This extra scattering and extra ozone result in the absorption of the UVB light. Thus, high levels of tropospheric ozone could be regarded as a benefit from the perspective of CWS CBA by counteracting the deleterious results of our inadvertent geo-engineering experiment with CFCs on the stratospheric ozone layer. The CBA of the U.S. EPA National Ambient Air Quality Standard for ozone has been investigated by Lutter and Woltz (1997) who estimated that health costs could outweigh health benefits.[18] In any case, this issue is part of a changing (atmospheric) baseline since currently the application of the Montreal (and later) Protocols on chlorine emissions are beginning to have an impact on the amount of chlorine in the stratosphere, and over the next 50 years or so the ozone layer is likely to return to its pre-1970s state. Clearly these effects could be included as part of a CBA.

A related impact due to the higher levels of UVB radiation in the troposphere associated with the perturbed stratospheric ozone layer is that, in the general background atmosphere with generally lower levels of ozone and aerosols, the chemical activity of the troposphere will be enhanced. This occurs because the increased UVB will, while actually destroying more ozone, lead to an enhanced production of the tropospheric "detergent", OH. This, in turn, will lead to a reduction in the lifetime of GHGs such as methane. Clearly the costing of scenarios such as this becomes very difficult. Nonetheless, future CBA should attempt to assess these very real effects.

[18] The benefits estimated by Lutter and Woltz (1997) are probably overestimated since they assumed that the application of CAAA would lead to a decrease of 10 ppbv in seasonally averaged ozone levels. This is probably an overestimate since it is often overlooked that background ozone is actually the global accumulation or generation of regional chemistry in the industrial countries and biomass burning in the developing countries along with ozone input from the stratosphere. Thus local or regional air quality problems have a global context and this should not be overlooked.

Tropospheric Climate Effects: As the CWS study looks to the future one aspect that has been mentioned but was not addressed in any manner, again presumably due to lack of resources, is what may be expected in terms of air quality in an atmosphere in which CO_2 has increased substantially (see also Section 6.3 for a discussion on GHGs). The scientific consensus is that the atmosphere will undergo a general warming (e.g., IPCC, 1996). However, the regional impacts of such a changed climate are not yet clear. One might anticipate increased warm episodes in the summer which would tend to lead to an increased number of pollution episodes for the same level of emissions. Perhaps one saving grace will be that with an adherence to the Kyoto Agreement, the concomitant decrease in CO_2 will also bring the benefit of decreased emissions as a result of the improved technology. However, one must be wary that the combustion technology which leads to a decrease of CO_2 does not result in an increase of NO_x, CO and VOCs. On the other hand, even though anthropogenic emissions may decrease, if there are more warm episodes then we can expect that biogenic emissions, which are temperature and light sensitive, may increase. Already we have seen in Table 18 that the 14MT of reactive biogenic VOCs dominate anthropogenic VOC emissions on a Canada-wide basis. Included among the biogenic emissions are VOCs and also NO_x which can give rise to ozone production. Likewise an increase in temperatures, if also associated with reduced rainfall in forested areas, may result in increased frequency of forest fires. As noted above, forest fires already may represent up to about 40% of the fine particle mass on an average basis.

Future Studies: As research contributes to our understanding of anthropogenic impacts on our atmosphere and to our understanding of climate change, the interconnectedness of processes that affect the atmosphere is slowly becoming clearer. Aerosols which can cause health problems also act to cool the atmosphere and so temporarily ameliorate the global warming effects of GHGs. Carbon-based aerosols (soot), on the other hand, can act to enhance global warming. As noted above, increased surface UVB due to decreasing *stratospheric* ozone can be (temporarily) ameliorated by enhanced *tropospheric* ozone and higher levels of aerosols. Some of these aspects can and should be included as part of a future CBA study.

4.8 Policy Applications

The policy options for achieving improved ambient air quality are limited by those factors that can be changed by regulatory activity. In this case, that means emission reductions. The limitations in scope for influencing ambient air quality must be understood, if the expectations for improved air quality for given levels of emission reduction are to be realistic.

A number of issues arise that may interfere with a linear correspondence between implementing specific, regulated emission reductions and achieving corresponding improvements in air quality. These include factors such as:

- relative contributions of controllable emission sources to ambient air quality levels
- ambient air quality contributions from trans-border sources
- non-uniform geographic distribution of emission sources

The nature of this problem is illustrated by reference to air pollutant inventory data in Table 18. These data show that emissions regulations that target only point sources (including utilities) could achieve a major impact on SO_2 but only a negligible impact on primary sources of PM_{10} and $PM_{2.5}$. For example, estimated benefits could not be assumed to correspond to a 75% reduction in ambient $PM_{2.5}$ if a 75% reduction in $PM_{2.5}$ emissions was mandated. As noted above, ozone is a secondary pollutant that is determined by emissions of NO_x and VOC. Likewise, a substantial portion of fine particulate is also generated from secondary sources determined by VOC, NO_x and SO_2 emissions. Thus the relative contribution of both primary and secondary sources must be understood to make meaningful links between policies to reduce pollutant emissions and expected ambient air quality benefits.

The primary and secondary source aspects clearly point to varying strategies that may be much more effective than across-the-board reductions. Thus, again referring to Table 18, if SO_2 emissions are a major contributor to secondary $PM_{2.5}$ in ambient air, then targeting reduction strategies for SO_2 at point sources and utilities may be far more effective at achieving low ambient levels of $PM_{2.5}$ than requiring primary removal of $PM_{2.5}$ across-the-board. Additionally, there are serious questions about the value of focusing solely on anthropogenic sources of $PM_{2.5}$, both direct and secondary, when such a large fraction is from forest fires and open sources.

Likewise, we could not assume that even a 75% reduction of all emission sources would achieve a 75% reduction in the levels of the reduced pollutant in ambient air if a substantial source of that pollutant comes from trans-border pollution. This is a substantial issue in those regions of Canada that border on heavily populated and industrialized regions of the U.S.

4.9 Jurisdictional Issues

The regulatory scheme in Canada is very different from that in the U.S. While the U.S. Clean Air Act is binding federal legislation that can set very specific requirements that must be implemented in every state, there is much less federal authority to legislate on environmental matters in Canada. Likewise, even where authority may exist in Canada, the practice has been to allow most environmental regulation to be implemented by the provinces. This regulatory requirement has the effect that there are distinctly different regulatory schemes in place across Canada. Even with Canada-Wide Standards set for ambient air quality, the means to achieving emission reductions to support those ambient air quality goals have to be implemented by means of the differing regulatory schemes maintained by each province. This reality seems likely to lead to substantial differences in the levels of emission reduction that will ultimately be achieved. For the same reasons costs may differ.

In addition, with increasing globalization of markets and industrial activity, the specific regulatory schemes implemented in any one jurisdiction will be influenced by the practices in other jurisdictions. Technology of production will be driven by market forces and considerations other than regulated emission reductions. Thus the costs for achieving future emission reductions that may be estimated by assuming across-the-board implementation of a particular technology is likely to differ from the actual costs and practices that will be adopted in reality.

4.10 Conclusions

The major uncertainties associated with the part of the CWS study that deals with the reduction of ambient air quality levels to meet CWS is the likely non-linearity associated with the reduction of emissions and the concomitant changes in air quality. In the CWS study it was assumed that the effects would be linear. This seems unlikely in many cases, bearing in mind the major role that open sources and forest fires in particular play in the formation of $PM_{2.5}$. In addition, the emission database is quite uncertain, particularly on a detailed level as compared to Canada wide averages usually presented.

The primary and secondary source aspects clearly point to varying strategies that may be much more effective than across-the-board reductions. For example, targeting reduction strategies for SO_2 at point sources and utilities may be far more effective at achieving low ambient levels of $PM_{2.5}$ than requiring primary removal of $PM_{2.5}$ across-the-board. Likewise, the value of focusing solely on anthropogenic sources on $PM_{2.5}$, both direct and secondary, is questionable when such a large fraction of emissions is from forest fires and open sources.

4.11 Recommendations

Definition of the baseline is essential in a CBA study. The baseline may change because of factors such as the implementation of current or future regulations, changing economic conditions, and possible changes in atmospheric climate. Thus the Panel recommends that future CWS studies have the resources to include an appropriate and transparent definition of the baseline with reasonable estimation of the relevant components.

It is still not evident if extreme or chronic events with respect to high PM and ozone levels are important in causing health impacts and there are insufficient PM_{10} and $PM_{2.5}$ continuous measurements to address this question. Also measurements of PM_{10} and $PM_{2.5}$ are critical for the evaluation of emission inventories and 3D physical-based modeling. Furthermore, it will be necessary to have adequate spatial measurements to ensure both the efficacy of the reductions and compliance with the reductions. To help address these important questions and given the paucity of continuous PM_{10} and $PM_{2.5}$ data the Panel recommends that a more systematic continuous measuring program be adopted for PM_{10} and $PM_{2.5}$.

One of the aspects that pervades all aspects of the CWS study is the requirement for an accurate emission inventory, with good spatial and temporal characteristics: these are necessary for both CBA and physical-based modeling. But this will require the active collaboration of federal and provincial governments and the industrial sector with involvement of NGOs. Thus the Panel recommends that adequate resources and administrative structures be provided at the federal and provincial level for improving the spatial and temporal resolution of emission inventories of PM_{10}, $PM_{2.5}$ and ozone precursor species across Canada. This could involve support from a consortium of many levels of government (from federal to municipal), industry, and NGOs. We note that the emission inventory work that is proceeding in the Greater Vancouver Regional District provides an example to the rest of the country. Furthermore, given the importance of NH_3 in the formation of secondary $PM_{2.5}$ and the lack of an adequate baseline inventory, the Panel recommends that NH_3 should be added to emission inventory studies.

One means of attacking the problem of accurately relating reduction of emissions to the attainment of CWS is to use physical-based 3D models combining both gas phase and aerosol formation and chemistry, embedded in a meteorologically-based model. Use of such models also allows a more detailed and targeted approach to be taken to infer impacts. This work is currently on-going in Canada. The Panel recommends support for the on-going work on comprehensive or integrated 3D physical-based aerosol modeling that includes both ozone and PM chemistry and meteorology in Canada and its use for estimating ambient air quality changes with targeted reductions.

Source-receptor statistical modeling potentially represents a powerful method of identifying emission sources, but this requires a detailed chemical knowledge of the emitted pollutants. This is rarely available in Canada and many studies have had to use surrogates from the U.S. The Panel recommends that every effort should be made to develop Canadian emission data.

4.12 Summary

Table 21 provides a summary of the CWS approach to estimating emissions and air quality changes from PM and ozone emissions reductions and the Panel's assessment of the key limitations, uncertainties and recommendations for alternative approaches.

Table 21: Summary of Panel's Assessment of CWS Approach to Estimating Air Quality Changes Associated with PM and Ozone Emissions Reduction

ISSUE	**EMISSIONS ESTIMATION**
CWS APPROACH	Baseline emissions data from Environment Canada 1995 Residual Discharge Inventory System (RDIS) – fixed baseline No direct account taken of secondary aerosol production Transboundary (TB) sources not directly taken into account Natural emissions not directly included but indirectly included via subtraction of background levels Air Quality (AQ)– used several year average for ozone, TPM, PM_{10} and $PM_{2.5}$
PANEL CRITIQUE Key Limitations	RDIS – on a global basis NO_x amounts probably accurate to about 20-30% based on fuel usage. PM sources are much more uncertain. Spatial emissions are also much more uncertain. Transboundary sources– small effect for Greater Vancouver Regional District (GVRD), 100% for Atlantic region, about 50% for the Windsor Quebec Corridor (WQC) Natural emissions – uncertain, but likely to vary from important to dominant away from urban centres, both for VOCs and $PM_{2.5}$ Open sources – potentially major contribution to PM_{10}, but with large uncertainty Limited existing knowledge of composition of aerosols Monitoring – currently limited mostly to every 6 days for PM_{10}, $PM_{2.5}$, limited $PM_{2.5}$ data
RELATIVE UNCERTAINTIES (Probably Minor, Potentially Major)[19]	RDIS + natural sources + secondary sources – Potentially major uncertainties in spatial distribution of emissions and PM emissions in particular Transboundary sources – potentially major for ozone and PM Natural sources– potentially major for $PM_{2.5}$ away from urban centres, probably minor for ozone AQ monitoring: probably minor for ozone while composition of aerosols is not well determined on a regular basis. This is of concern for assessment of health effects using epidemiological studies.

[19] Likely Significance Relative to Key Uncertainties on Net Benefits Estimate: Probably minor (alternative assumption or approach could influence overall estimate by <20% difference), Potentially major (>20% difference). Adapted from U.S. EPA study The Benefits and Costs of the Clean Air Act 1990 to 2010" Nov. 1990 study in which 5% difference was used see pg. 21, 33, 65, 79, 98.

DIRECTION OF BIAS[20]	Difficult to determine for ozone. In urban centres, will depend on whether or not in a non-linear regime. This will depend on the NO_x/VOC ratio. If this is altered it could affect the linearity. PM is likely to be dominated by natural emissions away from urban centres; open sources remain uncertain and thus the cut backs applied to anthropogenic sources could sometimes be dominated by the unregulated sources.
RECOMMENDATION/ ALTERNATIVE INPUTS, TOOLS, APPROACHES	Improvement of the emission database on a year by year basis. Include forecast for baseline, projections/effects of other regulations coming on line. This would be inline with the GVRD. Improved spatial details for emissions. Transboundary – this would seem to be best handled by physical based (Eulerian) 3D modeling. Additional use of source-receptor analysis would be very useful but will require upgrading and measuring Canadian source speciation. Need to improve estimates of natural emissions. Could improve year by year effect using remote sensing technology and measurements. Correlation methods with proper source specification would improve the situation. Upgrade the current monitoring system to continuous monitoring. More rural monitoring to help assess open source/background emissions. More information on the composition of aerosols both for source identification and epidemiological studies.
ISSUE	**TRANSLATING EMISSIONS CHANGES TO AIR QUALITY CHANGES**
CWS APPROACH	Reduction of ambient ozone and PM levels to match CWS – quasi linear for ozone and linear for $PM_{2.5}$ and PM_{10} reduction factor, R. Linear (scaled) application of R to emissions without (direct) consideration of long range transport or natural emissions.
PANEL CRITIQUE Key Limitations	Linearity would appear to be too limiting for ozone, perhaps also for $PM_{2.5}$ and PM_{10}. Data for correlation studies estimated from modeling studies that were (a) at limited horizontal resolution and (b) reductions applied in the model were across the board.
RELATIVE UNCERTAINTIES (Probably Minor, Potentially Major)	Potentially major
DIRECTION OF BIAS	Likely to overestimate changes in air quality for a given reduction in emissions. Could even get the direction of change wrong in certain cases.
RECOMMENDATION/ ALTERNATIVE INPUTS, TOOLS, APPROACHES	Use physical based modeling with improved emission inventory: this would address both limitations simultaneously. Development of Canadian emission database, particularly for particle emissions, would allow for an improved assessment of effects by statistical methods. Use of integrated (3-D) Model with ozone and PM capabilities embedded in meteorological framework which is state of the art.

[20] The U.S. EPA report "The Benefits and Costs of the Clean Air Act 1990 to 2010" Nov. 1999 used the following: Overestimate, Underestimate, Unable to determine based on current information

4.13 References

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5 Estimation of Avoided Health Effects Associated With CWS For PM and Ozone

The monetary value of avoided health effects associated with various Canada-Wide Standards (CWS) for PM and ozone are estimated using the Air Quality Valuation Model (AQVM). The value of avoided health effects are referred to as “benefits” in the AQVM and the CWS cost-benefit analysis (CBA) documentation. The AQVM is a spreadsheet model that quantifies avoided cases of premature mortality and morbidity and the monetary value associated with those avoided health effects. This chapter will focus on the approach used in the AQVM to generate the quantitative health effect estimates. Issues associated with economic valuation will be addressed separately in Chapter 8.

AQVM uses concentration-response (C-R) functions derived from the epidemiological literature to link changes in air pollutant concentrations with changes in adverse health effects. The uncertainties underlying the published studies selected to derive the C-R functions for the various health endpoints contributes to the uncertainty in the estimation of avoided health effects.

5.1 Basis for the Mortality Risk Estimates in AQVM

The CWS health benefits analysis estimates the monetary value of avoided deaths associated with both PM and ozone reductions. The overall benefit estimation is dominated by the premature mortality endpoint because of the relatively higher valuation of a statistical life than for any other benefit endpoint and the comparatively steep C-R function. Thus, the selection of a mortality exposure-response function for the benefits estimation is extremely influential on the bottom line of the CBA.

Both time-series studies and prospective cross-sectional studies were selected in developing the mortality C-R functions for PM_{10} and $PM_{2.5}$. The Schwartz et al. (1996) time-series study of six U.S. cities (of summed daily mortality excess deaths attributable to PM_{10}) was used to develop the “low” value of the C-R parameter for PM_{10} and $PM_{2.5}$. The Pope et al. (1995) cross-sectional analysis of annual mortality rate differences attributable to $PM_{2.5}$ in 50 U.S. cities was used in developing the “high” value of the mortality C-R parameter estimate. The "central" value of the C-R parameter estimate of premature mortality was based on a two-thirds to one-third relative weighting of the C-R parameter from the time-series (Schwartz et al. 1996) and cross-sectional cohort studies (Pope et al. 1995), respectively. The C-R parameter for the time series study was 5.5 fold lower than the cohort study for $PM_{2.5}$ effects.

The Panel questions this approach and find that the weight of evidence is towards a different approach to estimating mortality impacts on several grounds. The excess in daily mortality, even when summed over the full year, does not reflect the total mortality impact of long-term cumulative exposure, and the extent of premature mortality cannot be determined from time-series analyses. Furthermore, some of the life-shortening associated with the daily time-series analyses is of the order of a few days. This raises difficulties for the subsequent valuation phase of the analysis. By contrast, prospective cohort analyses of annual mortality by Pope et al. (1995), Dockery et al. (1993) and Abbey et al. (1999) provide a basis for a fuller accounting, as well as for determining life-years lost. The Panel agrees with the following rationale cited in the U.S. EPA *Assessment of the Benefits and Costs of the Clear Air Act 1990 to 2010* (U.S. EPA, 1999) (herein called the "812 study") for using the Pope et al. study as the basis for developing the primary PM mortality estimates… "*Pope et al. studied the largest cohort (over 295,000 members of the American Cancer Society (ACS) cohort), had the broadest geographic scope (50 metropolitan areas), and effectively controlled for potentially significant sources of confounding. Even though Pope et al. (1995) reports a smaller premature mortality response to elevated PM than Dockery et al. (1993), the results of the Pope study are nonetheless consistent with those of the Dockery study*".

While the U.S. 812 study identifies the Pope study as the strongest of the PM cohort studies it notes the following limitations of the study: the selection of a largely white and middle class population may produce a downward bias in the PM mortality coefficient because short term studies indicate that the effects of PM tend to be significantly greater among groups of lower socio-economic status; the migration of cohort members across study cities is not considered, and PM was the only pollutant included in the study resulting in a possible overestimate of the PM risk because it may capture the mortality impacts of other pollutants correlated with premature mortality (e.g., ozone or other gaseous pollutants). An alternative estimate of premature adult mortality based on the Dockery et al. (1993) prospective study which covered 8,000 individuals in six U.S. cities was also presented as an alternative estimate in the U.S. 812 study. Using the Dockery study increased the health benefits estimate by $100 to $150 billion (in 2010) for the U.S.

The recently completed re-analyses of the six cities and ACS cohort studies of annual mortality rates sponsored by the Health Effects Institute (HEI, 2000) found that the original analyses used sound methodologies and highly quality-assured data sets, and that the results were not greatly different when alternate model assumptions within reasonable limits were used. Within each population, educational attainment was a significant predictor of mortality, with the effects being largely concentrated in those with no post-secondary school education. The fact that the ACS cohort, consisting of a largely upper middle class group of volunteers had a lower overall C-R parameter than the more randomly selected populations in the six cities study, is consistent with this subsequent finding on the importance of educational attainment on premature mortality risk.

The consistency of the results of the ACS and six cities cohort studies is well illustrated in the Figure below, which appeared in the 1997 EPA PM Staff Paper. The points representing the six cities in the Dockery et al. (1993) study appear to fit well within the range of responses seen in 600,000 people living in 151 U.S. communities with sulphate and TSP, also reported in the Pope et. al. (1995) study.

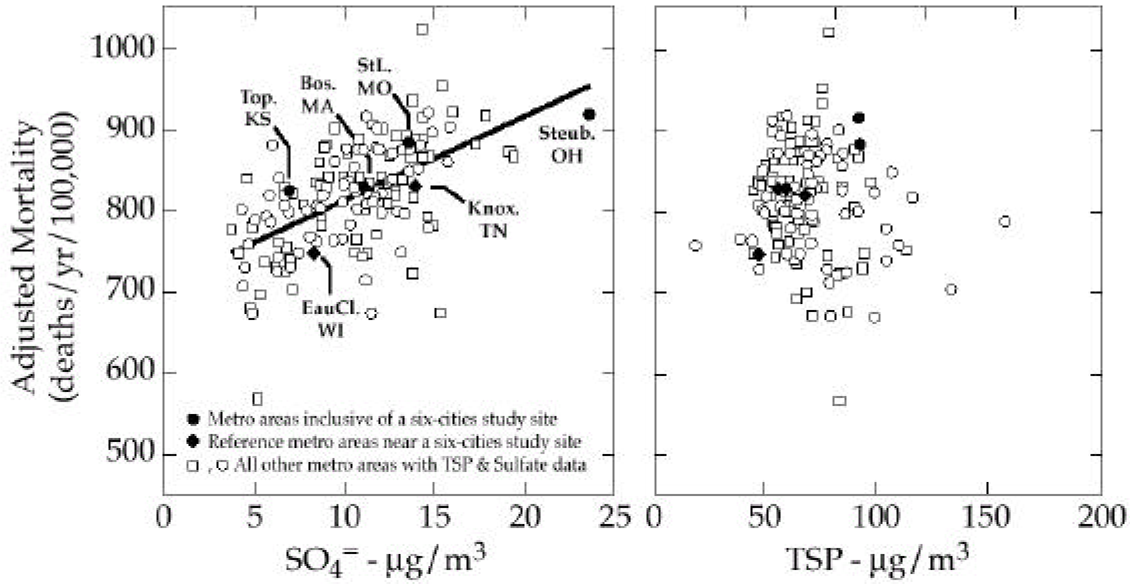

Figure 6. Age-sex-race adjusted mortality rates for $SO_4^=$ and TSP in about 600,000 adults living in 151 ACS communities having $SO_4^=$ data from Pope et al. (1995), as well as mortality rates in six other communities that were studied by Dockery et al. (1993). Adapted from Figure V-6 of PM Staff Paper (EPA, 1996).

Another recently completed HEI-sponsored study by Samet et al. (2000) examined daily mortality in 90 U.S. cities using standardized air quality and daily mortality data. They found that there was a regional variation in relative risk, with the relative risks per unit of PM_{10} being lower in western cities. Western U.S. cities also generally have somewhat lower ratios of $PM_{2.5}$ to PM_{10} and much lower ratios of $SO_4^=$ to PM_{10} than eastern cities. This may also help to explain the lower relative risks for the more broadly distributed ACS cohort in comparison to the six cities cohorts from the eastern part of the U.S.

The critical role of PM source control in the reduction of annual mortality rates has long been known. The first strong indirect evidence was included in the Royal Commission report following the Dec. 1952 London Smog (Ministry of Health, 1954). Annual mortality from bronchitis was far greater in the U.K. than in other Northern European countries with much lower ambient smoke levels. With the switch to smokeless fuels resulting from the U.K.'s Clean Air Act, there were dramatic reductions in annual mortality from chronic bronchitis and respiratory tract cancers. This is clearly illustrated in Table 22, taken from a paper by Chinn et al. (1981).

Table 22 Standardized Annual Mortality Rate Regression Coefficients on Smoke* for 64 UK County Boroughs (Chinn et al. 1981)

Sex	Ages	Mortality in years	Cancer of Trachea, Bronchus & Lung	Chronic Bronchitis
Males	45-64	1969-1973	0.07	0.02
		1958-1964	0.53++	0.32+
		1948-1954	0.71+++	0.48+++
	65-74	1969-1973	0.15	-0.06
		1958-1964	0.68++	0.31
		1948-1954	0.87+++	0.37+
Females	45-64	1969-1973	-0.02	-0.02
		1958-1964	-0.64++	0.33+
		1948-1954	0.49+	0.49++
	65-74	1969-1973	0.07	0.03
		1958-1964	0.25	0.40+
		1948-1954	0.61++	0.31

* Based on index of black smoke pollution 20 years before death of Daly (1959)
+ $p < 0.05$
++ $p < 0.01$
+++ $p < 0.001$

Thus, while the relative risk for annual mortality based on the ACS cohort may underestimate the extent of the risk for people living in eastern North America, and especially for people of lower socio-economic status, it provides, at present, the most reliable quantitative basis for estimating the risk factor of greatest economic impact of the overall cost-benefit bottom line.

More refined estimates of the relationships between PM and other classical (WHO-terminology) or criteria (EPA terminology) community air pollutants should be available when the next round of CWS are considered. The EPA-supported Harvard PM Health Effects Center is currently doing a follow-up study of mortality events in recent years on the six-cities cohort, and Thurston and colleagues at New York University, in collaboration with Pope at Brigham Young University, have begun to study recent years' mortality events among the ACS cohort. With the availability of the lessons learned in the HEI reanalysis study and the substantial numbers of recent years' mortality events available to the study teams, even more thorough and definitive analyses should be forthcoming by mid-decade.

Significant associations with daily mortality have been reported for ozone in some studies, albeit generally at much lower relative risks than for PM. In the US 812 study benefits associated with ozone reductions were estimated only in a sensitivity analysis, with a cautionary note about the uncertainties surrounding the potential ozone-mortality relationship. The Panel supports the CWS approach in carrying out separate estimates of mortality and morbidity benefits for both PM and ozone since the following steps were taken in AQVM to minimize the chance of overstating the ozone health benefits: i) all of the ozone health effects estimates were based on analyses that included a measure of PM in the models and ii) ozone mortality estimates were drawn from studies in many different locations across which the degree of colinearity between ozone and PM varies (Chestnut et al., 1999).

5.2 Basis for the Morbidity Risk Estimates in AQVM

The CWS CBA analysis estimates the monetary value of avoided health effects associated with PM and ozone reductions for the following non-fatal health endpoints: chronic bronchitis (CB) for $PM_{2.5}$ and PM_{10}; respiratory hospital admissions (RHAs) for $PM_{2.5}$, PM_{10} and ozone; cardiac hospital admissions (CHAs) for $PM_{2.5}$ and PM_{10}; net emergency room visits (ERVs) for $PM_{2.5}$, PM_{10} and ozone; asthma symptom days (ASDs) for $PM_{2.5}$, PM_{10} and ozone; restricted activity days (RAD) for $PM_{2.5}$ and $PM_{10;}$ minor restricted activity days (MRADs) for ozone; net days with acute respiratory symptoms for $PM_{2.5}$, PM_{10} and ozone; and children with acute bronchitis (B) annual risk factors for $PM_{2.5}$ and PM_{10}. The Panel views the selection of health endpoints as comprehensive given the current epidemiological literature.

There are clearly health effects occurring in children, such as the short-term changes in pulmonary function following ambient ozone exposures (Spektor et al., 1988), and reductions in the rate of respiratory symptoms and lung growth associated with long-term average exposures to fine particles (Dockery et al., 1996; Gauderman et al., 2000). There has also been a report of an association between PM and postneonatal infant mortality (Woodruff et al., 1997). However, there is questionable clinical significance associated with small changes in symptom rates and pulmonary function. Furthermore there are no established valuations in the economic literature for these effects or for neonatal mortality. Thus, they have essentially no impact on the overall valuation of health damages from ambient exposures to PM or ozone.

Epidemiological studies examining the health effects associated with particulate matter have used various measures of PM. Some have used PM_{10} while others have used $PM_{2.5}$. The number of studies using $PM_{2.5}$ as the indicator of PM is more limited than the number using PM_{10} because of the relative sparseness of $PM_{2.5}$ monitoring data. A number of studies have used total suspended particulate matter (TSP), British Smoke (BS), coefficient of haze (CoH) and other measures of particulate matter. Quantitative estimates of the relationship between between PM_{10} and $PM_{2.5}$ and respiratory hospital admissions (RHAs) are developed from the Burnett et al. (1995) study of daily admissions for respiratory illnesses and daily particulate sulphate levels in ambient air from 1983 and 1988 in Ontario. The model controlled for ozone and temperature because of modest correlations with sulfates. The results were used to develop the “central” and “low” C-R parameter estimate in AQVM. The sulphate based result was converted to its PM_{10} equivalent assuming a ratio of sulphate to PM_{10} of 0.18 in Ontario.[21] Applying a constant ratio of sulphate to PM_{10} or $PM_{2.5}$ across all provinces is not justified based on actual measurements of ambient concentrations. Nationally, composite average sulphate concentrations observed at sites east of Winnipeg are 2.3 times higher than those observed at western sites (Dann, 1994 in CEPA WGAQOG 1999). Sulphate contributions of 65% of the fine particle mass have been observed in southwestern Ontario in the summer (Keeler et al., 1990 in CEPA WGAQOG, 1999). Using the regional measures of sulphate to $PM_{2.5}$ ratios would provide a more reliable basis for sulphate to PM_{10} conversions in each province.

5.2.1 Gaps and Uncertainties in the Health Effects Analysis

Despite the relative wealth of epidemiological data on PM and health, there are aspects of the problem that are still not understood. The following uncertainties affect the interpretation of the available evidence for health effects, and limit, to varying degrees its use in policy making (HEI, 1999).

Causality Assumption

A critical assumption in the estimation of health benefits from PM and ozone reductions is that the correlations between increased air pollution exposures and adverse health outcomes found in epidemiological studies indicate a causal relationship between the pollutant exposures and the adverse health effects.

Factors supporting the likelihood of a causal connection for PM include:

1. The coherence of the associations *(Is the effect seen in a variety of related endpoints as could be expected?*). As noted by Bates (1992), if people are dying in excess in association with ambient PM, then one would expect that less serious effects would also be occurring in the same time frame, and with somewhat greater relative risks. Such coherence has been demonstrated with data on hospital admissions for respiratory and cardiovascular causes, emergency department visits, lost time, etc. (U.S. EPA, 1996).
2. There are small, yet broadly consistent relative risks for excess annual and daily mortality and daily hospital admissions for cardiopulmonary categories associated both with fine particulate matter ($PM_{2.5}$) and coarse particulate matter (PM_{10}). However, these findings cannot be used to establish underlying biological mechanisms that may account for such associations. Nonetheless, the same kinds of epidemiological associations, also lacking mechanistic understanding, have been seen for many of the same response endpoints for other complex mixtures, i.e., cigarette smoke, for both mainstream and environmental tobacco smoke (ETS). A widely held consensus that a causal relation exists for mainstream smoking has become nearly universal among the scientific community and general public in recent decades as the weight of the (largely indirect) evidence has grown. This process of accumulating evidence in favour of a broad acceptance for causality for a range of cardiopulmonary effects is at an earlier stage for ETS and even earlier for community air pollution, but

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[21]The AQVM Methodology Final Report (p. 5-27) notes that average levels of sulphate particles in southern Ontario are about 18% of average PM_{10} levels in the same area according to Dann (1994).

appears destined at this time toward a more widespread acceptance as a prudent public health judgment.

3. The consistency of the associations, in terms of finding significant relative risks for PM among metropolitan regions in the humid eastern regions as well as the drier western regions of North America, and in South America, Europe, Australia, and parts of Asia, including regions with quite different ratios of PM to pollutant gas concentrations.
4. The inability of all of the hypothesized confounding factors, including weather systems, to account for the associations between ambient PM and health effects.
5. Newer data from both observational cohort studies and chamber studies using concentrated ambient particulates showing that ambient levels of particulate pollution adversely influences cardiac variability in humans and cardiac ischemia in dogs. There is also a small amount of evidence suggesting that blood may be more likely to clot on high air pollution days. Serum fibrinogen levels were reported to be higher among office workers on days of higher air pollution compared to days of lower levels (Pekkanen et al. 2000). Fibrinogen levels may also be adversely affected by exposure to concentrated ambient $PM_{2.5}$ in a chamber (Petrovic et al. 2000).

Weakening the argument for causality are:

1. A low relative risk. RRs for mortality are generally below 1.2 and sometimes as low as 1.02 with risks being expressed by a relatively small number of people with an unusual degree of susceptibility associated with the extremes of age, pre-existing disease, and/or greater than average exposures.
2. Lack of supporting evidence from experimental human clinical studies. With the exception of heart-rate variability, controlled studies involving relatively brief exposures of healthy individuals to single agents or binary mixtures have yielded few measurable responses even when the exposures are at concentrations far higher than the ambient concentrations associated with measurable responses in populations.
3. Absence of a plausible and empirically biological mechanism for toxicity. It is not clear how exposure to low levels of PM might produce the health effects observed in

epidemiological studies and whether certain attributes of PM may be more closely associated with these effects. This is perhaps the single most important missing piece of evidence to support a causal association.

One postulated mechanism is that exposure-mediated release of cytokines and chemokines recruit and activate inflammatory cells with untoward effects such as activation of coagulation. This hypothesis is in part supported by the study of Kennedy et al. (1998) where particulate copper triggered release of cytokines. Iron and vanadium and acidity have also been implicated in mediator release. One problem with this hypothesis is the finding that $PM_{2.5}$ consistently affects health despite differences in elemental composition. Not all particulates are capable of activating the immune response. Several investigators have observed reduced heart-rate variability in human subjects on days of higher particulate pollution. Reduced heart-rate variability correlates with sudden cardiac death. Godleski et al. (2000). However, similar findings could not be replicated in rats exposed to concentrated New York particulates (Gordon et al. 2000). In contrast, animal and human models have demonstrated that ozone increases IL-8 levels, neutrophils, inflammation, and increases the sensitivity to aeroallergens.

Although there is substantial literature showing that a decrease in annual average of exposure to PM improves population health status, there have been no studies assessing whether these benefits occur by decreasing average ambient concentration levels, or by reducing the daily or weekly variation of peak levels, or by affecting both. If $PM_{2.5}$ or ozone affect health through the release from cells of chemokines and cytokines, there is a vast amount of literature showing that cells can often adapt to a constant inflammatory stimulus. Below concentrations considered toxic, it seems that the peaks in exposure are more important for cells than the average concentration. When constantly exposed to the same concentration of stimuli or mediators, cells can either become desensitized by down-regulation of physiological receptor(s) (tolerance), or rendered more resistant by induction in increased amounts of detoxifying enzymes and anti-oxidant factors (adaptation). In addition, inflammatory cells may be induced to commit suicide (apoptosis) and would then disappear from sites of inflammation. There might potentially be no significant health benefits by decreasing the annual average concentration of air pollutants, if the daily or weekly variations in the concentrations of these pollutants are unchanged, maintaining significant endogenous inflammatory substances in primary target tissues (such as the lungs and heart).

For example, if we decrease exposure to a substance from a mean ambient concentration of 100 units to a mean of 50 units, there may be no health benefits if the range of daily variation remains the same (70 to 130 versus 20 to 80 both equal daily net variations in peak concentration of 60 units in both cases). With these assumptions in mind, measures aimed at decreasing PM and ozone exposure should also affect the daily or weekly variation in the concentrations of these pollutants.

4. Difficulty in determining which of the many types of particles is responsible for the associated adverse effects and the role played by other gaseous pollutants**.** The physical and chemical characteristics of PM are complex, reflecting the diversity of emission sources and fact that particles are continually evolving as they interact with other components of the atmosphere. PM may include solid or liquid compounds, including organic aerosols, sulfates, nitrates, metals, elemental carbon, and other material. Particulate air pollution is always present as part of a mixture of air pollutants, and PM levels are often highly correlated in time and space with levels of gaseous pollutants such as ozone, SO_2 and NO_2. It is therefore difficult to apportion causality to any one or to any particular binary mixture of the components. Fine PM (less than 2.5 microns in diameter) is generally viewed as having a more harmful impact than coarse PM (greater than 2.5 microns in diameter) however it is not clear whether the toxicity is related to the particle itself or to its chemistry. $PM_{2.5}$ may be merely a surrogate of the true exposure of interest. For example, Burnett et al. (1998) reported that CO and TSP accounted for the majority of daily mortality in Toronto Canada, whereas, the same authors reported that NO_2 had the largest single effects on mortality in 11 Canadian cities (Burnett, 1998). Kennedy et al. (1998) reported that the copper component of total suspended particulates caused a cytokine release from epithelial cells similar to the nature of release from the particles themselves. There is

heterogeneity of particle compositions between geographic regions and over time within a geographic region (Godleski, 2000). If metal content was critical to toxicity then heterogeneity of results would be expected between different geographic areas. Exposure to particles of similar concentration may have differing health effects, depending on their concentration, and therefore different magnitudes of improvement in health would be expected. However, studies across North America and Western Europe have found similar effects despite differing climates and sources of particles, suggesting that particle composition may not be critically important or that the toxic nature of the particle is unknown and stable.

Public Health Significance of Health Improvements

There is uncertainty about how much the putative *harvesting effect* (i.e. air pollution exposures advancing death by only a few days or weeks, as measured in daily mortality studies) minimizes the impact of mortality statistics. The majority of population studies have been daily time-series designs where day-to-day changes in air pollution are correlated with day-to-day changes in morbidity/mortality. These studies do not address the effect of continued exposure and cumulative health effects over the longer periods of time, and therefore may underestimate the long-term health impact. Time-series analyses do not easily allow determination of how many years of lost life is represented by a death. Would the person have died in the next few days or weeks anyway? Brunekreef (1997) estimated the reduction in life expectancy associated with the risk estimates of the Pope et al. (1995) and Dockery et al. (1993) cohort studies, using life table methods. The results show that long term exposure to air pollution can lead to a loss of several years of life. Research is currently underway to determine whether the association between daily mortality and PM is the result of a harvesting effect.

Non-threshold dose-response assumption

An important uncertainty in all of the particulate matter and ozone health effect estimates is whether there is a threshold level of air pollution below which further improvements in adverse health effects no longer occur with diminishing exposure, or whether the slope of the concentration-response function becomes significantly more gradual at lower concentrations (Chestnut et al. 1999, p. 4-5). AQVM 3.0 is designed with a default assumption that there is no threshold and a constant slope coefficient for PM_{10} health effects and also for ozone health effects during the ozone season (May-Sept).

The PM Science Assessment Document (PM SAD) notes that *"current epidemiologic data do not indicate an ambient concentration of PM below which no effects are found - a so-called threshold. Whether this reflects a linear exposure-response relation, or simply the limitations of epidemiological methods, is unclear. Lack of an observed threshold, with responses increasing monotonically from very low ambient concentrations up to much higher levels, was observed with remarkable consistency in many epidemiological studies on acute and chronic mortality and hospitalizations. There is little evidence, however for a dose-response relationship in the experimental literature. Even at high particle concentrations, acidic aerosols have been found to produce only small decrements in lung function in susceptible subpopulations* (CEPA/WGAQOG, 1999)

The Ozone Science Assessment Document (Ozone SAD) provided some evidence of a threshold concentration for respiratory hospitalizations based on an analysis of daily one hour maximum ozone levels and mortality and respiratory hospital data from 13 Canadian cities over an 11 year period (Burnett, 1998). A positive risk was observed for ozone concentrations above 20 ppb (0.31%) and a negative risk for ozone values below 15 ppb (-0.29%). No such evidence for a threshold was observed with the mortality data. Thresholds below which no measurable health effects occur are observed in individual subjects. Some argue that the threshold concept does not likely hold at the population level since there is a large range in susceptibilities from totally resistant healthy subjects to exquisitely sensitive subjects who are already ill and already very close to making the decision to seek emergency care.

The U.S. 812 study identified the possible existence of an effect threshold – or safe level of air pollution – as an important uncertainty that would impact both the estimates of specific health effects and ultimately on monetary benefits. In the absence of a scientific basis for selecting a particular threshold, the analysis assumed there are no effective thresholds and that air pollution has effects down to zero ambient levels. The potential impact of a range of possible alternative threshold assumptions for PM-related premature mortality was explored as a key sensitivity analysis using projected 2010 PM levels and the Pope et al. (1995) study. If the true mortality C-R relationship has a threshold, then Pope et al.'s slope coefficient would likely have been underestimated for that portion of the C-R relationship above the threshold, leading to an underestimate of the incidences of avoided cases above any assumed threshold. The effect of a range of possible effect thresholds for $PM_{2.5}$ (from 0 to 45 µg/m^3) on avoided mortality is illustrated in the 812 study. For example a zero threshold resulted in 20,000 avoided deaths (in 2010), a threshold of 20 µg/m^3 resulted in 7,000 deaths (approx.) nationwide (based on Pope (1995). The UK Economic Appraisal of the Health Effects of Air Pollution did calculations with and without an assumed threshold of 50 ppb for the effect of ozone of respiratory hospital admissions and deaths brought forward. The report notes that this had a significant effect on the results and suggests that the presence and absence of a 50 ppb threshold also be used in sensitivity analyses of other work quantifying benefits.

The AQVM 3.0 allows the user to conduct sensitivity testing by selecting alternative threshold levels for (1) long-term exposure risks for PM (mortality risks, chronic bronchitis, and acute bronchitis in children): and (2) short-term exposure risks (all other morbidity risks for PM), and all ozone mortality and morbidity risks. However, in the CWS health benefits analysis, the impact of alternative threshold level assumptions was not presented.

Misclassification of personal exposure to ambient particles and ozone

The majority of the epidemiological data considered in AQVM are ecological in design, that is, results are based on whole populations, not on individuals, and the level of exposure of the individuals to different ambient air pollutants is not directly measured. The concentration of PM and ozone measured at a single fixed ambient monitoring site is typically used as a surrogate for personal/population exposure within a given area, without knowing directly what degree of contact or intake of pollutants occurs at the individual level. The measurement error that may result can produce inaccurate estimates of the health effects associated with air pollution. The potential bias from exposure misclassification is a serious concern (CEPA/WGAQOG, 1999a, p. 14-36). Personal exposure studies done in Canada (Brauer and Brook, 1995, Liu et al., 1995) and the U.S. (Liu et al., 1993) have shown that ozone data from personal exposure monitors were significantly correlated with the data from the fixed ambient monitors. The central monitors are considered quite representative of the ambient ozone exposure of the population served by each monitor (CEPA/WGAQOG, 1999a, p. 14-37).

The assumption that ambient exposure data are an adequate surrogate for personal exposure to PM has not yet been validated. The findings of Abbey et al. (1993) utilized in AQVM for the chronic bronchitis risk estimate for PM_{10}, relied on exposure ambient concentrations adjusted for time spent indoors using questionnaire data and monthly adjustment factors. Correlations over time between personal measurements and central monitor values are stronger for $PM_{2.5}$ than for PM_{10}. Research on the effect of measurement error suggests that under most conditions it will result in underestimates of the actual effects associated with air pollution, though complex correlations between the measurement errors for multiple pollutants may produce errors in either direction. The PM SAD (CEPA/WGAQOG, 1999b) notes that *misclassification of personal exposure is of concern, although not a serious obstacle in studies of air pollution and health. Fine particles <1 to 2.5 um are fairly uniform across an urban area and have a slower rate of deposition that leads to more homogeneity, they also penetrate indoors more readily than coarse particles. Consequently, associations between fixed monitoring measurements and health outcomes on a population basis may be reflecting a fine particle effect. Therefore, on a population basis, the adverse health effects are associated with concentrations measured at the central site ambient monitors. Personal exposure is not misclassified; the personal exposure data is lacking though error in exposure estimates generally leads to an underestimation both of risks and of their statistical significance.*

Effects of particle composition on exposure measurement

The relative potency of nitrate vis-a-vis other PM components is not known. Complicating the issue are two kinds of exposure characterization uncertainties. One is that ammonium nitrate is semi-volatile and is partially lost from the filter before it can weighed (especially in hot summertime weather). Another uncertainty is displacement of nitrate from the filter as nitric acid when strong acid sulfates are also collected. Some of the nitric acid ends up as coarse particle nitrates on PM_{10} filters following neutralization on the surface of basic coarse dust particles. For now, it is reasonable to consider $PM_{2.5}$ and PM_{10} nitrates as no more or less toxic than other components.

Location of studies (regional differences)

Each C-R relationship derived in AQVM from studies conducted in various locations (typically in the United States and Southern Ontario) is applied throughout Canada to estimate health benefit estimates associated with avoided events. To the extent that pollutant/health effect relationships are region-specific, applying a location specific C-R function throughout Canada may result in overestimates of health effect changes in some locations and underestimates in other locations.

Exposure-mortality lags

It is not known whether there is a time lag – a delay between changes in PM exposures and changes in annual mortality rates – in the chronic PM /mortality relationship. The Health Effects Institute re-analysis of the Harvard Six Cities Study demonstrated that exposure-mortality lag is difficult to determine, largely because the temporal trends in exposure are so closely related in the six cities studied (HEI, 2000 pp. 146-147). The U.S. 812 study assumed a five-year lag structure, with 25% of deaths occurring in the first year, and another 25% in the second year, and 16.7 % in each of the remaining three years. If the lag period is underestimated the benefits will be overestimated and vice-versa. An exposure-mortality lag structure is not discussed in the AQVM methodology document or the CWS benefits compendium document. The 812 study presented a reasonable approach to addressing exposure-mortality lags. Future studies in Canada should also address this issue and its uncertainties.

The U.S. National Research Council Committee on Research Priorities for Airborne Particulate Matter has identified 10 priority research areas to inform policy decisions on PM (NRC, 1998, 1999, 2001). It is anticipated that substantial new information on biologically important components of PM, toxicological mechanisms and the relationship between personal exposures and ambient concentrations should be available for the EPA review of PM standards in 2002.

5.3 Conclusions

1. The CWS gave greater weight (2/3) to mortality derived from daily time-series data than to the mortality impact derived from cohort studies of annual mortality (1/3). The annual mortality data should be used as the primary basis for determining the mortality impact because they include not only the impacts of peak daily exposures, but also the cumulative effects attributable to baseline exposures over other time scales. The Pope et al. (1995) cohort study provides the firmest C-R parameter for the annual mortality impact because of the size of the cohort and the large number of North American communities. However, the C-R parameter from this study of largely middle class volunteers very likely is an underestimate when applied to the overall population. The HEI (2000) reanalysis of this study demonstrated that, within this cohort, the effect was larger for those with lesser educational attainment. Thus, it is reasonable to conclude that a more representative population than used in the Pope study would provide larger C-R parameters.
2. The accumulating evidence towards a broad acceptance of causality for a range of cardiopulmonary effects from fine particulates appears destined towards widespread acceptance as a prudent public health judgment.
3. The evidence for mortality causality is more convincing for finer particulate (i.e. $PM_{2.5}$) than for coarser particulates.
4. The CWS health benefits analysis has taken adequate steps to avoid overstating the ozone health benefits due to colinearity with PM.

5. The database for fine particulate matter across the country is limited and more air quality monitoring data focused on fine particulate would provide a better basis for adjusting future air quality standards.

5.4 Recommendations

1. The C-R functions for determining annual mortality risks and benefits associated with reductions in PM_{10} and $PM_{2.5}$ in AQVM should be based on the prospective cohort analyses by Pope et al. (1995), Dockery et al. (1993) and Abbey et al. (1999). The central C-R parameter should be taken from Pope et al. (1995), the low from the Abbey et al. (1999) study and the high from Dockery et al. (1993).
2. The mortality benefits estimation should be more heavily weighted towards C-R relationships assessed for $PM_{2.5}$ rather than PM_{10}.
3. There are challenges in providing realistic exposure conditions for human toxicology experiments that will satisfy research ethics review boards. The most useful experiments are likely to be achieved with concentrated ambient particulates and mixtures with other ambient pollutants to explore cardiopulmonary endpoints. These studies should be complemented with more field studies including individuals at greater risk who could not participate, ethically, in exposure chamber studies.

5.5 Summary

Table 23 provides a summary of the CWS approach to estimating avoided health effects associated with PM and ozone reductions and the Panel's assessment of the key limitations, uncertainties and recommendations for alternative approaches.

Table 23: Summary of Panel's Assessment of CWS Approach to Estimation of Avoided Health Impacts

ISSUE	**ESTIMATION OF AVOIDED HUMAN HEALTH EFFECTS**
CWS APPROACH	AQVM is used to compute number of avoided health events using C-R functions drawn from the epidemiological literature (see Tables 4, 5 and 6) using a weight of evidence approach. To reflect uncertainties in the literature, low, central and high estimates are selected based on likely ranges and are assigned a probability weighting. Health endpoints for PM include: annual mortality, chronic bronchitis, respiratory hospital admissions, cardiac hospital admissions, emergency room visits, asthma symptom days, restricted activity days, acute respiratory symptom, child acute bronchitis Health endpoints for ozone include: daily mortality risk, respiratory hospital admissions, emergency room visits, asthma symptom days, minor restricted activity days and acute respiratory symptoms. The Schwartz et al. (1996) time series study of daily mortality in 6 U.S. cities is used to develop the low C-R parameter for PM_{10} and $PM_{2.5}$. The Pope et al. (1995) prospective cross-sectional study of annual mortality rates is used for the high C-R parameter estimate. The central C-R parameter estimate is based on a two-thirds to one-third relative weighting of the Schwartz study (low parameter) and Pope et al. study (high parameter), respectively.
PANEL CRITIQUE Key Limitations	CWS gave greater weight (2/3) to mortality derived from daily time series data than to the mortality impact derived from cohort studies of annual mortality (1/3). The Pope et al. (1995) cohort study provides the firmest C-R parameter for the annual mortality impact because of the size of the cohort and the large number of North American communities. Annual mortality data should be used as the primary basis for determining the mortality impact because they include impact of peak daily exposures and cumulative effects attributable to baseline exposures over other time scales.
RELATIVE UNCERTAINTIES (Probably Minor, Potentially Major)[22]	Potentially major for estimation of reduction in mortality associated with PM and ozone reductions. Probably minor for other health endpoints.

[22] Likely Significance Relative to Key Uncertainties on Net Benefits Estimate: Probably minor (alternative assumption or approach could influence overall estimate by <20% difference), Potentially major (>20% difference). Adapted from US EPA study The Benefits and Costs of the Clean Air Act 1990 to 2010" Nov. 1990 study in which 5% difference was used see pg. 21, 33, 65, 79, 98.

Table 26: (cont'd)

DIRECTION OF BIAS	The effects of air pollution on health are likely underestimated because of random) errors in the accuracy of measuring exposure and outcome, and the use of daily time-series analyses which only captures acute effects. Further, the HEI reanalysis notes that C-R parameter from the Pope et al. cohort study of largely middle class volunteers is very likely an underestimate when applied to the overall population as the effect was larger for those with lesser educational attainment.
RECOMMENDATIONS /ALTERNATIVE INPUTS, TOOLS, APPROACHES	For PM_{10} and $PM_{2.5}$ the central concentration response parameter should be based on the Pope et. al. (1995) study, the low from the Abbey et al. (1999) study and the high from the Dockery et al. (1993) study. The mortality benefits estimation should be more heavily weighted towards exposure-response relationships assessed for $PM_{2.5}$ rather than PM_{10}. More human chamber studies using realistic exposure conditions to explore cardiopulmonary response. These studies should be complemented with more field studies including individuals with greater susceptibility to health effects who could not participate, ethically in exposure chamber studies.

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6 Non-Health Impacts

This chapter discusses the underlying assumptions, interpretations (stated and unstated), and uncertainties (statistical and model), associated with the estimation of non-health environmental endpoints in the CWS CBA. Improvements to existing models, alternative assumptions, and other possible approaches are suggested. It is important to note that although AQVM is designed to assess a variety of non-health endpoints, including visibility, agricultural damages from ozone, recreational fishing and acid precipitation, global climate change and greenhouse gases, only 'materials soiling' was assessed for the CWS CBA. The uncertainties discussed in this chapter relate to the broad range of non-health endpoints that are within the scope of the AQVM.

The reasons for limiting the selection of non-health endpoints for the CWS CBA were varied. Visibility was not assessed because there is currently no Canadian model for linking changes in emissions of ozone precursors and PM to changes in visibility. There is such a model for the U.S., but AES (Environment Canada) at the time felt that this model was inappropriate for use in Canada. If a model relevant to Canada emerges, it would not be difficult to incorporate benefits of improved visibility into the CWS review. Agricultural damages from ozone are not included in the CWS because resources (both time and money) were insufficient to calculate benefits. As well, the expected benefits of ozone reduction to agriculture production were assumed to be very small, overall, relative to the health benefits. Recreational fishing was not included in the CWS CBA because it is an endpoint that is sensitive to acidic deposition, and AQVM 3.0 does not model long-distance transport of precursor emissions to the site of deposition.

6.1 Visibility Damages

Visibility, or how far one can see, has aesthetic connotations and has also been shown to have a value to individuals. However, it is also related to air quality in the sense that aerosols that primarily restrict visibility are also a potential health hazard. Visibility is the ability to distinguish features in the distance and as such is based on the human ability to resolve low levels of contrasting brightness and colour in the distance. Visibility is not considered an air quality parameter under Canadian law and there does not appear to have been any Canadian studies conducted to develop the economic valuation of changes in visibility. Thus scientific and economic data from the U.S. have been used to develop a value for visibility in AQVM.

Normally, visibility values come under 4 categories of benefits, (1) residential active, (2) residential passive, (3) recreational active and (4) recreational passive. The active and passive values are those put on a direct experience and that for an option to experience in the future or even non-use value. Recreational usage refers to locations well outside urban centres such as national parks. Because of the lack of Canadian data, no attempt is made to value the impacts of visibility changes for recreational active and passive values. Also based on U.S. experience, residential passive values were estimated to be small and were not included in the model. Thus, of the four visibility values, only residential active was assessed. Based on U.S. studies the inclusion of only residential active values is likely to underestimate total visibility values by about a factor of two.

Much of the value data for residential active use have been assembled through application of the contingent value method (CVM) that involves the extensive use of survey data querying the respondents as to how much they would be willing to pay (WTP) for specified visibility changes. Clearly this is imprecise, but much effort has gone into designing the surveys to separate health and aesthetic values and eliminate biases in the responses. For example, in McClelland et al. (1991) WTP was about $300 per household (HH) per year (1996 Canadian dollars) for an improvement of about 14% in annual visual range (VR). Separating out the health effects dropped the cost to $54 for aesthetic value only, which fell further to $25 per HH when errors and extremes in responses to the survey were taken into account.

6.2 Materials Damage and Soiling for PM and SO_2

The economic loss in materials damage and soiling due to air pollution is associated with PM and SO_2. In the case of visibility, most studies are conducted in the U.S. and of course large climate differences might tend to skew the transfer of results to Canada. In AQVM, data from New York State have been used as a guide in developing estimates of the economic effects of materials damages and soiling. There is substantial variability in estimates of WTP and the studies are often confounded by the mixture of health aspects and aesthetic effects.

There are various types of materials damage caused by air pollution, including soiling of indoor and outside materials, erosion which can lead to safety issues because of possible underlying structural damage, blistering of paint, damage to fabrics and stone etc. Of course there are natural impacts, but particulate mass and SO_2, either as gas or oxidised to acid, appear to enhance natural processes. Chamber and field data have been used for other assessments but AQVM does not use these data as it has not been independently verified by other techniques. AQVM estimates of the economic effects due to materials damage include (1) the associated costs of more frequent cleaning in the household, (2) costs of maintenance due to PM and SO_2 and (3) the maintenance cost estimates for galvanized steel structures due to SO_2. Effects due to industrial soiling, stone building, and paint in non-household structures are not included due to lack of quantitative information.

6.2.1 PM_{10} Damages

All of the data accumulated has been for TSP and not PM_{10}, so AQVM assumes that the soiling damage is proportional to mass. This seems reasonable, but no evidence has been put forward to support such an assumption. Costs have been associated with soiling damage using the results of a study of household expenditures in 20 U.S. urban areas as a function of associated measurements of TSP and SO_2 (Manuel et al., 1982). Using this approach a statistical connection between costs and pollution levels is derived. The result, using U.S. data translated into 1996 Canadian dollars with inflation was $1.75/HH for a 1 $\mu g/m^3$ change in PM_{10}. This study did not include the value of time for do-it-yourselfers. A central value of $3.50 was selected based on McClelland et al. (1991) Two Cities study which obtained WTP estimates by household for changes in air quality in Chicago and Atlanta. An upper estimate of $8.75 (1996 Canadian dollars) was selected based on Watson and Jaksch's (1982) analysis of a 1970 survey of households in the Philadelphia area concerning the frequency of different household cleaning tasks.

6.2.2 SO_2 Damages – household and steel structures

The impacts of SO_2 are somewhat different depending on whether the surface is moist or dry. Estimates of household costs where chosen to be \$2.50 per HH (range, \$1.20 - \$3.80) (1996C) for each $\mu g/m^3$ change in SO_2. The impact on steel structures has not been empirically verified and should perhaps be treated as an upper bound. In some studies it has been assumed that reductions in air pollution will lead to concomitant reductions in maintenance. However, there is much anecdotal evidence to show that this is often not the case, as other issues drive maintenance practices. As noted above, results from a detailed New York State study were used as proxy data for Canada. In both cases the relationship between damage and change in SO_2 is linear.

6.3 Greenhouse Gases

This AQVM looks to the future and tries to assess the impact of future technological changes which is always extremely difficult. However, in 30-50 years, certainly with the temporal regime of this study, we may be in a modified climate regime induced by the warming impact of increased CO_2 and other GHGs. The impacts may be extensive, ranging from sea level rise, to increased dryness or rainfall (depending on how the relatively few degrees C in global temperatures manifests itself on the regional scale), to increased incidence of forest fires. Currently, the regional details of the climate models are not reliable, but they do offer a disturbing insight into the possibilities that may occur. Clearly the fact that the there *may* be more forest fires with increased emissions of particles in the fine fraction, warmer summer temperatures with more incidents of quasi-stationary highs leading to more ozone and smog episodes, and perhaps increased biogenic emission, should require that climate effects should become part of the cost-benefit analysis, *as soon as possible*. If the future regional climate scenario were as pessimistic as suggested above, then the health costs of PM, SO_2, ozone etc. would increase without alteration of emissions.

Another aspect that will eventually require addressing for specifying the baseline is the connection between GHG controls, such as may occur under the Kyoto Protocols, and concomitant changes in emissions that lead to air pollution. When fossil fuel burns efficiently in air the main products are water vapour and CO_2. However, many other products are produced in small (compared to CO_2) amounts but which are, in fact, the emissions that we have referred to above. Thus, if there are controls on CO_2, it may be that these will also lead to reductions in VOC and NO_x emissions associated with the burning process. However, we must remain vigilant that reductions in the main polluter (from a climate perspective, CO_2) do not occur at the expense of increases in the minor emissions. There are figures for the impact of CO_2 reductions, but to date there do not appear to be any figures that can be used to assess the associated possible reductions in VOCs and NO_x.

6.3.1 Agriculture Losses to Ozone

The omission of agricultural losses from the benefits estimate had two bases: i) limited resources to model the losses to Canada from the C-R relationships presented in AQVM; and; ii) the sense that these losses are so small relative to human health losses, that the omission of agriculture from the CWS CBA would not change the overall conclusions of the CBA. The losses indeed do seem small when the costs and benefits are aggregated over all members of the population, rather than distributed specifically to the population group that will feel the effect. Specifically, if the agricultural crop yield gains were distributed just among agricultural producers, rather than among all Canadians, then the benefit might be quite significant to agribusiness, and the sustainability of rural communities. This question of how to distribute the costs and benefits is certainly a complex one, as some of the benefits of increased agricultural yield may well be to the consumer of the crops, through lower prices arising from greater supply. Models which capture this complexity are not readily available. Further, the inclusion of agricultural benefits in the current CBA will be difficult, as the recommended best dose-response relationships for yield reduction are expressed in seasonal ozone doses, not one-hour maxima (WGAQOG, 1997). However preparatory for the future inclusion of agricultural crops in the CWS, the following assumptions of the AQVM, and their biases, should be considered.

i) One of the most important assumptions associated with the calculation of agricultural losses to ozone is the linearization of the C-R function between 30 and 50 ppb, despite certainty that the relationship is curvilinear. This has the result of overestimating the benefits of ozone reduction at lower concentrations, and underestimating the benefits of ozone reduction at higher concentrations. The AQVM methodology document suggests that this assumption is consistent with the available data, and that this assumption and the data will be improved in subsequent versions. However, the source of these improved data is unclear, as current data-gathering efforts for ozone and agricultural yield are not in progress, at least to the best of the knowledge of this Panel. If no new data are forthcoming (and even if they are in progress) it is not clear why one of the curvilinear functions (Weibull in the original NCLAN data analyses, gamma in some retrospective analyses), which are mathematically defined ($Y=\alpha\cdot \exp[-(\chi/\sigma)^{c}]$, $Y=\alpha(X + 1)^{\gamma} \cdot e^{-\beta x}$) cannot be used to calculate yield gains with incremental reductions in ozone concentration.

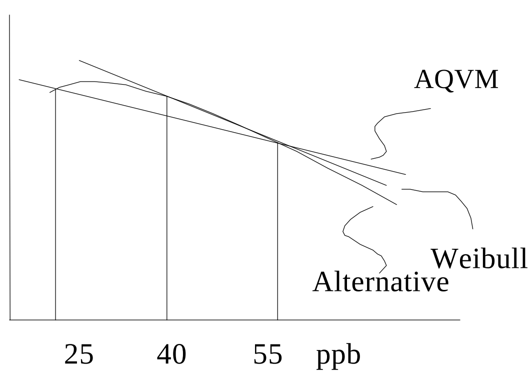

Figure 7: A representative Weibull dose response function, the linearized function used for the AQVM demonstrating the concentration dependent biases inherent in linearization, and the alternative linearization, using 0.040 ppm as the reference ozone concentration.

ii) An assumption that is related to the issue of curvilinearity between 0.025 and 0.050 ppm, is what the appropriate reference concentration is for estimating the improvement in crop production with reductions in ozone concentration. Both the OMOEE (1989) and AQVM efforts used 0.025 ppm, and many parts of Canada not influenced by transboundary air flow or anthropogenic activity routinely achieve 80% of 7-h seasonal means below 30 ppb, suggesting that the choice of

0.025 ppm as the reference concentration for ozone is not inappropriate for CWS. However, there is considerable scientific argument as to whether reference concentrations of ozone below 0.04 ppm are relevant, given biogenic production of ozone and the engineering and social limitations on emission reduction. If a decision was made to relate benefits to a reference concentration of 0.04 ppm, rather than 0.025 ppm, then the dose-response functions would have to be re-created. As a linear function, it would likely have a steeper slope as the flatter threshold part of the curvilinear response from 0.025 to 0.04 ppm would no longer be considered in determination of the 'line of best fit'. The benefit of this is that the concentration dependent bias in the estimation of benefits to crop production would be gone; the steeper line would result in increased benefits for an incremental decrease in ozone.

iii) The third key uncertainty in the modeling of agriculture losses to ozone is the omission of several crops that are important to regions of Canada where ozone concentrations are known to be high during the growing season: BC, the Québec-Windsor corridor, and the eastern Maritime region. The most notable omissions are potato, hay and canola, although canola is a more important crop in the Prairie provinces (low ozone regions) than in the high-ozone regions of the country. Hay was omitted because insufficient production/price data were available to calculate the benefits of yield improvements with reduction in ozone. Canola and potato were omitted because of insufficient dose-response data - these crops were not addressed in the NCLAN project. The concern with respect to these omissions is that hay and potato are more sensitive (OME, 1989) relative to the wheat and field corn that have been included in AQVM, and they are important crops in parts of Canada that experience high concentrations of ozone. Their omission may represent a significant underestimation of the benefits of incremental ozone reduction. Using a different modeling approach, OME (1989) estimated the yield losses to potatoes in Ontario alone, relative to the yield expected at 25 ppb, to be between 5.6% and 6.9%, and to hay in Ontario alone, to be approximately 4.4%. The ozone exposure characteristics associated with these estimates are most similar to the 40 and 50 ppb columns of Table 24 provided in the AQVM methodology document and are greater than the estimates of yield losses for corn reported. The estimates of the value of yield gain, for Ontario alone, for hay and potato ranged from a highs of $18.4M and $2.9M, respectively, to lows of $2.6M and $0.5M, respectively (OME, 1989). These estimates clearly miss the hay production of Quebec and the potato production of the maritime provinces, both of which are substantial.

Table 24. Estimate of Yield Losses for Six Crops from Various Ozone Levels (%)

CROP	MEAN OZONE CONCENTRATION - ppb			
	30	40	50	60
Corn	0	1.7	3.7	6.7
Soybean	3	5.5	10	15.3
Wheat	3	9	15	20.8
Hay (alfalfa)		5	8	11.5
Hay (other hay)		6.7	12.7	20
Tobacco			5	9

Note: Ozone is measured as June-September 9 a.m. – 9 p.m. hourly average. Yield losses are measured against an assumed background ozone level of 25 ppb.

Source: Heagle et al., 1988 in Chestnut, L.G., D. Mills & R.D. Rowe, 1999.

iv) There are a number of uncertainties surrounding cultural influences on crop yield improvement. One of the key factors not taken into consideration in these calculations is the substitution of a more tolerant crop for an ozone sensitive crop, resulting in benefits to agricultural crop production with no emission reduction. This certainly may happen, but there is considerable acreage in Ontario, certainly, and likely other parts of Canada that experience elevated ozone concentrations during the growing season and which will not support a wide variety of crops: pasture land tends to be pasture land because nothing else is economically supportable on it. For the owners of these kinds of land, substitution is not likely. Some potential for substitution lies in switching among cultivars, as there is considerable inter-specific variation in ozone sensitivity, but this is crop-species specific and may not be an option in most situations. The other aspect of substitution is that consumers of ozone sensitive crops could utilize an alternative crop, thus achieving the same cost saving as would result from emission reduction, and the lower cost for a commodity that would arise from production increase. The problem with this is that for potato and hay, for example, there really are no substitutes for their intended uses.

v) The benefits of ozone reduction are overestimated because the entire production of a crop, on a provincial basis, is used to calculate increased yield. However, almost certainly, parts of that province will see no improvement in air quality for ozone due to emission reduction, because the ozone concentrations in those areas are not influenced by emissions. The alternative is to map production of crops on a county basis, and then calculate the yield benefits to each crop relative to the improvement in air quality expected in that county with emission reduction. This may not be a very large source of overestimation of benefits as most of the agricultural production of the Quebec-Windsor corridor coincides with the zones of air quality that would be positively influenced by emission reduction. The exception to this would likely be the counties east of Lennox & Addington, in which agricultural production is quite important, but which have historically had a seasonal average mean ozone concentration of 30 ppb (OMOEE, 1989). By way of example, the counties of Dundas, Glengarry and Stormont, Frontenac, Grenville and Leeds, Lanark, Lennox & Addington, Ottawa-Carleton, Prescott and Russell accounted for 786 of the total of 5,868 ktonnes of field corn produced in Ontario in 1999 *(http://www.gov.on.ca/omafra/stats/crops)* - just slightly more than 13%. Since these counties would likely see little reduction in ozone concentration due to emission reduction but have had their yield included in the calculation of benefits, it is reasonable to suggest that the benefits for this crop have been overestimated by at least 13%, and likely more when the rest of Ontario's counties are similarly examined. So, the alternative to the current model is to use the statistics on the OMAFRA website to recalculate the yield improvements by county. The fact that most ozone monitors are in urban areas of the country may add imprecision to these estimates, also, as the mapping of air quality in regions of agricultural production is modelled, rather than extensively measured. So, there is a degree of speculation as to the actual air quality in Canada's agricultural regions; if the ozone concentrations are lower than predicted by the models, then the benefits to agriculture of emission reduction would be less than predicted. There really is no alternative to this, as the models are the best that can be done with the data that are currently being gathered.

vi) The expected changes in ozone concentration from emissions reduction are predicted on the basis of a 24h average, and the AQVM assumes that the changes in the 24h average are a good surrogate for changes in the 7h or 12h daily average. The issue of how to summarize ozone exposure for the purposes of predicting or preventing biological effects is a contentious one. Much effort in examination of various methods to quantify the amount of ozone that plants are exposed to has been expended over the years; without question, the best method is to measure the amount of ozone that actually is absorbed by the plants, as this will be most directly related to effects. However, that is not a practical approach from a regulatory perspective. So, attention has been turned to discussing whether there is a threshold concentration of ozone below which plants are not affected, so that none of that ozone exposure should be part of the exposure description. Another perspective is that elevated ozone concentrations in ambient air are random in their distribution, so that plants will have varying periods of recovery between exposures. Additionally, there are stages of growth and development that are more sensitive to ozone than others, and so the timing of these episodes is critical to their impact. Because most of these factors have not been extensively characterized relative to plant response to ozone, and likely will never be known in such detail that they can be incorporated into the regulatory process, plant exposure/response has been characterized using either a 7h or a 12h daily average concentration during the growing season. The 7h and 12h daily average concentrations will be higher than the 24h average, for most days of the plant growing season, and so a change in the latter must, mathematically, correspond to a larger change in the former. This means that the benefits to agriculture will be underestimated somewhat by predicting them on the basis of changes to the 24h average ozone concentration, as the real changes in the 7h or 12h average ozone concentration will be greater than those used in the yield response models. The degree to which this assumption likely underestimates the benefits to agricultural yield production could be estimated by regressing the 24h mean against the 7h and 12h means, for the months of May-September; the data required for this exercise are being collected as part of NAPS.

6.3.2 Agricultural Yield - Damages

The OME estimates of the dollar value of agricultural production increases, should ozone concentrations in Ontario be reduced to 25 ppb totaled $38M (mean), with a range of $14M to $61M (OME, 1989). The OME approach to estimating yield losses from ozone utilized the same data as the AQVM with the addition of crop yield data that had been generated in Ontario, so the database for the OME approach is larger than that for AQVM. One of the benefits of the OME approach is that it includes crops that are important to Ontario agriculture, such as potato and hay, which were not included in the NCLAN research project (the database from which the AQVM dose-response relationships were derived). The data comprising the OME base included unpublished government and university reports, and conference proceedings; the result was a database for crop response to 7-hour seasonal mean ozone concentrations of 40 and 50 ppb for 19 individual crops. For 12 of these crops, the database includes information that was not directly applicable to an Ontario context. So, a multi-component adjustment factor was applied to the plant yield responses for these 12 crops to compensate for geographic, agronomic and experimental variability among the gathered data. A total of 1000 points were available for allocation among the weighting factors. For example, in this weighting exercise, ozone exposure-plant response data gathered from field experiments in the SW United States had a 10/100 weighting in the amalgamated C-R function, whereas data gathered from field experiments in the SE, W or mid-W United States had a weight of 50/100 in the C-R function. Data gathered from Ontario or the NE United States were weighted at 100/100. Studies in which the experimental plants were irrigated were weighted at 1/100 whereas studies using non-irrigated plants were weighted at 100/100. Full weight (300/300) was awarded to a data set that included at least 120 yield loss data points; smaller data sets were weighted at some fraction of 300. Details of other adjustment factors can be found in the document itself. The C-R functions were linear or curvilinear, depending on what was an appropriate fit to the data; linearity was not forced. The adjusted dose-response functions were then compared to an analysis of the Ontario ozone database, which developed geographic distribution of regions corresponding to 50 and 40 ppb 7-h seasonal mean, based on monitoring data from 1974-1988. The yields for each of the crops under scrutiny were determined for each of the two regions, and then the predicted improvement in yield for the regions were calculated from the C-R relationships, assuming a roll-back of ozone concentrations to 25 ppb. The predicted increase in yield was then converted to $value of production increase by simply multiplying the increased tonnes by the average crop price from 1985-1987. There was no adjustment to the economic calculations for substitution, nor for adjustment in price/tonne because of increased supply. The strengths of the concentration-response functions derived from the OME vs. those derived for AQVM are:

- inclusion of crops important to regions of Canada where improvement in ozone would be expected to occur if CWS were achieved;
- allows curvilinearity of C-R functions where appropriate, thus avoiding the concentration dependent bias in the estimation of yield gains;
- calculates crop yield improvement relative to the amount of crop currently grown in a specific region, and the expected reduction in ozone concentration for that region;
- weighting of the data for relevance.

The only weakness of the C-R functions derived from the OME exercise relative to those derived for the AQVM is that the OME report is now 10 years old, and has not been updated. Presumably AQVM is more current. Having said that, the database was updated and re-examined for the Science Assessment Document for NAAQO for ozone (WGAQOG, 1997) as well, there have not been substantial research studies on crop-response to ozone in the last decade. Beyond that, both approaches have the same weaknesses, relating to the issues of scaling-up experimental results; so, there is no reason why it should not be either incorporated into the AQVM for the CWS CBA, or substituted for AQVM in CWS CBA.

6.4 Recreational Fishing and Acid Rain

There are a number of uncertainties in the recreational fishing and acid rain valuation, many of which arise from the economic models used to estimate lost opportunity for fishing. There are also some assumptions and many uncertainties in the scientific models that link changes in acid deposition with acidification in lakes, most of which revolve around the variation among lakes in the ability to absorb or buffer changes to pH.

In Alternative 3 for estimating change in consumer surplus per angler-day from a change in acid deposition, the AQVM model apparently does not take into account the fact that the pH scale is logarithmic. An increase in pH from a small reduction in sulphate input to fresh water lakes was interpolated from a two-point graph for data gathered in the Turkey Lakes Watershed of Central Ontario: an increase in pH from 5.0 to 5.8 when sulphate input was decreased from 27.8 kg/ha/y to 15.8 kg/ha/y, a reduction of 43%. The AQVM model assumes linearity between these two points, and calculates the expected increase in pH from a reduction in sulphate input of 1% as (5.8-5.0)/43, or 0.019 pH units. This calculation should instead be carried out by converting the pH units to their absolute quantities of H+ and working with the sulphate changes as real numbers, instead of as percentages. For example, reducing the sulphate deposition from 27.8 kg/ha/y to 15.8 kg/ha/y resulted in a reduction in H^+ concentration, in the lake water, from 1 x 10^{-5} M to 1.5 x 10^{-6} M, as indicated by the change in pH from 5.0 to 5.8. Removing 12 kg/ha/y sulphate from the deposition (a reduction of 43%) reduces the H^+ concentration by 8.5 x 10^{-6} M. So, a 1% decline in sulphate deposition from 27.8 kg/ha/y (a reduction of .278 kg/ha/y) would be expected to result in a reduction in H^+ concentration in the lake water of :

(.278/(27.8 – 15.8)) x 8.5 x 10-6 = 1.96 x 10^{-7} M H^+

Thus, the new H^+ concentration in the lake water when 0.278 kg/ha/y sulphate is removed from an input of 27.8 kg/ha/y is 1 x 10^{-5} – 1.96 x 10^{-7}, which equals a pH of 5.009, a small change from the 5.019, that AQVM would predict.

6.5 Omissions

The following non-health environmental endpoints were omitted from the CWS CBA:

Visibility: reduction in urban $PM_{2.5}$ will result in an improvement in visibility. Reduction of general fuel use is unlikely to improve visibility in wilderness areas since aerosols generated by forest fires and natural emissions likely will dominate this region. Even though the economic benefits associated with visibility improvements, will not be comparable in magnitude to those associated with mortality, the omission of visibility benefits will underestimate the benefits of $PM_{2.5}$ reduction.

Greenhouse Gas Changes: At present the drive to reduce fuel use as one means of reducing CO_2 emissions in order to meet Canada's Kyoto requirements may also lead to a reduction in PM and ozone. At this point it is not clear that the technology of reducing CO_2, the main carbonaceous product of burning, will actually lead to a reduction in PM, and NO_x. It is conceivable that in order to achieve a 10% reduction in CO_2 that a 20% increase in NO and PM could occur. At this point, the overall effect of the omission of greenhouse gas changes from the benefits assessment is uncertain.

Acid Rain: reduction in PM_{10} would be expected to reduce inputs of sulphate into ecosystems. It is unclear whether or not this reduction would have an impact on the pH of soils and surface waters, as the relationship between past sulphate reductions and such changes is not strong. It has been hypothesized that nitrate inputs, or stored nitrates in soils is buffering the predicted increase in pH resulting from reduced sulphate inputs. The influence of this omission on the estimation of the benefits of ozone reduction is unknown.

UV-B Radiation: penetration of UV-B radiation to the earth's surface might be expected to increase as a result of reductions in tropospheric ozone concentration. The impact of UV-B radiation on agricultural crops is likely the least important endpoint; health of aquatic organisms (including algae) and amphibians may be considerably more sensitive.

The influence of this omission on the estimation of the benefits of ozone reduction is unknown.

CO_2: reduction of CO_2 concentration in the atmosphere resulting from emission reductions could have an impact on plant productivity. However, there is scant evidence that CO_2 concentration limits productivity, at least in agricultural systems. It is likely that this omission does not influence the estimates of the benefits of ozone reduction.

Forestry: the effects of reduced ozone on forest productivity could be substantial, particularly on the west and east coasts of Canada. However, at this point, these losses/benefits are not quantified for Canada, although there are some US models of forest impact assessment. It is likely that this omission underestimates the benefits of ozone reduction, or has no effect on the estimate.

Unmanaged Ecosystems: wilderness areas have an emotional value that is quantified by the willingness to pay approach. The benefits to sustainability or diversity of wilderness areas from reduction in ozone have not been quantified; it is likely that this omission underestimates the benefits of ozone reduction, or has no effect on the estimate.

6.6 Summary for Non-Health Endpoints

Overall, the effect of limiting the selection of non-health endpoints for the CWS CBA can be summarized as:

- Omission of visibility: underestimates benefits of ozone and PM_{10} reductions
- Outdated model for materials soiling: unknown bias
- Omission of greenhouse gases: likely underestimates benefits of ozone and PM_{10} reductions
- Omission of agricultural yield: underestimates benefits of ozone
  - linearization of dose-response function: direction of bias depends on concentration
  - low reference concentration: underestimates benefits of ozone reduction
  - omission of key crops: underestimates benefits of ozone reduction
  - omission of cultural practices: unknown bias

- provincial scale of calculation: overestimates benefits of ozone reduction
- use of 24h average ozone concentration: underestimates benefits of ozone reduction

- Omission of recreational fishing: unknown bias
- Omission of acid rain: unknown bias
- Omission of UV-B: unknown bias
- Omission of CO_2: unknown bias
- Omission of forestry: underestimates or nil effect on the benefits of ozone and PM_{10} reduction
- Omission of unmanaged ecosystems: underestimates or nil effect on the benefits of ozone and PM_{10} reduction

6.7 Summary

Table 25 provides a summary of the CWS approach to estimating avoided non-health effects associated with PM and ozone emissions reductions and the Panel's assessment of the key limitations, uncertainties and recommendations for alternative approaches.

Table 25: Summary of Panel's Assessment of CWS Approach to Non-Health Effects Estimation

ISSUE	NON-HEALTH EFFECTS ESTIMATION
CWS APPROACH	Household materials soiling was only non-health endpoint considered. Other endpoints were considered to be minor relative to health.
PANEL CRITIQUE Key Limitations	Omits important endpoints relative to total of non-health endpoints such as. visibility, greenhouse gases, agricultural yield, forestry, unmanaged ecosystems.
RELATIVE UNCERTAINTIES (Probably Minor, Potentially Major)[23]	Potentially major from a distributional or sectoral standpoint. Ecosystem effects are highly uncertain but potentially major.
DIRECTION OF BIAS[24]	Underestimates benefits
RECOMMENDATION/ ALTERNATIVE INPUTS, TOOLS, APPROACHES	Include agricultural productivity at least[25] Use OME economic benefits, if AQVM cannot provide these numbers[26] Approach selection of non-health categories in a systematic fashion

[23] Likely Significance Relative to Key Uncertainties on Net Benefits Estimate: Probably minor (alternative assumption or approach could influence overall estimate by <20% difference), Potentially major (>20% difference). Adapted from US EPA study The Benefits and Costs of the Clean Air Act 1990 to 2010" Nov. 1990 study in which 5% difference was used see pg. 21, 33, 65, 79, 98.

[24] The U.S. EPA report "The Benefits and Costs of the Clean Air Act 1990 to 2010" Nov. 1999 used the following: Overestimate, Underestimate, Unable to determine based on current information.

[25] http://www.gov.on.ca/omafra/stats/crops

[26] Impact of Ozone Exposure on Vegetation in Ontario (1989) Ontario Ministry of the Environment ARB-179-89-PHYTO, ISBN 0-7729-6386-X

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7 Cost Analysis

This chapter examines the cost estimation component of the Canada-Wide Standards process. Our intention is to lay the conceptual foundation for cost estimation, to show that this foundation is the mirror image of the foundation on which benefit estimation lies, and to discuss some of the features of cost estimation that are often overlooked in cost-benefit studies. The estimation of cost is as difficult an undertaking as the estimation of benefits--something not often recognized. We then turn to an examination of the processes employed within the CWS approach.

7.1 Conceptual Overview of Cost Analysis

A number of conceptual issues arise in cost analysis, starting with a correct conceptualization of cost. For example, the most popular concept of regulatory cost in the analysis of environmental programs is abatement expenditures, i.e., out-of-pocket costs for abatement equipment. This is an exceedingly narrow measure and might have little to do with a better, but still imperfect measure--compliance cost, i.e., the cost of all the actions necessary to comply with a particular policy. For instance, a new environmental regulation might contribute to a change in how a product is made. This would not show up as an abatement expenditure but would be a compliance cost. Even here, although the cost of compliance can have a bearing on monetary measures of well-being, there is no simple conceptual link between the two. The correct perspective[27] is that compliance cost should be the total change in social welfare associated with compliance activities, not just the direct expenditures on the engineering measures required to achieve compliance. Complete accounting of compliance cost should include the cost of lost opportunities associated with compliance.

Some forms of regulation can be quite narrow in the range of responses they engender. These regulations tend to be tightly focused on target activities (e.g., selected industrial sectors) and do not "spill over" into sectors that are not direct targets of the regulation.

[27] Hazilla and Kopp (1990) conclude from their study of the costs of the Clean Air and Water Acts that monetary measures of change in well-being grow to exceed compliance costs over a ten year period.

However, depending on the nature of the activities to be regulated and the magnitude of the responses required, secondary effects of the policy can be felt beyond the direct target of the policy. When secondary effects are de minimis, they can be ignored in a cost study, and economic techniques of partial equilibrium analysis may be properly applied.[28] However, when secondary effects are thought to be large, a general-equilibrium analysis is called for (see Section 7.4)[29]. Likewise, there can be dynamic effects if policies alter the growth path of the economy, i.e. the future timing of investments and expenditures. Even in the case of economic incentives, as opposed to command and control policies, distortions can occur that can be costly.

7.2 CWS Approach to Estimating Costs

The CWS process employed a method of cost analysis developed by Stratus Consulting. This method is described in Chapter 3 of this report. Essentially, the approach employed a database of control technologies with cost estimates associated with each technology. An algorithm selects a set of technologies for each "policy" scenario projected (essentially a percentage reduction in aggregate emissions) based on the most cost effective control options.

The approach employed in the CWS process was an "engineering cost" or direct cost approach that does not consider behavioural or market responses to the change in regulation. As discussed above, the partial or general equilibrium effects of the regulation are ignored. There was no consideration of technical change and there was no consideration of intertemporal effects. Generally, this results in cost estimates that are biased upwards, relative to partial equilibrium analysis, all else held constant. General equilibrium analysis may produce cost estimates that are higher or lower than direct cost approaches, depending on the degree to which cost effects "ripple" throughout the economy. It should also be noted that direct cost estimates are *ex ante* estimates or assessments of the costs before the changes have actually

[28] A partial equilibrium analysis would focus on a narrow set of economic agents (producer and consumers) and would assume agents outside this set would be unaffected by the policy.
[29] A general equilibrium analysis makes no assumptions about affected parties and treats all agents in the economy as if they could be affected.

taken place. There is some evidence that *ex post* estimates of cost may be significantly lower than *ex ante* costs. U.S. EPA (1999) shows that *ex post* cost estimates can be 80% of *ex ante* cost estimates or less.

One issue that is receiving considerable attention in the academic and policy literature is the *Tax Interaction Effect* (See Parry & Oates (2000) for details on the tax interaction effect) . When cost analysis is performed, it is generally assumed that the economy is operating efficiently. However, it is well known that various taxes have distortionary effects that result in inefficient use of resources. Environmental policies or regulations may exacerbate this inefficiency and thus may induce even higher cost due to this effect. A common issue considered is the distortionary effect of income taxes. Income taxes tend to result in economic inefficiencies because the productive input (labour) is taxed and thus labour effort and investment is discouraged (at the margin). When environmental regulations further distort the signals for efficient use of labour resources, additional costs to society are experienced. Initial estimates of the magnitude of the tax interaction effect are substantial and suggest that social costs may exceed direct costs by 25% or more. If tax interaction effects are substantial, using direct costs as an estimate of social costs may not result in an underestimate of costs. The tax interaction effect discussed above is not included in the analysis developed for the CWS process. Ignoring this effect will bias the cost estimates downwards. However, additional evidence on this issue and the magnitude of the impact must be examined to fully assess the impact on Canadian cost studies.

7.3 Detailed Assessment of the CWS – Stratus Approach

7.3.1 Assumptions and Limitation of Analyses

The following assumptions used in the cost estimation analyses were documented in the Stratus Consulting Cost Study Methodology Report (Stratus 1999). The discussion below highlights some of the difficulties associated with these assumptions and their role within cost estimation.

Similarity of Canadian and U.S. Control Costs

It was assumed that Canadian industries will face similar control options to analogous industries in the United States.[30] The Stratus report for the CWS process employed a method of cost assessment using a U.S. database of technologies, costs and industrial structure (developed by E.H. Pechan and Associates). This allows for the significant database of information collected for U.S. Clean Air Act analysis to be employed to provide information for the Canadian situation. However, institutional differences between Canada and the U.S. are assumed to be "small" and all cost measures are simply converted to Canadian dollars using an adjusted exchange rate. It is not clear to the Panel that the institutional differences between Canada and the U.S. in terms of the cost of regulatory change are "small". There appears to be no analysis to determine the impact of such an assumption. In other sectors cost analysis has been notoriously difficult to "transfer" from country to country because of the different tax systems, different market structures between the two countries, different technologies and different initial regulatory systems. It would seem appropriate to examine this assumption carefully using test cases within Canada. Naturally, it would be best to augment the database with Canadian industrial information on technologies, costs and responses to regulatory changes. The use of a U.S. database also raises the issue of the appropriate exchange rate / purchasing power parity rate to use (as will be the case in the benefits section which also employs transfer of monetary estimates from the U.S.).

The conversion of 1990 U.S.$/ton to 1995 CDN$/tonne assumed a GDP deflator of 1.166029, and a 15% reduction in the relative cost of control technology inputs in Canada compared to the U.S. assumptions, which were not justified in the report. The 15% reduction seems especially arbitrary. It would be much more appropriate to apply true Canadian economic data instead of assuming this 15% factor.

The Panel is aware that Environment Canada has developed the Air Emissions Reduction Costing database (AERCo$t™) over the last six years to create a system that would estimate control system costs that would best represent the Canadian economic situation. We note that Stratus used AERCo$t results to estimate costs of controls for the transportation sector. The AERCo$t methodology is that of the U.S. EPA Office of Air Quality Planning and Standards (OAQPS) which is also used as the basis for the cost estimates prepared in the U.S. database used by Stratus. The AERCo$t database starts with actual flow rates associated with a particular process. If the inventory does not include a flow rate for that source, it is estimated by various methods. From the flow rate, the size of the control system required can be determined, and therefore, the equipment cost can be estimated more accurately. Typically, the basic purchase cost of the control equipment will probably be similar in Canada as in the U.S. However there are a number of additional costs included in the Total Capital Costs and Total Operation and Maintenance Cost associated with the installation and use of control systems. These additional costs, such as sales taxes, freight and delivery charges, construction and field expenses, performance tests, contractor fees, fuel, electricity, water and chemical costs, labour rates, waste disposal charges, interest rates, overhead, administration, property taxes and insurance will all be specific not only to Canada, but may vary among provinces in certain cases. These costs are particularly important for estimating Operation and Maintenance costs, which can be substantial for certain control systems. AERCo$t calculates each of these items individually to determine the most realistic cost possible for a Canadian industry.

Assumption of No Current Controls

Control efficiencies assume that no controls have been previously installed. While the report states that this assumption is likely untrue for many sources in Canada an analysis of U.S. EPA AIRS data (www.epa.gov/airsdata/) on current control installation levels for all types of sources showed that the majority of emission sources had an insignificant level of controls in place for the pollutants of concern. It was assumed that a similar pattern would exist in Canada but some validation of this assumption would be helpful.

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[30] Supporting arguments cited: a free trade environment will lead to similar business/environmental compliance strategies and multinationals in Canada and the US will use similar control methods.

Assumption of Treating Least Expensive Sources

The least expensive 15% sources in each sector will represent the pool of Canadian sources from which reductions will occur under new regulations. Because control options for up to 75% emission reduction are presented, these least expensive sources should account for up to 75% of all emissions from the sector. Stratus tested the sensitivity of this assumption by examining the use of the least expensive 10% and 20% of sources. They report that the inclusion or exclusion of an additional 5% of control cost data, either side of the base case (15%) also had little impact upon total control costs with the exception of NO_x reductions in Alberta, New Brunswick and Nova Scotia. The 10% control cost scenarios are the most expensive and the 20% the least expensive. The 10% control cost scenario was explained as the selection of control options with a smaller number of observations for reducing of NO_x from the electric power generation sector – the cost for this option is 5 times more. In the case of the 20% control cost scenario, this is due to the availability of cheaper technology options *(*see Stratus Consulting, 2000).

The use of the EPA dataset assumes that the distribution of small and large, expensive and inexpensive emission sources in Canada is similar to that in the United States. By assuming that only the least expensive controls will be required to reduce emissions, the total potential emission reductions is limited. Because control efficiencies are engineering efficiencies, with no consideration for policy specifications, they may overestimate emission reductions or underestimate total costs. By using the least cost sources to estimate costs, the cost results apply only to that subset of sources within each sector. The results provide no insight into the costs of controlling higher cost sources of emissions. Furthermore, it is not clear exactly what "least expensive 15%" means and additional clarification of this procedure is required. However, any attempt to employ cost minimizing control technologies raises issues concerning the focus of the analysis.

Direct cost analysis is intended to provide cost estimates for a change in a regulatory regime. If is it assumed that the least expensive approaches to meeting the regulatory requirements are employed, this provides an estimate of the least cost approach to meeting the regulation, however, it does not provide an estimate of the actual cost of the regulation if the regulatory approach does not involve movement to a least cost reductions strategy. As discussed above, the cost of regulatory change will depend on the approach taken to implement the emissions control. Incentive based policies, designed to achieve the lowest costs of meeting a standard, should result in the lowest cost of regulatory change. However, if command and control approaches are employed, the costs of a regulatory change will likely be higher. Also, the "15% cheapest" assumption appears to operate within each province (although it is not clear as to whether this was in fact the approach used). The result is that the CWS approach provides information on the cost of control of a relatively small sub-sector of the overall emitting sectors within the economy. If sector wide controls are implemented, the CWS approach will underestimate costs.

Cost Floor Assumptions

Cost floors were used as a conservative lower boundary for estimated costs. The costs floors of $150 USD/ton for all pollutants except NO_x ($100 USD/ton) were based upon industry experience and a review of the various input data sources. The sensitivity of this assumption was tested by Stratus (see Stratus Consulting Inc., 2000). Removal of the cost floors was stated as having little impact upon the total costs of control. Negative control costs occurred mostly for technologies that reduce VOCs ("stripper and equipment" and "new CTG level control"). Because these technologies are applied to the "pulp and paper" and "manufactured products" industries respectively (neither of which are the most significant emitters of either pollutant – relative to other industries), the impact of the cost floor assumption upon total costs was minimal. However, cost floors significantly impact the results of the SO_2 analyses, doubling the costs of 25% reduction in the electric power generation and chemical fertilizer manufacturing industries.

In some cases the CWS / Stratus model results in "negative costs", that is, the technology prescribed in the emission reduction scenario implies that control costs will actually be lower than they are today. Some of these negative (or in some cases very low) costs appear to be difficulties with the EPA / Pechan database. The sensitivity analysis performed by Stratus illustrates that in general these few negative cost cases did not significantly affect the cost of control estimates. However, for some industrial sectors, the costs of control were increased by as much as 25% when assumptions on low or negative control costs were changed.

Assumptions on Specific Technologies

- The application of SCR to natural gas fired boilers was excluded in developing the control costs because it is an extremely expensive means of NO_x control.
- Control options of "small" sources were not included in the dataset used to develop control costs. The following control options were excluded from the dataset:
    - Controls that reduced less than 1 ton of NO_x
    - Controls that reduced less that 0.5 tons of VOCs
    - Controls that reduced less than 1 ton of PM_{10} or $PM_{2.5}$
    - Controls that reduced less than 1 ton of SO_2
- Control option data for "other paper converters" industry (SIC 2740) were used to approximate options for the "corrugated box" industry (SIC 2732)
- Control option data for "chemical industries" (SIC 3700) was used to approximate options for the "ammonia" and "red phosphorous" industries (SICs 3713, 3714).
- Transportation control costs for Yukon and Northwest Territories were assumed to be the average of the transportation control costs experienced by all other provinces.
- Control options not available in the input data were estimated using industry expertise and a review of the literature and are identified in Appendix A of the Stratus Consulting Report.
- Little or no documentation was provided on how the CWS approach examined the Transport sector. It was stated that Mobil5 and AERCo$t were used to calculate transport sector costs, but the Panel received no detail on these calculations.

Scaling Assumptions

To simplify the scaling methodology, decision rules were developed that the computer program could use to develop the scaled cost estimate. The control options with the control efficiency closest to the target policy level (25%, 50% and 75% reduction) were selected. This was not a problem for industries and pollutants with a wide range of control options (whose costs were a positive function of the control efficiency). However, for industry/pollutant combinations with only one control option or in cases where more effective controls were less expensive than the less effective controls, it is possible that the decision rules resulted in an unreasonable scaled cost calculation. Furthermore, the control option descriptions used for the scaled analysis are misleading. For example, assume that the only control option listed for PM_{10} from a type of construction industry was water suppression and this control may have been listed in the U.S. EPA data as being capable of a 25% reduction. The scaling methodology assumes that at some higher cost, this technology would be capable of up to a 75% reduction. It may be impossible to reduce 75% of PM_{10} emissions using water suppression and may require additional types of control not included in the data. However, the Panel understood that the initial policy reduction levels do not exceed 45%, which reduces the likelihood of this type of problem.

The CWS / Stratus approach requires assumptions regarding the application of control technologies beyond the ranges that are defined in the database. When the control technologies were not able to reduce emissions to the degree required (e.g. 75%) it was assumed that the technology could achieve this level of reduction and the costs were proportionally scaled assuming a linear cost function. There are two significant assumptions in this approach, first that a control technology can be applied to higher levels of emission reduction and second that the marginal costs of reducing additional emissions are constant. Caution is suggested when interpreting results that were scaled in such a fashion and it is recommended that all such scaled cost estimates be identified in the analysis. A question that is raised about this issue is the accuracy of the emission reduction cost estimates for the higher (75%) levels of emission reduction. Since the attainment of target policies (Target A and B below) never exceed 45% emissions reduction, this may not be a significant effect, however, even for lower levels of emission reduction the identification of the impact of the scaling assumptions would be helpful.

The relationship between the CWS reductions of 25, 50 and 75 percent of emissions, and attainment of target ambient air quality level is not clear in the presentation of the costs of control. Specifically, a table specifying the relationship between provincial emission reductions (by percentage) and ambient air quality targets is unclear. The targets specify ambient standards for PM and ozone (e.g. Target A: PM_{10} 60ppb, $PM_{2.5}$ 30 ppb and ozone

65 ppb) while the emission reduction targets are expressed in percentage terms by province and by pollutant (PM_{10}, $PM_{2.5}$, SO_x, NO_x, and VOC). As well, the percentage emission reductions specified to meet Targets A and B outlined in the report, never exceed 45% in any province and yet the cost simulations are performed for 25, 50 and 75% emission reductions.

Control Cost Database Issues

The CWS / Stratus approach is based on U.S. SCC (source classification code) data that are based on generators of common emissions rather than industry categories. These data are converted to SIC (Standard Industrial Classification) codes. The U.S. typically analyzes control costs on a SCC, or emission formation process basis. Analyzing emission reductions on an SIC basis may provide a better understanding of the impact of potential regulations upon certain sectors of the economy but it does lead to confusion in analyzing control options for industries whose end products also produce emissions (i.e. the woodstove industry). The report suggests that the best way to deal with this problem is to incorporate both SCCs and SICS into the analysis. The CWS approach, since it employs aggregates of industrial sectors in order to be able to utilize the U.S. database on technologies and costs requires analysis to be done at a relatively high level of aggregation. The approach assumes that industrial sectors in Canada are very large, with many sources that could be controlled by the same system. This may be the case in the U.S. where the industrial base is much larger, but it cannot be assumed to be the case in Canada. If there are only a few sources in the sector, or the control option only applies to a few sources, this method breaks down.

If assessment within province and within industrial sector is required, the Stratus approach is lacking. In addition, if disaggregation to the individual plant level is required for detailed analysis of a specific case, the Stratus approach cannot be employed.

Alternatively, AERCo$t allows the user to conduct certain sector or provincially based analyses that would be very useful for this type of assessment. Rather than applying a scaling methodology as described by Stratus, the program allows the user to select strategy options such as Lowest Cost at the control system or the sector or Provincial levels, Lowest Achievable Emission Rate, or Lowest Cost Effectiveness ($/tonne of pollutant removed). The analysis strategies allow the determination of the lowest cost option to achieve any level of overall reduction, including 25%, 50% or 75% if these levels are achievable in the sector or Province. This strategy selects the lowest cost means of achieving each level of overall control, without modifying control efficiencies or costs. This method is possible, because the control costs and reductions are all calculated at the process level, and can therefore be rolled-up by various methods. The results, assuming that the input data are reliable, are then more realistic.

AERCo$t also attempts to deal with the issue of selecting a control system when one already exists at a specific site. The database is programmed to reject certain control options when the Federal emissions inventory, the Residual Discharge Information System (RDIS) indicates that a control system is already in place at the site. While this offers the potential for improved cost estimates, it relies upon the accuracy and completeness of the emissions inventory data. While in the case of missing flow rate information certain assumptions can be made, it is more difficult to assume the presence of a control system, unless the industry is contacted directly. The inventory is far from being complete at this time with respect to identifying existing controls at specific sites.

7.3.2 Additional Concerns Regarding Cost Estimation

The following is a summary of more general concerns associated with the Stratus / CWS approach to cost estimation.

<u>Lack of Behavioural Response to Control Requirements</u>

A significant shortcoming of the broad-based approach is its inflexibility in assigning control options to each sector. Only “technical control options” were considered. Consideration of creative alternatives such as fuel switching, boiler tuning, repowering or episodic controls – alternatives that are very likely to be used by industries attempting to comply with new emissions regulations- were deemed to be beyond the scope of the project. Failure to include such alternatives is likely to have overestimated total control costs. Stratus qualified their findings by recommending that the results only be used in the context of a screening-level of policy analysis, and not be expected to represent the type of source-specific control cost results that could be obtained from a detailed analysis of individual sources of emissions in Canadian industries.

Lack of Consideration of Baseline

In developing cost estimates a clear description of the baseline is required. The CWS / Stratus approach does not appear to have a well-defined baseline. A well-defined baseline will include the following considerations:

- Over the life of the "project" (period of analysis of the regulatory change) technological change will affect the industries and thus the costs of control will change over time. The net present value of control costs should reflect such technological change.
- The assumptions about current levels of abatement must be made explicit and be carefully evaluated. It appears that the CWS approach assumes that there is currently no abatement of any of the relevant emissions. This assumption should be carefully assessed because if there currently is a level of abatement effort, and the analysis assumes none, the projected costs of meeting the new regulation could be understated because marginal costs generally rise with increased abatement.
- A baseline must include expected regulatory changes (as currently enacted and projected to impact the emitting sectors). This involves explicit consideration of the fact that additional regulatory requirements will come on stream before the end of the time horizon, affecting the costs of emission reduction. The costs of emission reduction from an existing policy that will become effective in the near future cannot be considered as costs of a new regulatory change. The result will be an understatement of the costs of new emissions reductions for the same reason as above. Furthermore, the baseline should include a compliance baseline. If compliance is not complete, abatement costs for the new regulation could be overstated.
- The baseline should include projections of economic growth that generates increased industrial activity. Economic growth without the change in

regulation is required to assess the costs on the projected economy, rather than only the current economy. If the emitting sectors of the economy are expected to grow in the future, the regulatory process needs to be evaluated in light of the projected larger size of the affected industry. Without consideration of economic growth, the costs of meeting an ambient air quality regulation will be substantially underestimated. There is likely an interaction between the projection of technological change and the projection of economic growth as one would assume that the technology available to industry in the future would be better (and more cost effective) than that available to existing firms, which would tend to reduce the underestimation of costs that will be caused by ignoring economic growth.

- The CWS approach does not appear to include many of the important components of a baseline. CWS appears to assume a static industrial structure, no economic growth and no assumptions of improved technology in the future. The costs of reaching emission reductions are examined for the current industrial complex (Arnold, 1995).

Transparency of the Cost Model

There are significant difficulties understanding exactly how the cost model was implemented and there are difficulties interpreting the results. The assumptions of “15% cheapest”, and the scaling assumptions are examples of issues with inadequate transparency. Furthermore, it is not clear how the model apportions control actions in cases where marginal costs of control are identical.

Concerns Regarding the Canadian Emissions Inventory Data (RDIS)

The accuracy of the RDIS database as the basis for emissions is questionable because it is a voluntary emissions registry with inadequate detail and is somewhat dated. There was much negative commentary about this inventory from industry. The quality of any emissions inventory will ultimately depend upon the quality of the data provided by industry, so there seems to be an excellent opportunity for industry to improve the quality of the inventory by taking an initiative to work co-operatively with government on addressing this need.

No Treatment of Uncertainty

There is no evaluation of uncertainty in the cost analysis approach. Some (limited) sensitivity analysis has been performed, but there is no consideration that the estimates of cost (even the direct estimates) are likely expected values rather than deterministic amounts. The inclusion of potential behavioural responses to emission regulation and partial or general equilibrium effects will also affect the level of uncertainty of the cost estimates. Information on uncertainty is an important component of policy analysis and attempts should be made to reflect the degree of uncertainty in the cost estimates. Note that uncertainties are reported in the benefits estimates and thus by not reporting variances in the cost estimates it appears that these measures are somehow more "accurate" than the benefit measures. Given all the problems with the CWS cost estimation approach the Panel does not believe that cost estimates are necessarily more accurate than the benefits estimates.

Economies of Scale in Emissions Reduction and Multiple Pollutant Technologies

The analysis treats control options for each pollutant independently. In truth, several types of control technologies impact more than one pollutant. In most cases, these co-control issues would result in the overestimation of costs. To appropriately estimate the total costs of reducing multiple pollutants it is necessary to consider both the impact of a single control on multiple pollutants and the interaction of controls aimed at separate pollutants. In many sectors, the contribution of co-benefits may be significant, depending on the assumed baseline levels of controls. There are many studies that focus on the ancillary benefits of greenhouse gas mitigation policies, in terms of conventional air pollutant reductions. For example, Burtraw et al. (1999) report on various GHG reduction simulations that generate significant reductions in NO_x, SO_2, particulates, VOCs and other pollutants. For the CWS process, the ancillary benefits would be associated with reductions in GHGs and conventional pollutants other than PM and ozone. Furthermore, the CWS/Stratus approach does not appear to address economies of scale in the treatment of emissions.

7.4 Broadening the Scope of Cost-Benefit Analysis: General Equilibrium Methods and Trade Analysis

The scope of a Cost-Benefit Analysis is an important consideration. CBA typically examines the direct benefits and costs associated with the policy issue, for a pre-determined area (or accounting stance) and time. The degree to which *ripple effects* throughout the economy or linkages with other countries through international trade or international ecological connections are considered is a challenging issue to resolve and is often not part of such analyses.

The CWS approach, for example, uses a National accounting stance, with Provincial level sub-analyses, and uses a 30 year time frame (2005-2035) for the benefits analysis and annual cost to control 1995 emissions. This approach tends to ignore "ripple" effects throughout the economy (both costs and benefits) and it treats all elements outside of the study area as constant and/or exogenous. Both of these shortcomings can be addressed by expanding the scope of the analysis.

Ripple effects within the national economy can be addressed using general equilibrium methods. These methods were described in Section 2.2.4. Note that general equilibrium methods have been employed in the analysis of U.S. air quality regulatory policy. These methods would provide additional information on the indirect costs associated with regulatory change, but substantial investments are required in order to develop the tools to implement adequate general equilibrium analysis tools. Nevertheless, general equilibrium effects of policy reform should be a legitimate area of study and is one of the issues that must be considered beyond simple cost-benefit analysis.

To illustrate the importance of this issue, we note that in the EPA's recent analysis of the 1990 Amendments to the Clean Air Act's costs and benefits, the so-called tax interaction effect was incorporated for the first time in an EPA regulatory analysis. This effect represents the loss to society from regulations that are costly enough to raise the price of goods and services, and thereby reducing the real wage, and increasing the "deadweight" loss from labor taxes. These costs were thought by some of those reviewing the study to be 25% or more of the direct costs of the Clean Air Act Amendments. In addition to ripple effects within the study area (i.e., Canada), linkages between Canada and other nations could also be considered. If regulatory policy changes result in changes in trading patterns (or competitiveness), these impacts should be considered in the policy analysis.

7.5 Conclusions

The CWS approach to regulatory cost analysis summarizes a significant amount of information on control technologies, costs, and methods for attaining emissions reduction targets. It is based on direct control costs, an approach that has its limitations if, as we expect, there are general equilibrium impacts on the economy. However, we also recognize the significant effort that is required to capture these economy-wide impacts and suggest that this is a long term research issue. The analytical approach makes many simplifying assumptions, as do all practical approaches to policy analysis.

Many limitations of the CWS approach to cost estimation have been identified above, when held against the benchmark of the U.S. Prospective Study (U.S. EPA, 1997), or the U.S. Retrospective Study (U.S. EPA, 1999). This is a very high benchmark, but the CWS ambient air quality standards for ozone and PM are likely to be the most expensive single environmental standards to meet in Canadian history. As such these CWS deserve thorough treatment. Fortunately, some elements of the cost analysis can be improved at lower cost and with less effort than others. Extensions of cost analysis to include general equilibrium and international trade considerations can provide important information for policy analysis. The scale of the analysis (national including direct and general equilibrium effects; international including trade effects, etc.) is an important element to consider and will also help identify the impacts of the regulatory proposal, in terms of benefits and costs as well as the incidence of the impacts.

7.6 Recommendations

The Panel suggests that the CWS cost estimation be improved by taking the following relatively low cost steps:

- Improved consideration of Canadian industry and source emission categories (SIC and SCC combined) and treatment options, to the plant level including "ground-truthing" of control costs.
- Consideration of the likely pollution intensity and marginal product of new technologies (both production and abatement).
- Assessment of existing emissions control implementation.
- Consideration of non-technical approaches to emissions reduction (fuel switching).
- Consideration of co-benefits or multiple pollutant reductions with individual technologies.
- Careful consideration of the baseline and explicit description of the assumptions involved in the baseline. The development of the baseline may include the consideration of alternative regulatory approaches including incentive approaches for emission reduction.
- Increased transparency in the modeling of direct costs.
- Assess the degree of uncertainty in the cost estimates.

The Panel believes that the AERCo$t model can address some of these issues. Elements that will require substantial additional resources and research include:

- Improvement of the RDIS database for the basis for cost analysis to a level comparable to the current U.S. inventory.
- Assessment of the degree to which partial or general equilibrium methods should be applied to regulatory policy. The development of general equilibrium models can be a costly exercise, and they carry a set of assumptions that must also be evaluated carefully, however, in many cases these models represent the best available technology for assessment of economy wide impacts of regulatory change. The U.S. Retrospective study, for example, chose to employ the Jorgenson-Wilcoxen dynamic general equilibrium model of the U.S. economy (Jorgenson and Wilcoxen (1990b)).
- Research on the tax interaction effect, in a Canadian context.

The Panel endorses the use of a cost-benefit framework for the analysis of environmental regulation that includes an accurate assessment of the costs of regulatory change. The Panel recognizes the empirical limitations of CBA and recommends the following:

- Continued development of methods for accurate assessment of costs and benefits, including methods for the analysis of general equilibrium (including tax interaction) effects and international trade impacts of regulatory change.
- Investments in human capital in the area of CBA of environmental regulation so that policy makers and the Canadian public can be confident that cost and benefit measures accurately reflect Canadian values and preferences and Canadian institutional arrangements.

7.7 Summary

Table 26 provides a summary of the Panel's assessment of the CWS approach to estimating costs associated with PM and ozone emissions reductions including the key limitations, uncertainties and recommendations for alternative approaches.

Table 26: Summary of Panel's Assessment of CWS Approach to Cost Estimation

ISSUE	**BASELINE ASSUMPTIONS**
CWS APPROACH	Assumes no other existing or future air quality management policies, a static industrial structure, no economic growth, no existing abatement technologies in place, no future improvements in technology.
PANEL CRITIQUE Key Limitations	CWS does not attempt to define or quantify baselines
RELATIVE UNCERTAINTIES (Probably Minor, Potentially Major)[31]	Potentially major
DIRECTION OF BIAS[32]	Projected costs of meeting new regulations could be understated

[31] Likely Significance Relative to Key Uncertainties on Net Benefits Estimate: Probably minor (alternative assumption or approach could influence overall estimate by <20% difference), Potentially major (>20% difference). Adapted from US EPA study The Benefits and Costs of the Clean Air Act 1990 to 2010" Nov. 1990 study in which 5% difference was used see pg. 21, 33, 65, 79, 98.

[32] The U.S. EPA report "The Benefits and Costs of the Clean Air Act 1990 to 2010" Nov. 1999 used the following: Overestimate, Underestimate, Unable to determine based on current information

Table 26: (cont'd)

RECOMMENDATION/ ALTERNATIVE INPUTS, TOOLS, APPROACHES	Definition of baseline is essential in a CBA study. Future CWS studies need resources to include proper estimates of: Impact of current and projected Canadian and U.S. regulatory policy Technological change Compliance baseline Projections of economic growth Demographic changes
ISSUE	**COST OF EMISSION REDUCTION**
CWS APPROACH	Based on 1995 emissions Based on U.S. control cost data analyzed at process (SCC) level Smallest sources not included, costs less than $100/ton for NO_x controls and $150/ton for all other pollutants were eliminated Only considered the 15% least expensive sources Assumed that no control systems are currently in place Conversion of 1990 U.S. $/ton to 1995 CDN$/tonne assumed GDP deflator of 1.166029 and 15% reduction in relative cost of control technology inputs Costs are based on direct regulatory approaches without consideration of the potential for market instrument mechanisms
PANEL CRITIQUE Key Limitations	Assumes that all processes in a sector can be controlled by the same system, and that the cost will be independent of the size of the process Assumes similarity in cost and technology structure between the US and Canada Assumes that costs are linear with emissions, this is only valid in certain cases Costs are based on engineering costs that do not consider behaviour or market responses Tax interaction effect is not included Lack of consideration of baseline (technological change, current levels of abatement, regulatory change, economic growth) No evaluation of uncertainty Lack of transparency in implementation of model and interpretation of results Accuracy of Canadian emissions inventory data (RDIS) Impact of single control on multiple pollutants and interaction of controls aimed at separate pollutants not considered
RELATIVE UNCERTAINTIES (Probably Minor, Potentially Major)[33]	Some assumptions may have potentially major effects on cost estimation.

[33] Likely Significance Relative to Key Uncertainties on Net Benefits Estimate: Probably minor (alternative assumption or approach could influence overall estimate by <20% difference), Potentially major (>20% difference). Adapted from US EPA study The Benefits and Costs of the Clean Air Act 1990 to 2010" Nov. 1990 study in which 5% difference was used see pg. 21, 33, 65, 79, 98.

Table 26: (cont'd)

DIRECTION OF BIAS[34]	On balance it is likely that costs are underestimated if the tax interaction effects are as significant as they appear to be in the recent literature.
RECOMMENDATION/ ALTERNATIVE INPUTS, TOOLS, APPROACHES	Low Cost Improvements: Improved consideration of Canadian industry and source emission categories (SIC and SCC combined) and treatment options Ground truthing of control costs to the plant level Assessment of existing emission control implementation Consideration of non-technical approaches to emissions reduction (fuel switching) Consideration of co-benefits or multiple pollutant reductions with individual technologies Development of the baseline including consideration of alternative regulatory approaches (incentive approaches to emission reduction) Increase transparency in modeling of direct costs Assess degree of uncertainty in costs estimates Higher Cost Improvements: Improve RDIS General equilibrium methods should be applied to regulatory policy Assess costs under incentive based regulatory schemes Research on tax interaction effect in a Canadian context Continued development of alternative decision-making frameworks as methods to triangulate with traditional CBA Investment in human capital to improve CBA of environmental regulation

[34] The U.S. EPA report "The Benefits and Costs of the Clean Air Act 1990 to 2010" Nov. 1999 used the following: Overestimate, Underestimate, Unable to determine based on current information.

7.8 References

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8 Valuation of Health and Non-Health Benefits

8.1 Valuation of Health Effects

A wide variety of possible health effects arise from improvements in air quality. To perform a cost-benefit analysis, these effects need to be monetized and aggregated. This process of valuation is challenging and is discussed in detail in Davis, Krupnick and Thurston (2000). The original approach to valuing mortality risk reductions was the human capital approach. It viewed the value of a person's life as their productive value, adding up the lost productivity from premature death as a measure of loss. It was generally recognized that this measure was quite partial and problematic, not reflecting people's preferences for reducing death risks, and basically assigning non-workers a zero value. But, the human capital estimate was easy to calculate and was thought to be better than nothing. Because, at least in developed countries, superior alternatives are available, this approach is now rarely used in such countries.

The two most common approaches to estimating willingness to pay for health improvements include hedonic labor market studies and stated preference methods, such as contingent valuation surveys. The former statistically relate wage differentials to mortality or morbidity risk differences across occupations and industrial/commercial sectors, under the theory that in competitive labor markets, workers in risky jobs should receive wage premiums equal to the value they place on avoiding increased mortality or morbidity risks. Workers are asked their perception of the death risks they face to address the issue that their behaviour would be consistent with perceived risks rather than historic risk estimates and these two types of risks might diverge. Labor market studies are numerous and form the foundation for most VSL estimates. However, they are problematic for being applied to health effects of air or other pollutants because the behavioural context being observed and/or the population observed in this behaviour are different than that applicable to the health effects. In particular, epidemiological studies suggest that reducing air pollution lowers death rates primarily among persons over 65. These benefits, furthermore, are more likely to accrue to people with chronic heart or lung disease and may occur with a lag.[35] There is a growing consensus that the appropriate, if challenging, valued "commodity" is an increase in the probability of surviving to all future ages given a shift in the survival function.

Attempts have been made to adjust estimates of risk reductions from the labor market literature for age and latency. Under certain strong assumptions, one can convert the value of a statistical life (VSL) from a labor market study (or other source) into a value per life-year saved (Moore and Viscusi, 1988). The value of a life-year can then be multiplied by discounted remaining life expectancy to value the statistical lives of persons of different ages. To illustrate this calculation, suppose that the value of a statistical life based on compensating wage differentials is $5 million, and that the average age of people receiving this compensation is 40. If remaining life expectancy at age 40 is 35 years and the interest rate is zero, then the value per life year saved is approximately $140,000. If, however, the interest rate is 5 percent, then *discounted* remaining life expectancy is only 16 years, and the value per life-year saved rises to approximately $300,000.[36]

The United Kingdom Department of Health (U.K. Dept. of Health, 1999) has recently presented another, relatively ad hoc approach to adjusting VSLs for a variety of shortcomings. The elaborate set of adjustments to the standard VSL ($2.4 million) illustrates the problems with this standard probably more than it increases certainty about what the "true" VSL is. The authors start from a standard VSL of $2.4 million. The upper bound estimate is 70% of the VSL ($1.7 million), adjusted because the affected group is elderly. For the mid and low estimates, the high estimate is adjusted further to account for shorter life expectancy (assumed to be 12 times shorter based on an interpretation of the short-term mortality studies) and the worse health status of those

[35] The delay in the realization of risk reductions could occur either because the installation of pollution control equipment today will not benefit young people until they become susceptible to the effects of pollution (the air pollution case described above), or because the program reduces exposure today to a substance that increases risk of death only after a latency period (e.g., asbestos).

[36] Similar adjustments can be made to account for the effect of latency periods. According to the life-cycle model, a 40-year-old's WTP to reduce his probability of dying at age 60 should equal what he would pay to reduce his current probability of dying at age 60, discounted back to age 40.

affected relative to others their age. This is $130,000. For the low estimate, larger adjustments are made for the same reasons, to yield a VSL of only $3,100!

There is also a small literature of consumer preference studies that attempts to estimate the WTP to reduce death risks from purchase or other actual decisions by consumers, say in purchasing smoke detectors (Dardis, 1980). These studies tend to find lower VSLs. One problem with some of these studies is statistically separating the mortality risk-reducing attribute from other attributes of value to individuals.[37]

The stated preference approaches, of which contingent valuation and conjoint analysis are the two most prominent, are survey approaches that set up choice situations and use the (hypothetical) choices (to be willing to pay some amount, or to vote yes on a referenda, or to prefer one package of attributes over another) to recover preferences for mortality risk reductions. The ability of ratings-based conjoint analysis to recover preferences is a matter of debate, however choice-based stated preference methods are consistent with economic theory. Also, both of these approaches may suffer from a variety of their own biases and their results have been shown to be very sensitive to question wording and ordering. They are capable of being molded to whatever population and context are appropriate, however. And respondents can be tested for their cognition and understanding of the issues being examined in the survey. (See Hammitt and Graham (1999) for a detailed discussion of the CV-mortality risk valuation literature).

Some of the best known CV studies for mortality risks (Jones-Lee et al., 1985; Hammitt and Graham, 1999) look at traffic fatalities rather than deaths in a pollution context, hence we make a distinction between these two types of CV studies. One Canadian study used conjoint analysis to examine WTP for reduced mortality risks in a pollution-type context (Desvousges et al., 1996) but it was assumed that a product could deliver a *certain* improvement in lengthening of life, rather than a probabilistic one. Several studies have used CV approaches to examine WTP in a

[37] We ignore here the large body of literature using an hedonic property value approach. This approach provides a revealed WTP for air pollution reductions but is dependent on housing market perceptions about pollution and links to all types of effects, health being only one. It has the advantage (some would say disadvantage) of not using any concentration-response information.

context applicable to mortality risk reductions from pollution (Johannesson and Johansson, 1995; Krupnick et al., 2000).

Johannesson and Johansson were the first to test for WTP for an increased life expectancy (one year in expectation) added between ages 75 and 85. They find implied VSLs ranging from $70,000-$110,000 for the sample surveyed by phone in Sweden. This study is problematic, however, as it does not provide any indication of whether respondents understood the complex scenario and offers respondents what is actually an unrealistically large reduction in risk.

The most recent study that may be useful for understanding WTP of groups at risk from air pollution in the context of the nature of this risk is Krupnick et al. (2000), which surveyed 930 individuals living in Hamilton, Ontario. Their estimates of mean WTP translate into values of a statistical life of approximately C$3.8 million (1999 C$) for a 1 in 10,000 annual risk reduction and C$1.2 million for a 5 in 10,000 annual risk reduction, or U.S.$3.04 million and U.S.$0.96 million, respectively. These are 10 to 70 percent lower than Health Canada's age-adjusted VSL of C$4.3 million (1999 C$) and one-half (or less) the size of the U.S.$6 million (1999 US$) figure used by the U.S. EPA.

Krupnick et al. (2000) also find that WTP does not vary much by age, up to 65. Persons 40 to 49 years old do have slightly lower WTP than persons 50 years of age and older; however, mean WTP (C$657 for the 5 in 10,000 annual risk change), which translates into a VSL of C$1.3 million, remains approximately constant until about age 70, decreasing by about one-third thereafter. This latter WTP (C$417, or a VSL of about C$800,000) is probably the most relevant one for use in valuing most of the lives "saved" from air pollution reductions. Regardless of the measure of physical health status used (with one exception), WTP was found not to vary appreciably with physical health status either—an important result for environmental policy, since older people and people with chronic conditions are often the beneficiaries of improvements in environmental quality. Individuals with cancer, however, were found to be willing to pay over 40% more for a mortality risk reduction than their counterparts without cancer, and individuals in better mental health have a larger WTP than those scoring lower on tests of their mental health.

Table 27 provides information on health effects of conventional air pollutants that have been (or could be) monetized, based on our understanding of the literature. For each of these effects, we list the techniques used to provide monetary values. WTP is *willingness to pay* measures, or those that provide estimates of preferences for improved health that meet the theoretical requirements of neoclassical welfare economics. COI is *cost of illness* measures, obtained by totaling up medical and other out of pocket expenditures. COI measures typically underestimate WTP. *Consensus* refers to the way in which these values were determined, implying that they do not have much of an evidentiary basis. Each of these approaches and effects are discussed in more detail in Davis et al. (2000). The AQVM draws on the same literature underlying this table. Because estimates of the value of a statistical life drive the benefits analysis, more detail on this measure is provided in Figure 9.

Table 27 Status of Valuation of Health Effects

Health Effects	Valuation Estimates Available?	Basis
Mortality: Adults	Y	WTP (caveats)
Mortality: Neonatal/fertility	Y	WTP; Number of studies on-going
Mortality: Children	Soon	Number of WTP studies on-going
Cancer Mortality and Morbidity (various types)	Y	COI; WTP
Chronic Bronchitis	Y	WTP (caveats)
Acute Bronchitis	Y	COI
Hospital Admissions	Y	Hospital Costs
Emergency room visits	Y	Emergency room costs
Lower respiratory illness	Y	WTP (caveats)
Upper respiratory illness	Y	WTP (caveats)
Respiratory symptoms	Y	WTP
MRAD (Minor Restricted Activity Days)	Y	Consensus
RAD (Restricted Activity Days)	Y	Consensus
WLD (Work Loss Days)	Y	Wage
Asthma Day	Y	WTP
Change in asthma status	N	

Table 28 provides a small sample of the midpoint values typically used by practitioners of health benefits analyses, as well as ranges of these values. We picked the unit values for health endpoints chosen by four major studies or models in the U.S., Canada and Europe, ordered from highest to lowest based on the first of these studies--the U.S. study on the Costs and Benefits of the 1990 Clean Air Act Amendments--and put them in common currency and constant dollars.

The Value of Reducing Health Risks

Individuals often make choices that reflect consideration of health risks. They may purchase automobiles with enhanced safety equipment. They may purchase air purifiers because of concerns over air quality and the potential for illness. Purchases of bicycle helmets, sunscreen, or carbon monoxide detectors are all indicators of the choices that consumers make that reflect concerns over health and safety risks. The trade-offs that individuals make in the marketplace relating to health risks provide information on the amount that people would be willing to exchange for a reduction in the risk of illness or death. Workers also have the opportunity to make choices about activities in the workplace and part of that choice may reflect considerations about relative safety risks and the relative wages in different jobs. Economists sometimes rely on data from occupational choices to calculate the value of reducing health risks. In addition, highly structured surveys can be used to identify trade offs that people would make in response to small changes in health risks. All of these approaches provide information on the value to an individual of reductions in mortality or morbidity risks.

Imagine that we observe two occupational categories, and we are able to control statistically for all the non-safety related differences between these jobs to find the difference in wage associated with differences in safety. We find the difference to be \$500 per year and to be associated with an increase in the risk of a fatal accident of 1 in 10,000 per year. This indicates that individuals are willing to trade off \$500 in income for a 1 in 10,000 reduction in mortality risk. A program that reduced mortality risks by this amount for 10,000 people would generate benefits of \$5 Million (10,000 x \$500). Knowing the value of small risk reductions for individuals leads to the calculation of the benefits of a risk reduction program for the affected population. Note that reducing mortality risks by 1 in 10,000 for a population of 10,000 people is statistically equivalent to reducing 1 mortality or 1 *statistical life*. Thus, the estimate of \$5 Million has been referred to as the *value of statistical life (VSL)*. VSL is a misleading label and is better represented as a *value of reducing risk of death*.

Though conceptually simple, this type of calculation has plenty of practical problems when used as a measure of preferences for reducing mortality risks. In the labour market, workers may not have the economic freedom to choose among occupational alternatives. Further, it is not easy to control for all the differences in occupational categories unrelated to safety that may be contributing to differences in wages. Also, one must account for the risk of injury separately from accounting for the risk of mortality. If products like bottled water or organic food are used to assess willingness to pay for reductions in risk there are difficulties in separating out the risk reduction aspects from the other benefits arising from the product (taste, convenience, etc.), and questions arise regarding the quantitative measurement of the risk reduction arising from such products versus the range of beliefs that may be determining willingness to pay. Survey methods, including contingent valuation methods, can control for many of these issues, but other concerns associated with the survey approach arise. Note also that the discussion above does not consider the dimension of time. The concept that is more relevant to most discussions of environmental policy is the tradeoff individuals make to increase the probability of living for an additional specified period of time (e.g. 1 year of life beyond expected values). In the jargon of the literature this is referred to as the *value of a statistical life year* but again it should be thought of as the value of reducing risks of premature mortality, where premature is defined relative to population life expectancy.

Researchers continue to develop methods to refine the estimates of how individuals make trade off decisions relating to health and safety risks.

Based on: Burtraw and Krupnick (1999)

Figure 9 The Value of Reducing Health Risks

Table 28 Comparison of unit values used in several major studies or models. ($1990).

Values	US EPA[a]			US TAF[b]			Canada AQVM[c]			Europe ExternE[d]
	Low	Central	High	Low	Central	High	Low	Central	High	Central
Mortality	1560000	4800000	8040000	1584000	3100000	6148000	1680000	2870000	5740000	3031000
Chronic Bronchitis	-	260000	-	59400	260000	523100	122500	186200	325500	102700
Cardiac Hosp. Admissions	-	9500	-	-	9300	-	2940	5880	8820	7696
Resp. Hosp. Admissions	-	6900	-	-	6647	-	2310	4620	6860	7696
ER Visits	144	194	269	-	188	-	203	399	602	218
Work Loss Days	-	83	-	-	-	-	-	-	-	-
Acute Bronchitis	13	45	77	-	-	-	-	-	-	-
Restricted Activity Days	16	38	61	-	54	-	26	51	77	73
Resp. Symptoms	5	15	33	-	12	-	5	11	15	7
Shortness of Breath	0	5.3	10.60	-	-	-	-	-	-	7
Asthma	12	32	54	-	33	-	12	32	53	36
Child Bronchitis	-	-	-	-	45	-	105	217	322	-

a. The Benefits and Costs of the Clean Air Act 1990 to 2010. (U.E. EPA 1999e) Low and high estimates are estimated to be 1 standard deviation below and above the -mean of the Weibull distribution for mortality. For other health outcomes they are the minimums and maximums of a judgmental uniform distribution.

b. Tracking and Analysis Framework *(www.lumina.com/taf/index.html)*, developed by a consortium of U.S. institutions, including Resources for the Future. Low and high estimates are the 5% and 95% tails of the distribution.

c. Air Quality Valuation Model Documentation (Stratus Consulting Inc. 1999) for Health Canada. Low, central, and high estimates are given respective probabilities of 33%, 34%, and 33%.

d. ExternE report, 1999. Uncertainty bounds are set by dividing (low) and multiplying (high) the mean by the geometric standard deviation (2).

The table shows quite close agreement on the size of the best or midpoint VSLs and VSCs (value of a statistical case of chronic respiratory disease). The differences that do exist may be explained partly by currency conversions and partly by researchers not always adjusting such values over time for inflation.[38] Also, the rank ordering of preferences noted above is found to be very similar across the studies, although not every study considers the same set of health endpoints. The low VSLs for TAF and AQVM result from adjustments to the VSL for age effects. ExternE takes the VSL and converts it to a value of a life-year for subsequent analysis. These efforts have yielded values ranging from $50,000 to $300,000 a life year.

In our judgment, this close agreement is the result of several factors, including replicability of findings in original studies in different locations (i.e., independent choices made by different research teams), and the consensus reached by research teams on a common pool of studies, results and interpretations. We believe that the social cost of electricity studies in the U.S. and the ExternE effort in Europe have something to do with this commonality (see Lee et al., 1995 and ExternE 1996; 1999). In addition, the Canadian studies have been informed by the AQVM model developed by Bob Rowe and others who have been active participants in the U.S. social costing debate as well (Hagler Bailly, 1995). Many studies in the U.S. pre-date and presage these efforts.

The ranges around these estimates are all somewhat different, seemingly without pattern. This result perhaps could be expected since there is no treatment of uncertainty that is universally accepted. The EPA mortality results are based on one standard deviation from the distribution (the Weibull) that best fit the mean WTP estimates from 26 studies. The Canada results are based on a representation of uncertainty as a three-point probability distribution, which includes expert judgment. The TAF distributions are Monte Carlo-based, assuming, unless otherwise indicated by the original studies, that errors about mean estimates are normally distributed, with variances given in the concentration-response and valuation studies relied upon for the underlying estimates. Bounds are defined as 5^{th} and 95^{th} percentile. Error bounds in the latest ExternE report are established as one half (low) and twice (high) the geometric mean.

[38] Note that these studies were published before the recent literature questioning the traditional estimates of the VSL.

The endpoints being valued are not all comparable to one another. The unit values for mortality risk, chronic lung disease risk, and acute symptoms all are derived from a WTP approach that may be thought of as capturing, however imperfectly, the full value to the individual of reducing the risk or the symptom. The other values are only partial, mainly relying on COI techniques. They are meant to capture the more severe manifestations of either acute events or chronic states and may, without proper adjustments, double count WTP benefits or provide significant underestimates of the WTP to reduce such effects. Indeed, it is fairly common practice to adjust such COI estimates by a factor to bring them up to a WTP estimate, so as to eliminate such underestimation. AQVM (Stratus Consulting Inc., 1999), for instance, recommends using a factor of 2-3 to make this adjustment. The evidentiary basis for the generality of this adjustment across endpoints is quite weak.

8.2 The Health Components of Environmental Valuation included in the CWS Process

The information used to value health changes in the CWS process is the information contained in the Air Quality Valuation Model (AQVM). This model has undergone extensive review in Canada, particularly as part of the recent process to set standards for sulfur in gasoline and diesel fuel. Table 28 shows that the benefit values for AQVM are not out of line with those appearing in other major efforts at cost-benefit analysis of alternative ambient air quality standards, i.e., the endpoints examined are quite typical of similar efforts around the world and the values used are generally within consensus ranges of values appearing in the current literature. The Panel notes that the consensus about this literature is changing and deficiencies within this literature as a whole are being more broadly and deeply recognized. New estimates addressing the issues of statistical life years lost and the adjustment of VSLs for health status and demographic differences will improve the valuation components of the CWS process. Because VSL or VSLY estimates dominate the overall benefit estimates, continued research on, and evaluation of, these estimates is critical. The expectation is that, in a new consensus, the values in the AQVM and elsewhere may need to be lowered, although how far is unclear and for which endpoints beyond mortality risk is unclear.

While the unit values and distributions used in the AQVM will have to be updated as the literature matures, it should be noted that the approaches used to evaluate health benefits are largely consistent with economic theory, are based on recent, but not the most recent, literature, and include estimates of variance or uncertainty.

8.3 Valuation of Non-Health Effects

A variety of non-health impacts of changes in PM and ozone (as well as other pollutants) on recreation, agriculture and forestry have been presented in the literature (Freeman, 1993) and in evaluations of the benefits of air quality improvement associated with the U.S. Clean Air Act (U.S. EPA, 1999a; 1999b; 1999c; 1999d). However, two issues arise when considering the benefits associated with non-health impacts. First, in most previous studies, the non-health component has comprised no more than 1% of the total benefits estimate. For example, in the U.S. prospective study, non-health benefits make up only 4% of the estimated total benefits of air quality regulation. Variation in mortality valuation is the major contributor to the variation in the total benefit estimates. Thus, even though there is significant potential to include non-health benefit estimates, the magnitude of these benefits may be relatively small. However, many highly uncertain areas are excluded from current benefits estimates / endpoints (e.g. ecosystem impacts) and thus additional research in these area may reveal a somewhat different pattern in non-health benefits in the future.

A second consideration is that there are relatively few Canadian studies of non-health benefits. Most of the non-health benefit estimates are transfers from U.S. studies. While "benefit transfer" is the only option in these cases, there are many concerns about the credibility of benefits transfers, and it would make transfer much easier to evaluate (and less needed) if there were more Canadian studies to evaluate the transfers upon.

8.4 The Non-Health Components of Environmental Valuation included in the CWS Process

The CWS for Particulate Matter and ozone includes only one non-health benefit category – household material soiling. This category of impact is very small, resulting in approximately 1% - 2% of the economic value of improved air quality in most of the scenarios.

The issues that arise surrounding the non-health components include:

1. Why were no other non-health components selected to be included in the analysis?
2. Why was household soiling selected?

Addressing issue 1, the main modeling tool for the CWS process is the Air Quality Valuation Model (AQVM) that contains non-health impact estimates for

- Visibility (change in visual range)
- Household Soiling
- Materials Damage
- Recreational Fishing
- Agricultural Crop Damages (for selected crops).

In comparison to the CWS process, the U.S. Retrospective and Prospective Studies of the U.S. Clean Air Act included consideration of

- Visibility (change in visual range)
- Household Soiling
- Materials Damage
- Recreational Fishing
- Agricultural Crop Damages (for selected crops).
- Forestry Losses
- Reduced Worked Productivity

While the policies being examined are quite different, and thus not comparable, it is illustrative to examine the magnitudes of value across categories, and the assessment of the various categories examined in the U.S. process. In the U.S. Study, household soiling was not included in the analysis of benefits because it was felt that the studies were old and unreliable. The impacts of particulate matter on visibility were included in the U.S.

study, as were impacts of ozone on agricultural and forestry yields. Impacts of SO_x and NO_x on recreational fishing were also considered.

In the CWS benefits analysis, recreational fishing was not included, even though the AQVM includes measures of recreational fishing benefits and those benefit measures contain relatively "state of the art" Canadian and U.S. empirical measures of recreational fishing activity and value. The CWS benefits analysis does not include agricultural impacts even though these impacts, and approaches to modeling these impacts, have been well documented in the Working Group on Air Quality Objectives and Guidelines (WGAQOG) studies (WGAQOG, 1997; 1999). Some agricultural impact measures are included in AQVM. Forestry impacts are not included in AQVM but the U.S. 812 study contained detailed analysis of forestry impacts arising from ozone emissions, including the use of the USDA Forest Service Timber Assessment Market Model (TAMM) that considers Canadian as well as U.S. timber producing regions (U.S. EPA, 1999e). Visibility is included in the U.S. study and the AQVM, however, there is some controversy over the visibility results (see Cropper letter to Browner, Oct 29, 1999 in U.S. EPA, 1999e) as they are based on somewhat dated research results, and they focus on visibility in national parks and recreational contexts. In terms of relative magnitudes within the U.S. study, visibility benefits produce the highest impacts associated with air quality improvements with worker productivity, forestry and agricultural impacts relatively similar in magnitude.

The results for Household Soiling that were included in the CWS process (and are part of AQVM) have been criticized in reviews of AQVM and elsewhere. These estimates are based on dated research results that do not consider various joint-production issues arising in the measurement of household soiling benefits. They also appear to be relatively minor in the overall analysis.

In summary, the process for identifying which elements to include in the CWS process and which to exclude is unclear. The decision to include household soiling, and not any of the other, better defined and better measured benefit categories, in the CWS process appears not to be based on the magnitude of the impacts, or on the assessed quality of the valuation information. There is room for improvement in the non-health benefit estimates, especially for forest impacts (including maple sugar production and other non-timber products) as well as consideration of improved assessments of visibility benefits. However, these issues are somewhat secondary to the development of a process for the determination of categories to include or exclude in the analysis.

8.5 Public and Stakeholder Concerns Regarding Valuation

A variety of concerns often accompany attempts to employ CBA in an environmental regulation context. Three major concerns that are often identified by stakeholder groups and the general public are presented below. Each concern is followed by the Panel's view on these issues.

1. VSL estimates appear to be very high relative to amounts that are spent on public programs to reduce risks to human life, or amounts that the public actually spends to reduce health risks. Studies of expenditures on public safety programs show that median costs per expected life saved are "low" ($40,000) relative to VSL estimates.

It is not at all surprising that public spending on risk reduction per life-year-saved is less than the value of this benefit to society. First, we note that it is incorrect to compare a VSL to a value of a life year saved. A VSL of $4.1 million translates, using the Viscusi and Moore approach, to a value per life year of about $250,000. Thus, the appropriate comparison *begins* as that between $250,000 and $40,000. Second, the median is an incorrect statistic for comparison and use in CBA. It should be the mean, which will be higher than $40,000 because of the common skewness of the distribution of values. Most importantly, the statistics used to calculate median expenditures on health risk reduction programs are based on decisions by public administrations and not the tradeoffs or willingness to pay by individual consumers. The wide variation in "cost per life saved" measures arises from inefficient allocation of program expenditures. Costs per expected life saved in some programs, for example, are in the billions of dollars while costs for others are measured in the thousands (Tengs et al. 1995). These figures illustrate that additional investment in lower cost programs would be more efficient, relative to the expensive programs. However, these are still opportunity cost measures, and they are not reflective of the economic value of risk reductions. Finally, just as the satisfaction we get from buying a car is typically more than the price, the satisfaction we get from improved health will be far more than its price. This follows from the operation of markets. Prices of cars (or life-saving interventions) are determined on the market based on the marginal willingness to pay, i.e., the WTP of the purchasers with the least willingness to pay. Others, who value such improvements more receive a windfall gain, just as those who would have been willing to pay more for a sports car are happy that the price is lower than their maximum WTP for it.

The nature of the risks with respect to many life saving interventions is quite different than that for environmental interventions. Many analysts expect that people are willing to pay more for reducing risks involuntarily borne and viewed as uncontrollable than those that dominate the analyses of costs per expected life saved. Finally, the Panel is persuaded that the $4.1 million VSL is likely higher than it should be if the advanced age and poor health state of those affected by air pollution is taken into account, as well as deficiencies in the literature underlying this estimate. Some idea of the extent of the overestimation can be obtained from a very recent study completed for Health Canada (Krupnick et al., 2000). This contingent valuation study of 930 individuals in Hamilton, Ontario found that the VSL for those over 70 – the relevant population whose mortality is believed to be affected directly by air pollution – was about $800,000 for a 5 in 10,000 annual risk reduction. The Panel recognizes that one study cannot serve as the sole basis for revising a more traditional and more widely accepted estimate, but this recent Canadian work does suggest a possible downward revision for the appropriate VSL.

2. The use of VSL measures generates very large aggregate values that are difficult to rationalize given the sizes of other health related programs. For example, the aggregate value of reducing PM to background levels appears to be very large relative to the entire health care program in Canada.

The Panel agrees that the standard damage function approach to estimating mortality risk reduction benefits (involving multiplying a dose-response coefficient by the VSL and the target population) results in large benefits. As noted above, it may indeed be the case that the VSLs are too large, or the dose-response coefficients are too large. In addition, however, people may very well have very strong preferences for improving their current life expectancy. However, the extrapolation of those preferences to the case of eliminating pollution (to background) - standard fare in press accounts - is not necessarily appropriate. It may very well be the case that the WTP for further increases in life expectancy diminishes as life expectancy increases and diminishes at an increasing rate, which would imply a declining VSL with larger reductions in pollution. Also, it is difficult to draw any conclusions about the willingness to pay for improved health care outcomes from examining the highly centralized Canadian health care system. It is this WTP compared to the WTP for health improvements from reduced pollution that is the relevant comparison.

3. The economic valuation results for certain components of morbidity values appear to reflect a “worst case” scenario. For example, the estimates of Chronic Bronchitis used in the CWS process appears to be based on more severe cases than the dose response function is based on.

The Panel does not agree with this comment. In the case of chronic bronchitis, the Viscusi et al. (1991) estimate *is* for a severe case of chronic bronchitis. But the Krupnick and Cropper (1992) study is used to correct for this because this study allows respondents with family members who have this disease to describe its severity and shows how different degrees of severity affect WTP. However, these studies provide a weak evidentiary basis for estimating benefits of reducing such cases. Both studies survey about 300 people, the former in a shopping mall in North Carolina, the latter, people obtained through a newspaper ad in the *Washington Post.* Acknowledging that these studies are the only ones available and that not relying on them risks the assignment of a zero value to this endpoint in a CBA, the Panel believes that public policy as important as the CWS should rest on a firmer empirical foundation.

8.6 Conclusions and Recommendations

The benefits of environmental improvements are denominated in dollars, for the sake of comparison with costs. But, money is only a metric to convert what economist's call "utility" or satisfaction into something more concrete. Thus, the value of improving health is obviously much more to an individual than saving on out-of-pocket expenses, time, work, or other tangible consequences of illness. It includes, most pointedly, the avoidance of pain and suffering and premature death (with all that goes along with this). Money is only a convenient way of expressing those preferences for avoiding these consequences.

As noted in Chapter 2, many people are uncomfortable about placing monetary values on health and life. The Panel appreciates these concerns, but notes that the monetary value is only a means to express preferences for different health outcomes, one of which is changes in the risk of death or in life expectancy.   This is completely different than placing a "value on human life." Because of the lack of public understanding about these issues, the Panel believes there is a need for better communications about the meaning of health (and environmental) benefits estimates. This involves communication from experts to the policy-makers, from policy-makers to decision-makers (politicians) and from politicians to the public.

Because it is so difficult to convert people's preferences into money when they can't express these preferences through market transactions, the valuation of health is a difficult empirical problem. Accordingly, the Panel believes there is a need for more research on empirical methods for health valuation and notes the efforts by Health Canada to fund research in this area and expects the responsible agencies to lead the way in incorporating the results of this and future research (assuming the research meets high professional standards) into the AQVM and regulatory analyses.

Non-health benefits were apparently excluded from the CWS because they were judged to be small relative to benefits of mortality reduction, but this assumption was predicated on the magnitude of the VSL, which might be too high, bringing the original assumption to ignore non-health benefits into question. In particular, ecological impacts have been ignored because of the lack of methods to predict or to value them, but they represent a substantial uncertainty and could be very large if nonuse values for vulnerable ecological resources could be reliably valued.

Finally, we note that the process of developing consensus and buy-in to analyses as complex as that of a CBA to underlie the CWS requires openness and transparency. We feel that government goals for the commercialization of policy models have hampered the goal of public acceptance of such models and the analyses based on them. In the future, effort should be placed on communications and increasing transparency of the process of CBA within the CWS.

8.7 Summary

Table 29 provides a summary of the Panel's assessment of the CWS approach to valuation of health and non-health benefits, including the key limitations, uncertainties and recommendations for alternative approaches.

Table 29: Summary of Panel's Assessment of CWS Approach to Valuation of Health and Non-Health Benefits

ISSUE	**VALUATION OF HEALTH BENEFITS**
CWS APPROACH	Use of AQVM; discount rate = 2%, 5%, 7.5%
PANEL CRITIQUE Key Limitations	No major limitations. At the time, represented consensus among economists on appropriate interpretation and treatment of literature except that almost all benefit measures are transfers from the US.
RELATIVE UNCERTAINTIES (Probably Minor, Potentially Major)[39]	Major uncertainties about the VSL because of benefits transfers involving the hedonic wage and accidental death studies to the air pollution context.
DIRECTION OF BIAS[40]	Probably biased upwards on net, but biases run in opposite directions.

[39] Likely Significance Relative to Key Uncertainties on Net Benefits Estimate: Probably minor (alternative assumption or approach could influence overall estimate by <20% difference), Potentially major (>20% difference). Adapted from US EPA study The Benefits and Costs of the Clean Air Act 1990 to 2010" Nov. 1990 study in which 5% difference was used see pg. 21, 33, 65, 79, 98.

[40] The U.S. EPA report "The Benefits and Costs of the Clean Air Act 1990 to 2010" Nov. 1999 used the following: Overestimate, Underestimate, Unable to determine based on current information

Table 29: (Cont'd)

RECOMMENDATION/ ALTERNATIVE INPUTS, TOOLS, APPROACHES	Maintain reliance on willingness to pay approach. AQVM needs to be updated regularly as new literature is produced and accepted. Alternative approaches could be used in sensitivity analyses.
ISSUE	**VALUATION OF NON-HEALTH BENEFITS**
CWS APPROACH	Household soiling only non-health endpoint assessed using AQVM.
PANEL CRITIQUE Key Limitations	Estimates for household soiling are based on dated research. Unclear process for identifying which non-health benefit categories to include in CWS CBA. Almost all benefit measures are transfers from the U.S. Limited Canadian information.
RELATIVE UNCERTAINTIES (Probably Minor, Potentially Major)	Ecosystem effects and values are highly uncertain and potentially large.
DIRECTION OF BIAS	Underestimate
RECOMMENDATION/ ALTERNATIVE INPUTS, TOOLS, APPROACHES	Update and improve AQVM with non-health benefits. Include non-health benefits in a systematic fashion. Research to improve Canadian components of valuation database and ecosystem valuation estimates.

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9 Policy Analysis and Decision-Making

Cost-benefit analysis of regulatory change is “required” under the November 1999 Government of Canada Regulatory Policy (Privy Council Office, 1999). The CWS approach is applied to meet these requirements. The conceptual foundations, assumptions and challenges of CBA have been outlined in Chapter 2. Ultimately, the analysis of costs and benefits of air pollution control is aimed at making a decision about the levels of control that are most *efficient* for society. However, efficiency is never the sole criteria for making major societal decisions. Equity, feasibility and legality are some of the other considerations that must always be considered in establishing public policy. In this section we compare CBA to other methods that have been proposed for assessing evidence for regulatory decision-making, including cost effectiveness analysis and multi-criteria analysis. We also discuss the issue of CBA as a “stopping rule” for standard setting, an issue recently being debated in the U.S. Court system.

Although scientific information forms the basis for CBA, this information is itself highly uncertain, making the CBA also highly uncertain. Rarely will regulatory analyses of complex environmental policy questions be definitive enough to dictate a specific course of action. Rather, there will usually be enough uncertainty that different policymakers will reach different conclusions based on how they interpret the evidence. This is reality. We may prefer to have analyses that are highly certain and definitive so that the analysis will make the tough choices for us. Realistically, we cannot expect that any analytical approach (cost-benefit, risk assessment, etc.) can be the primary driver in making a public policy decision without the application of considerable judgment to balance all of the other factors that must be considered.

Requiring the analysis to inform the decision-making process rather than dictating the decision reveals a need to understand the merits of the various approaches that could be used to analyze the problem. This requirement also demands that any analysis must be explicit about the uncertainties inherent in the analytical approach.

9.1 Standard Setting and Cost-Benefit Analysis

There are differing opinions about the use of CBA for standard setting. In the U.S., for example, certain environmental laws mandate that cost-benefit analysis be used, others are noncommittal, and others, most notably the Clean Air Act, appear to preclude its use in setting air quality standards. Since the passage of the Act in1970, the U.S. EPA, backed by Appellate Court decisions, has interpreted language in the Act to require the EPA to set its air quality standards only considering the criterion of protecting the public against adverse health effects with a margin of safety. Recently, however, the U.S. Supreme Court has taken up the question of whether Congress really meant for CBA to be ignored when setting air quality standards (U.S. Supreme Court, 2000).

According to those arguing for the use of CBA, the difficulty with approaches that ignore cost and benefit information is that they do not provide a framework for analyzing the advantages and disadvantages of alternative standards and, in particular, do not provide a "stopping rule" for deciding when the standard is too loose or too tight. CBA can provide this stopping rule, albeit with many uncertainties and caveats. In the recent submission to the Supreme Court of the United States, a group of eminent economists and public policy scholars identified the important advantages of the use of a cost-benefit framework, including better resource allocation and an organized comparison of favorable and unfavorable effects of a proposed policy. The document points out that policy makers should not be bound by CBA results, but that measures of benefits and costs should be fundamental to regulatory analysis (U.S. Supreme Court, 2000).

The debate in the U.S. provides some insight for Canadian regulatory analysis. While CBA will not generate "answers" that prescribe outcomes, employing a cost-benefit framework, including analysis of distributional issues, should provide a mechanism for regulatory analysis that is transparent and allows all relevant information to be considered in the analysis. CBA should also provide a way of identifying a stopping rule for setting standards. Not that there would be slavish adherence to the results of such an analysis; only that it would provide guideposts to standard-setting that have a more firm foundation than that based on standard practice. The various "alternatives" to cost-benefit analysis presented in this chapter are consistent with a cost-benefit framework within which all factors, including distributional factors, are incorporated into the analysis. Thus, the methods outlined below should be considered complementary rather than competing approaches for developing environmental regulations.

9.2 Cost-Benefit Analysis and Alternative Policy Analysis Approaches

In addition to CBA a variety of options have been proposed and considered for conducting an analysis for deciding on environmental standards (U.K. Department of Health Ad Hoc Group, 1999), including cost effectiveness analysis and multi-attribute utility analysis. How these different approaches could be applied to the issue of air quality standards and health can be illustrated by considering Table 30, which presents a stylized environmental policy analysis case. The physical (health science) evidence regarding life-years saved and reduced COPD events associated with the revised regulation are presented, along with the monetary direct cost (to industry) associated with the new regulation. This breaks down the decision into the attributes associated with the regulatory change. (Note that other attributes could also be considered, including who is affected, or equity considerations, and other health and economic impacts).

Table 30: Simplified Example of Multi-attribute Analysis

ATTRIBUTE	CURRENT STANDARD	NEW STANDARD
Life Years Saved (relative to current standard)	-	1000
COPD events reduced (relative to current standard)	-	5000
Direct Cost to Industry (relative to current standard)	-	$4B
Employment in Sector	200,000	190,000

Life Years Saved or some estimate of fatalities-avoided is typically the result of the health risk assessment that predicts the level of fatalities under the current standard and those anticipated under the new standard to allow a difference to be calculated. Likewise, some measure of morbidity, like COPD events reduced, can be estimated for each scenario. Finally, direct cost to industry can be estimated by an economic analysis of changes in industry costs associated with meeting the new standard. In this characterization of the issue, the direct cost is measured in monetary terms, while non-monetary measures of the health benefits are presented. Thus the uncertainty associated with estimating the monetary value of health benefits is avoided. This presents the benefits and costs in their own inherent units and allows one to make a judgment as to whether the change to a new standard is *worth it*. The policy maker's challenge is to examine this information and make a judgment for the overall well-being of the public.

CBA attempts to provide information on the overall public well-being associated with this policy by constructing monetary measures of the benefits (health impacts) and costs (direct industry costs) and thus making both measurable in the same units. The uncertainty in these benefit and costs estimates, and the perception that important attributes of a decision are being *left out* often lead to calls for alternative forms of analysis. As we shall see, there are advantages and disadvantages in these other forms of analysis.

Multi-attribute or multi-criteria analysis is a method of evaluating trade-offs over various attributes of a situation, like the policy issue presented above. If one is uncertain about the monetary valuation of health estimates, for example, one could simply present members of *the public* with the table of changes in health states (small changes in life expectancy) and cost to industry associated with the regulatory change. *The public* could be asked if they would accept the new regulations (and the cost) or refuse the new regulations. Of course, such questions are fraught with difficulty (e.g. Mitchell and Carson, 1989). This is essentially multi-attribute decision analysis where the population providing the trade off information is *the public*. This trade-off question could also be structured as a referendum where people vote on whether to accept the new regulations or continue with the old ones.

An alternative is to simply present the table above in a form of multi-attribute analysis to *decision-makers* and ask them to make the choice. Then the multi-attribute analysis focuses on the *decision-makers* and hopes that they represent the *public*. While benefit-cost analysts go to great lengths to capture the tradeoff information as if the public were allowed to make the choice, the *decision-maker* model assumes that the decision-makers (e.g. politicians) can reflect the views of the public in the way they make choices from policy alternatives. Naturally, this is a very controversial issue.

A third option with multi-attribute analysis is to use *stakeholder* groups, but the challenge in choosing stakeholders in such a case, and the possibility of stakeholders entering the process with well-entrenched positions also makes this option difficult to implement.

CBA and multi-attribute analysis essentially seek valuations of the attributes or measures of the tradeoffs associated with the policy question. If the decision-makers choose the new standards in a multi-attribute setting, they are implicitly valuing the attributes associated with the new standards higher than the attributes of the current standards. The cost-benefit approach tries to assess the "values" from individual decisions aggregated over the population instead of relying on trade off analyses conducted by decision makers or stakeholder groups. Clearly, both approaches have their challenges but multi-attribute analysis can examine a richer set of attributes (equity, legal, etc.) than CBA.

No approach presents a panacea for decision analysis. In all cases the assumptions of the approaches must be carefully evaluated, and considerations beyond efficiency that are necessary in public policy must be included in the policy evaluation. However, a carefully structured cost-benefit analysis provides the information for a multi-criteria analysis and it provides information on the incidence of costs and benefits. Therefore, a well-conducted CBA can form a strong groundwork for environmental policy analysis and decision-making.

9.3 Cost Effectiveness Analysis

One can also use *cost-effectiveness* approaches in which one decides on some *acceptable* degree of risk, and then finds the least costly way to achieve that level of risk. Quality-adjusted life year calculations (QALYs) can be used in a cost-effectiveness approach. QALYs are a composite measure of the number of years of life gained or lost by a particular decision, but weighted according to the expected quality of life during those years, and to this added measures of the improvement in quality of life (say from reduced morbidity). Years of poor health are weighted as a fraction of years of good health. QALYs provide a metric of preferences over alternative health states that allows one to determine if procedure A is more effective at meeting a chosen standard than procedure B. However, one still needs to determine the target level of risk, that the cost effectiveness analysis is based on. So, cost-effectiveness analysis presumes that we know the level we are aiming for, when in fact the debate usually includes determining what that level should be.

Recently QALYs have been used in a form of cost-benefit analysis where the QALYs are assigned a monetary value ($50,000 per QALY is a consensus conversion factor in the medical community, where medical procedures costing more than this would be rejected as not *worth it* (Carrothers et al., 1999). However, this monetary amount is *ad hoc (*other analysts use $100,000, for instance, an equally *ad hoc* estimate) and is not based on individual preferences. Thus this approach is not consistent with the principles of cost-benefit analysis that were outlined previously in this report.

A comparison of these three different approaches on a variety of dimensions is summarized in Table 31. None of these approaches provide a perfect solution to informing the decision under substantial uncertainty. Likely the best solution is some combination or cross-check among these differing approaches.

Table 31: Comparison of approaches to policy analysis (based on U.K. Department of Health Ad Hoc Group 1999)

CRITERION	CBA MONETARY VALUATION	COST-EFFECTIVENESS	MULTI-CRITERIA ANALYSIS
Different health outcomes on similar scale	monetary scale	standardized health outcome measure (e.g. quality /duration of life)	abstract scale of scores based on choices between alternatives
Ready comparison of costs and benefits	costs and benefits both in monetary units for easy comparison	can compare quality and duration of life per unit cost but cannot determine if benefits exceed costs	could express costs and benefits as scores but cannot determine if benefits exceed costs in resource terms
Compatible with techniques usually used in health or environment	not generally used in health services but standard for many environmental and public health issues	measurement of health gain in terms of QALYs increasingly used in health services. Other forms of cost-effectiveness used in environmental policies	currently not widely used in either health or environment policy context, but growing use in both areas. Can use with CBA or CEA.
Takes account of individuals' views	conceptually based on individual views but resultant average values often applied in empirical analysis	quality of life states are scored by individuals, but other aspects (i.e. dread of particular diseases) not taken into account	often uses views of experts, stakeholders, or policy makers rather than lay people. If only experts used, may not reflect wider views.
Takes account of views of society (e.g. equity), (sometimes overrides individual views)	focuses on efficiency but the results can be used to describe the equity implications of alternative actions	loss of quality of life treated the same regardless of age or type of disease	could include equity as a criterion / attribute
Approach reasonably well developed	yes	yes	yes – although not as well developed as the other methods presented here

9.4 Beyond Efficiency: Distributional Issues

CBA and most forms of economic analysis focus on the efficiency aspects of the regulatory change. CBA can also provide information about "who" is affected and to what degree specific sectors or groups of individuals are disproportionately affected by a regulatory change. However, CBA cannot identify the weight that the public (or policy makers) place on these distributional issues.

There are several key distributional issues in the CWS process. It is widely recognized that the health improvements of setting tighter standards will disproportionately benefit the elderly and those who are already somewhat compromised in their respiratory or cardiovascular function. In addition, there are regions in the country where the health benefits will be more significant because of the concentration of population, and the current emission levels. The impact of regulatory change on different income groups or socio-demographic groups is uncertain, but may also be an element in the policy analysis.

Impacts on communities, or on specific sectors of the economy could also be considered. For example, non-health impacts on agriculture may be relatively small (in comparison with the health and mortality impacts) but may be very important to the sector and to the communities that are supported by the sector. Costs of regulatory change may also be focused on a few industries, and may also have significant impacts on communities if the regulatory change results in plant closures or changes in economic structure. While these impacts could be measured using well-constructed analysis methods, the weight that these sector specific or community impacts should take in the overall policy analysis is difficult to determine, and requires some form of multi attribute decision analysis. In addition, such information may be very useful in strategies for implementation of regulatory change.

9.5 Conclusions

- CBA attempts to aggregate individual (or household) valuations / tradeoffs to construct an aggregate assessment of whether changing regulations is "worth it", and thus examines the *efficiency* of changing regulations (but does not address equity and other elements of policy analysis).

- Multi-criteria analysis allows a broader set of elements to be considered including equity issues and other elements not included in CBA. *Valuation* of health effects, for example, is not required within a multi-criteria analysis as the method can be used with non-monetary measures of the attributes. However, *values* are implicit in the outcome of the multi-criteria analysis. The key challenges in multi-criteria analysis include who chooses, how the choices are presented / structured and how the information is presented / structured.

- Cost effectiveness analysis doesn't address the problem of setting air quality standards because the environmental quality target is assumed to be known, but is a useful tool once environmental quality targets are defined.

- Distributional issues, including impacts on specific groups of people, industries, or generations, must be considered in regulatory analysis. Such analysis is complementary to a good CBA and can be included in an MAA.

9.6 Recommendations

- Continued development of communication regarding alternative decision-making frameworks, including multi-attribute methods as methods to "triangulate" with traditional cost-benefit analysis.

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10 Conclusions

The task of the Expert Panel was to provide an independent, expert review and critique of the socio-economic (SEA) analyses – in this case a cost-benefit analysis (CBA) -- conducted in developing the Canada-Wide Standards on PM and ozone. Through a review of the models and associated data and assumptions used in the analyses, the Panel was asked to produce a report to address the following questions:

a. What are the strengths, merits, limitations, gaps and the degree of uncertainties of the proposed approaches, models, and their inputs and outputs?

b. By what means could the models and analytical approaches be improved, so as to minimize uncertainties and maximize the relevance, reliability and utility of outputs?

c. What other approaches and/or tools could be used to conduct these analyses?

The Panel draws the following conclusions from its assessment of the components of the CBA undertaken for the PM and ozone CWS decision-making process:

<u>Estimating Air Quality Changes from Emission Reductions</u>

1. The CWS study adopted a statistical approach using linear assumptions to determine air quality changes associated with reduction in emissions of PM_{10}, $PM_{2.5}$, SO_2, NO_x and VOCs. There are inherent problems that occur when trying to estimate the response of an inherently non-linear system by external estimation rather than by internal scaling. A number of factors may interfere with a linear correspondence between emission reduction and air quality improvements. These include factors such as the relative contributions of controllable emission sources to ambient air quality; ambient air quality contributions from trans-border sources; and non-uniform geographic distribution of emission sources. In the case of sulphur emission from

fuel, the assumption of a linear response of the system is reasonable since the putative changes in air quality are quite small (in the few percent range). However in the CWS study the changes envisaged are much larger. The assumption that a local change in emissions will result in a local change in gas phase is reasonable for CO and perhaps PM_{10}. Neither secondary $PM_{2.5}$ nor ozone fit into this “reasonable” category and the uncertainty is quite high as the sign of the change could possibly be incorrect in some cases.

2. The primary and secondary source aspects clearly point to varying strategies that may be much more effective than across-the-board reductions. For example, targeting reduction strategies for SO_2 at point sources and utilities may be far more effective at achieving low ambient levels of $PM_{2.5}$ than requiring primary removal of $PM_{2.5}$ across-the-board. Likewise, the value of focusing solely on anthropogenic sources of $PM_{2.5}$, both direct and secondary, is questionable when such a large fraction of emissions is from forest fires and open sources. In addition, a fundamental requirement for CBA and comprehensive modeling is an emissions inventory for the species of interest. The Canadian emission database is quite uncertain, particularly in total amount and the spatial and temporal distribution of emissions

<u>Estimation of Health Effects</u>

1. The CWS gave greater weight (2/3) to mortality derived from daily time-series data than to the mortality impact derived from cohort studies of annual mortality (1/3). The annual mortality data should be used as the primary basis for determining the mortality impact because they include not only the impacts of peak daily exposures, but also the cumulative effects attributable to baseline exposures over other time scales. The Pope et al. (1995) cohort study provides the firmest C-R parameter for the annual mortality impact because of the size of the cohort and the large number of North American communities. However, the C-R parameter from this study of largely middle class volunteers very likely is an underestimate when applied to the overall population. The HEI (2000) reanalysis of this study demonstrated that, within this cohort, the effect was larger for those with lesser educational attainment. Thus, it is

reasonable to conclude that a more representative population than used in the Pope study would have a greater coefficient of response.

2. The accumulating evidence towards a broad acceptance of causality for a range of cardiopulmonary effects from fine particulates appears destined towards widespread acceptance as a prudent public health judgment.

3. The evidence for mortality causality is more convincing for finer particulate (i.e. $PM_{2.5}$) than for coarser particulates.

4. The CWS health benefits analysis has taken adequate steps to avoid overstating the ozone health benefits due to colinearity with PM.

5. The database for fine particulate matter across the country is limited and more air quality monitoring data focused on fine particulate would provide a better basis for adjusting future air quality standards.

<u>Estimation of Avoided Non-Health Impacts</u>

1. The process for identifying which non-health elements to include in the CWS process and which to exclude is unclear. The decision to include household soiling, and not any of the other, better defined and better measured benefit categories, in the CWS process appears not to be based on the magnitude of the impacts, or on the assessed quality of the valuation information.

2. There is room for improvement in the non-health effect estimates, especially for forest impacts (including maple sugar production and other non-timber products) as well as consideration of improved assessments of visibility improvements. However, these issues are somewhat secondary to the development of a process for the determination of categories to include or exclude in the analysis.

Cost Analysis

1. The CWS approach to regulatory cost analysis summarizes a significant amount of information on control technologies, costs, and methods for attaining emissions reduction targets. It is based on direct control costs, an approach that has its limitations if, as we expect, there are general equilibrium impacts on the economy. However, we also recognize the significant effort that is required to capture these economy-wide impacts and suggest that this is a long-term research issue. The analytical approach makes many simplifying assumptions, as do all practical approaches to policy analysis.

2. The tax interaction effect is not included in the CWS analysis of costs. Initial estimates of the magnitude of the tax interaction effect are substantial and suggest that social costs may exceed direct costs by 25% or more. If the tax interactions effects are as significant as they appear to be in the recent literature it is likely that costs are underestimated.

Valuation of Health Benefits

1. Benefit values contained in the Air Quality Valuation Model are not out of line with those appearing in major efforts at cost-benefit analysis of alternative ambient air quality standards, i.e., the endpoints examined are quite typical of similar efforts around the world and the values used are generally within consensus ranges of values appearing in the current literature. The Panel notes that consensus in the literature is changing and deficiencies within this literature as a whole are being more broadly and deeply recognized. New estimates addressing the issues of statistical life years lost and the adjustment of VSLs for health status and demographic differences will improve the valuation components of the CWS. The expectation is that in a new consensus, the values in the AQVM and elsewhere may need to be lowered, although how far is unclear and for which endpoints beyond mortality risk is unclear.

Valuation of Non-Health Benefits

1. Non-health benefits were apparently excluded from the CWS because they were judged to be small relative to benefits of mortality reduction, but this assumption was predicated on the magnitude of the VSL, which might be too high, bringing the original assumption to ignore non-health benefits into question. In particular, ecological impacts have been ignored because of the lack of methods to predict or to value them, but they represent a substantial uncertainty and could be very large if nonuse values for vulnerable ecological resources could be reliably valued.

Policy Analysis and Decision-making

1. Many limitations of the CWS approach to cost estimation have been identified when held against the benchmark of the U.S. Prospective Study (U.S. EPA, 1997), or the U.S. Retrospective Study (U.S. EPA, 1999). This is a very high benchmark, but the CWS ambient air quality standards for ozone and PM are likely to be the most expensive single environmental standards to meet in Canadian history. As such these CWS deserve thorough treatment. Fortunately, some elements of the cost analysis can be improved at lower cost and with less effort than others. Extensions of cost analysis to include general equilibrium and international trade considerations can provide important information for policy analysis. The scale of the analysis (national including direct and general equilibrium effects; international including trade effects, etc.) is an important element to consider and will also help identify the impacts of the regulatory proposal, in terms of benefits and costs as well as the incidence of the impacts.

11 Recommendations

The Panel offers the following recommendations to improve the rigour and credibility of socio-economic analysis as an input to decision-making on Canada-Wide Standards for PM and ozone.

<u>Estimation of Air Quality Changes From Emissions Reductions</u>

1. **The Panel recommends that future CWS studies have the resources to include an appropriate and transparent definition of the baseline with reasonable estimation of the relevant components.** Definition of the baseline is essential in a CBA study. The baseline may change because of factors such as the implementation of current or future regulations, changing economic conditions, and possible changes in atmospheric climate.

2. **The Panel recommends that a more systematic continuous measuring program be adopted for PM_{10} and $PM_{2.5}$.** It is still not evident if extreme or chronic events with respect to high PM and ozone levels are important in causing health impacts and there are insufficient PM_{10} and $PM_{2.5}$ continuous measurements to address this question. Also, measurements of PM_{10} and $PM_{2.5}$ are critical for the evaluation of emission inventories and 3D physical-based modeling. Furthermore, it will be necessary to have adequate measurements to ensure both the efficacy of the reductions and compliance with the reductions.

3. **The Panel recommends that adequate resources and administrative structures be provided at the federal and provincial level for improving the spatial and temporal resolution of emission inventories of PM_{10}, $PM_{2.5}$ and ozone precursor species across Canada. NH_3 should be added to emission inventory studies.** One of the aspects that pervades all aspects of the CWS study is the requirement for an accurate emission inventory, with good spatial and temporal characteristics: these are necessary for both CBA and physical-based modeling. This will require the active collaboration or federal and provincial governments and the industrial sector with involvement of NGOs. This could involve support from a consortium of many levels

of government, from the federal to the municipal, industry and NGOs. We note that the emission inventory work that is proceeding in the Greater Vancouver Regional District provides an example to the rest of the country. Furthermore, given the importance of NH_3 in the formation of secondary $PM_{2.5}$ and the lack of an adequate baseline inventory, the Panel recommends that NH_3 should be added to emission inventory studies.

4. **The Panel recommends support for the on-going work on comprehensive or integrated 3D physical-based aerosol modeling in Canada that includes both ozone and PM chemistry and meteorology and its use for estimating ambient air quality changes with targeted reductions.** One means of attacking the problem relating reduction of emissions and the attainment of CWS is to use physical-based 3D models with both gas phase and aerosol formation and chemistry. Use of such models also allows a more detailed or targeted approach to be taken to infer impacts. This work is currently on-going in Canada.

5. **The Panel recommends that every effort should be made to develop Canadian emissions data.** Source-receptor statistical modeling potentially represents a powerful method of identifying emission sources, but this requires a detailed chemical knowledge of the emitted pollutants. This is rarely available in Canada and many studies have had to use surrogates from the U.S.

Estimation of Avoided Health Effects

1. **The Panel recommends that C-R functions for determining annual mortality risks and benefits associated with reductions in PM_{10} and $PM_{2.5}$ in AQVM be based on the prospective cohort analyses by Pope et al. (1995), Dockery et al. (1993) and Abbey et al. (1999). The central C-R parameter should be taken from Pope et al. (1995), the low from the Abbey et al. (1999) study and the high from Dockery et al. (1993).**

2. **The Panel recommends that mortality benefits estimation should be more heavily weighted towards the exposure-response relationship assessed for $PM_{2.5}$ rather than PM_{10}.**

3. **The Panel recommends that controlled human exposure studies be conducted using concentrated ambient particles and mixtures with other ambient pollutants to explore cardiopulmonary endpoints. The Panel recognizes however, that there are challenges in providing realistic exposure conditions for human toxicology experiments that will satisfy research ethics review boards. Chamber studies should be complemented with more field studies including individuals at greater risk who can not participate, for ethical reasons, in exposure chamber studies.**

<u>Estimation of Avoided Non-Health Impacts</u>

1. **The Panel recommends that the approach to selection of non-health endpoints for inclusion in an assessment of non-health benefits be done in a systematic fashion.**

2. **The Panel recommends that future assessments of non-health benefits for the CWS for ozone include an assessment of improved agricultural productivity.**

<u>Cost Estimation</u>

The Panel recommends that the CWS cost estimation be improved by taking the following relatively low cost steps:

1. **Improved consideration of Canadian industry and source emission categories (SIC and SCC combined) and treatment options, to the plant level including "ground-truthing" of control costs.**
2. **Consideration of the likely pollution intensity and marginal product of new technologies (both production and abatement).**
3. **Assessment of existing emissions control implementation.**

4. **Consideration of non-technical approaches to emissions reduction (fuel switching).**
5. **Consideration of co-benefits or multiple pollutant reductions with individual technologies.**
6. **Careful consideration of the baseline and explicit description of the assumptions involved in the baseline. The development of the baseline may include the consideration of alternative regulatory approaches including incentive approaches for emission reduction.**
7. **Increased transparency in the modeling of direct costs.**
8. **Assess the degree of uncertainty in the cost estimates.**

The Panel believes that the AERCo$t model can address some of these issues. Elements that will require substantial additional resources and research include:

1. **Improvement of the RDIS database for the basis for cost analysis to a level comparable to the current U.S. inventory.**
2. **Assessment of the degree to which partial or general equilibrium methods should be applied to regulatory policy. The development of general equilibrium models can be a costly exercise, and they carry a set of assumptions that must also be evaluated carefully, however, in many cases these models represent the best available technology for assessment of economy wide impacts of regulatory change. The U.S. Retrospective study, for example, chose to employ the Jorgenson-Wilcoxen dynamic general equilibrium model of the U.S. economy (Jorgenson and Wilcoxen (1990b)).**
3. **Research on the tax interaction effect, in a Canadian context.**

Valuation of Health and Non-Health Benefits

1. **The Panel believes there is a need for better communications about the meaning of health (and environmental) benefits estimates. This involves communication from experts to the policy-makers, from policy-makers to decision-makers**

(politicians) and from politicians to the public. Monetary value is only a means to express preferences for different health outcomes, one of which is changes in the risk of death or in life expectancy. This is completely different than placing a "value on human life." There is a lack of public understanding about these issues.

2. **The valuation of health is a difficult empirical problem because it is so difficult to convert people's preferences into money when they can't express these preferences through market transactions. The Panel believes there is a need for more research on empirical methods for health valuation and notes the efforts by Health Canada to fund research in this area. The Panel expects the responsible agencies to lead the way in incorporating the results of this research (assuming the research meets high professional standards) into the AQVM and regulatory analyses.**

3. **The Panel concludes that government goals for the commercialization of policy models has hampered the goal of public acceptance of such models and the analyses based on them. In the future, effort should be placed on communications and increasing transparency of the process of CBA within the CWS.** The process of developing consensus and buy-in to analyses as complex as that of a CBA to underlie the CWS requires openness and transparency.

<u>Policy Analysis and Decision-making</u>

The Panel endorses the use of a cost-benefit framework for the analysis of environmental regulation that includes an accurate assessment of the costs of regulatory change. The Panel recognizes the empirical limitations of CBA and recommends the following:

1. **Continued development of methods for accurate assessment of costs and benefits, including methods for the analysis of general equilibrium (including tax interaction) effects and international trade impacts of regulatory change.**
2. **Continued development of communication regarding alternative decision-making frameworks, including multi-attribute methods as methods to "triangulate" with traditional cost-benefit analysis**

3. **Investments in human capital in the area of CBA of environmental regulation so that policy makers and the Canadian public can be confident that cost and benefit measures accurately reflect Canadian values and preferences and Canadian institutional arrangements.**

APPENDICES

Appendix A: Frequently Raised Concerns About Cost Benefit Analysis[42]

Many of the critiques of CBA encountered in everyday policy debates are echoes of the more conceptual issues that we address here (Kopp, Krupnick and Toman, 1997). Criticisms include the following:

(i) The environment is a public good that is not exchanged in markets and therefore defies economic valuation. Thus, the use of CBA to evaluate environmental policies is inappropriate.

(ii) Environmental protection is often desirable for reasons that cannot be quantified--social, spiritual, and psychologic values that defy monetization.

(iii) CBA does not take the “rights” of future generations into account.

(iv) Economic benefit measures are hypothetical measures of benefits and are not actual benefits that can be measured in terms of savings in health case costs or other “real” benefits.

Criticisms of CBA focus on several overlapping points: the notion that preference satisfaction gives rise to individual well-being, the elements of the individual social-welfare index, the notion that economic value is a measure of preference satisfaction, the empirical and philosophic problems encountered in quantifying economic value, the presumption that the well-being of society can be defined as some aggregation of the well-being of individual members of that society, and the methods by which the aggregation is performed. In the following section, we discuss each of those criticisms more fully. As indicated below, the response of CBA analysts to the criticisms is that CBA is largely an attempt to measure preferences formally. Legitimate questions can be raised about the practice of such measurement or the method of aggregation to describe social welfare. In contrast, we argue that the basic criticisms of the preference satisfaction concept are less persuasive.

[42] Extracted from Kopp, Krupnick and Toman (1997)

1. Is Individual Preference Satisfaction an Appropriate Method for Judging Social Welfare?

We noted in above that "preference satisfaction" forms the philosophic foundation for CBA. We can greatly simplify the discussion of the limitations of CBA by prefacing our remarks with a brief discussion of instances in which society consciously chooses to make satisfaction of individual preferences subservient to higher-order social determinations. For example, it may be one's preference to drive while intoxicated, but society has determined (in a political process) that such behavior will not be permitted. The point is that society can choose to make preference satisfaction subservient to particular and explicit social determinations without undermining the intellectual integrity of CBA.[43] However, there might be other circumstances in which CBA of social determinations is useful in helping to decide whether the social structures need to change. For example, blanket prohibitions on exposure to potentially hazardous substances might deliver relatively little benefit compared with their costs, particularly as the technologies for detecting very low levels of contamination improve.

2. Equity Considerations

It is often argued that CBA takes the existing distribution of income as given and does not consider the equity implications of the policies that it seeks to evaluate. In terms of the six criticisms of CBA noted earlier, this criticism points to the anonymous manner in which the welfare changes of individuals are aggregated to obtain estimates of the change in social welfare.

The criticism is valid as far as it goes. Anonymous weighting of individual welfare does not take equity into account. However, that need not be the case (Burtraw and Kopp, 1994; Slesnick 1999). Because one can weight in any number of ways, the problem is that someone must state explicitly what the weights should be. Inasmuch as there is no established "right" to equity in the distribution of individual well-being, where would a policy-maker get the needed weights? She might decide to use her own weights, but the

[43] Laws that bar discrimination are other obvious examples of instances where the preferences of some have been over-ridden by the political decisions of society as a whole.

transparency of the CBA method would reveal them immediately, and those who disagreed could easily counter with their own weights. No equity weights have been sanctified through some political process, and anonymous aggregation has become the default in CBA. It has no claim to moral superiority or scorn. Even with this approach, however, more disaggregated CBA can provide important information about the incidence of effects.

3. Preference Satisfaction

CBA is meant to convey some normative information to decision- makers, namely, whether a policy could make the society better off than the status quo. The normative character of CBA is derived from the assumption that the satisfaction of individual preferences gives rise to individual well-being and that social well-being is a function of individual well-being. The preference satisfaction assumption is crucial to the normative properties of CBA, but one can do little to establish the validity of the assumption.

The root of the disagreement regarding the use of individual preferences is the determination of what would make up an index of social welfare or aggregate well being. An index serves to aggregate elements of a list into a single value. In the simplest case, which will suffice here, aggregation to a measure of individual satisfaction is accomplished by weighting the elements and summing. But where do the weights come from? In welfare economics, the weights are derived from the economic values obtained from the observed choices of individuals, which economists attribute to underlying preferences.[44]

Accepting the proposition that economic value is linked to the intensity of individual preferences and that choices based on preferences permit one to infer economic values does not imply that it is simple to infer these values. The problem of measuring values is most severe for tangible and intangible items that are not traded on organized markets,

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[44] The welfare economist Harsanyi states the economic view most directly, "The principle that, in deciding what is good and what is bad for a given individual, the ultimate criterion can only be his own wants and his own preferences" (1955). The "Principle of Autonomy" that Harsanyi articulates does not depend on the reasons one has for particular preferences. What matters for Harsanyi is that individuals apply the weights and the weights are permitted to be specific to each individual.

where one can observe the tradeoffs faced by individual and the choices they make, as discussed further below.

4. Elements of the Individual Social-Welfare Index

Two criticisms of the individual welfare indexes used in CBA bear on the elements that make up the index. The first has independent standing even if one accepts preference satisfaction. It argues that many preference-based factors can be influenced by a policy and that CBA includes only a subset of them as elements of the individual welfare index. That is a valid concern. For reasons of time, budget, tractability, and available information, some preference-based factors that might be affected by a policy might be left out of the index. To the extent that that happens and to the extent that the excluded factors are heavily affected by the policy and have high economic value (a large weight in the index), the results of the CBA will be affected in an unknown direction. How one can deal with this possibility is discussed below when we address implementation issues.

Like the first criticism, the second is logically valid even if one accepts preference satisfaction. It acknowledges that preferences are linked to individual well-being but claims that there is more to well-being than preferences. Naturally, if one defines preferences in such a narrow way as to exclude important attributes that affect well-being, this argument has some force. For example, if one were to limit preferences in the manner of simple models of "egoism,"[45] important aspects of well-being could well be left out. Another example of such a limitation in CBA is the exclusion of what economists call "nonuse" values implied in S. 343 (the Comprehensive Regulatory Reform Act of 1995).[46] However, it can equally be argued that these limitations are entirely arbitrary and the concept of preferences is rich enough to encompass all facets of life that give rise to well-being. Thus, the importance of this argument seems to rest on how one chooses to define preferences and on whether one can identify factors other than preferences that affect well-being.

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[45] Models of "egoism" generally restrict preference to those things that benefit the individual directly. Thus a "preference" for self-sacrifice in the attainment of some worthy goal, for example, would be excluded.

[46] See the report of the Senate Judiciary Committee on S.343, May 25, 1995, page 59.

One such class of factors often mentioned is categorized as "ethical" considerations, including fairness to future generations or the integrity of conduct within the current generation in maintaining "critical" environmental resources. Some philosophers, such as Bryan Norton (1994), maintain this view. Others strongly dispute that ethical considerations are not a reflection of preferences, given a broad-enough conception of preferences, and that the dispute is one of data and measurement rather than basic concept (Kopp, 1992).

5. Economic Value Is Not a Measure of Preference Satisfaction

The criticism here is relatively straightforward--that the economic value of some thing is not related to the well-being that a person enjoys as a result of that thing. For example, this argument implies that if one is willing to pay $3.00 for a bottle of imported beer and only $1.50 for a bottle of domestic beer, it is not possible to say that the person's well-being is greater if he or she is given an imported beer than it would be if he or she given a domestic beer.

For this argument to hold, it seems that one must assume that actions (choices) are not motivated by preferences or that people cannot make choices that reflect their preferences.[47] We have already addressed this argument above.

6. Economic Value of Some Things Cannot Be Measured

It is argued by some that there are things that humans cannot put a price tag on.[48] Aspects of the environment often fall into this category. That might well be true, but it does not imply that individuals cannot determine how important aspects of the environment are to them. As above, economic values are inferred from the choices made by individuals. It would be wrong to think of economic values as dollar-denominated numbers in one's brain to be downloaded when a person is asked the worth of a beautiful ocean sunset; rather, such a value might be inferred from the things that one gives up to

[47] If the argument is that preferences are not linked to well-being and therefore economic value is not linked to well-being, one is restating the preference satisfaction critique.

[48] A corollary to this statement is that there are some things that <u>should not</u> have a price tag placed on them.

see the sunset (e.g., the cost of travel to the ocean).[49] To economists, the importance of things (tangible or intangible) is revealed by what a person will give up to obtain them. The lower bound on the value of the item obtained is equated to what was given up. If the thing given up was money, the value can be expressed in monetary units; otherwise, it is expressed in the natural units of the thing given up.

This discussion also addresses the concern that benefit measures are not "real" measures of benefit since individuals do not actually have to pay or the implementation of the policy would not actually increase financial wealth or reduce costs. CBA focuses on social welfare and not only on private, market related, benefits and costs. Thus the measures of welfare are grounded in the amount that an individual would be willing to pay or tradeoff for a particular improvement in quality. In this sense a policy may not result in a change in financial transactions or a change in GDP, but it may significantly enhance social welfare. This is because many things that are considered components of welfare are not priced in the market and are not measured in GDP calculations, nevertheless, individuals are willing to make significant tradeoffs to retain or enhance these things. More specifically to issues surrounding air quality regulatory reform, increases in air quality may reduce monetary costs associated with medical care, all else remaining constant, however, reduced medical costs reflect only some of the benefits arising from improved air quality.

7. The Well-Being of Society Is Not Necessarily an Aggregation of Individual Well-Being

In the 18th century, economists seeking to avoid issues of interpersonal comparisons of well-being put forth the principle of Pareto optimality as a rule to be used when one seeks to decide among alternative public policies. A policy alternative is a Pareto improvement if at least one person's utility can be raised without lowering any other person's utility. That a Pareto improvement would be an improvement in the well-being of society seems relatively uncontroversial (other than for those who, as discussed above, reject the entire

[49] Analyses of the economic value of recreational experiences have used this approach, quantifying the monetary value of those things given up to recreate, to calculate a lower bound on the value of recreation experiences.

concept of utility as an indication of well-being). Unfortunately, few policies would pass the Pareto test--more often, there are both winners and losers.

As a consequence, a weaker compensation test was proposed. The so-called Kaldor-Hicks notion of compensation implies that a policy is preferred to the status quo if all those who benefit from the policy (the benefactors) could in principle compensate those who suffer (the sufferers) and still remain better off.[50] In the context of the compensation principle, the benefits of a policy are equal to the increased utility enjoyed by the benefactors, and the costs of the policy are equal to the compensation of the sufferers (see Kaldor, 1939, and Hicks, 1939). Alternatively, the benefits of a policy are equal to the maximal amount of money that people would be willing to pay to live in a world with the policy in force rather than not; conversely, the cost is equal to the minimal amount of money that people would require to live in a world in which they bore the costs of the policy.[51]

The compensation principle also suggests a way of representing the social welfare of effects of a policy in terms of the aggregate of changes in individual monetized effects. More precisely, the benefits of a policy could be said to exceed the costs if the aggregate of all beneficiaries' willingness to pay (WTP) for the program exceeds the aggregate of all sufferers' willingness to accept (WTA) compensation to live with the program. The major advantage of this approach from the perspective of CBA is that information on the monetary values of benefits or costs to various individuals can be simply aggregated to evaluate the social benefits and costs.

A number of objections to that approach are found in the literature. Over 40 years ago, the economist Kenneth Arrow proved an "impossibility theorem" stating that no simple representation of total social welfare--additive or otherwise--simultaneously satisfied a number of intuitively desirable properties. Although the truth of the theorem is not in dispute, it does not point to any alternatives for practical application of economic analysis in public-policy venues.

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[51] In reality, benefactors and sufferers may be one and the same.

Aside from this theoretical objection from within economics, there are philosophic objections to both the compensation approach in particular and any welfare-aggregation measure in general. A common concern is that this fundamentally utilitarian approach leads to ethical quandaries, e.g., when a few people can benefit a lot by making the lives of others (either now or in the future) miserable. In effect, the problem here is one in which compensation cannot be or is not paid.

An alternative perspective is one based on some concept of justice, such as the Kantian imperative to treat others fairly or Locke's view that people have the right to be secure against losses imposed by the actions of others. In the environmental-policy arena, these perspectives are manifest in concerns for resource stewardship across generations and for fairness in access to current benefits (environmental justice). Unfortunately, no definition of what constitutes justice in these contexts is widely accepted. With the exception of Rawls's (1971) justice criterion, that the utility of the least well-off be maximized, it is not easy even to translate the criteria into measurable quantitative terms; this is not a disadvantage to their advocates, but it make them obviously incompatible with CBA.

A more practical concern with aggregate net-benefits measures is the equal weighting placed on all individuals. As noted before, however, such a weighting is not an inherent requirement of CBA; instead, it is a default assumption that reflects a lack of consensus about alternative weights to reflect distributional concerns.

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Appendix B: Monetary Values for Morbidity Effects in AQVM 3.0

Table B1: Monetary Values for Morbidity Effects in AQVM 3.0

Morbidity Effect	**Estimate per Incident**[52] (1996 $CDN)			**Primary Sources**	**Description of Approach**	**Type of Estimate**
	Low	Central	High			
Adult Chronic Bronchitis *(p. 5-30 – 5-33).*[53]	$175,000	$266,000[54]	$465,000	Viscusi et al. (1991) Pricing Environmental Health Risks: Survey Assessment of Risk-Risk and Risk-Dollar Trade-offs for Chronic Bronchitis, *Journal of Environmental Economics and Management* 21 (1): 32-51. Krupnick and Cropper (1992). The Effect of Information on Health Risk Valuations. *Journal of Risk and Uncertainty*, 5 : 29-48.	WTP surveys to assess trade-off options for risks of developing (severe) chronic bronchitis vs. 1) cost of living associated with hypothetical residence locations options where in some locations risks of developing chronic respiratory disease are lower but cost of living is higher and 2) risk of death in auto accident.[55] Krupnick sample (n=190) had a relative with a chronic respiratory disease. Viscusi sample (n= 390) was more reflective of general population. The median estimate from the Viscusi study ($457,000 1990 $US) was selected as the basis for the central estimate. The 20th percentile value of $300,000 and 80th percentile value of $800,000 (1999 $US) were selected as the low and high estimates. These values were converted to 1996 CDN $ by multiplying by the 1990 PPP index[56] of 1.22 and inflating using the CDN CPI values[57] of 119.5 for 1990 and 135.7 for 1996. Using Krupnick's estimated elasticity with respect to severity of 1.16 (average case is 58% lower WTP than for severe case) the Viscusi et al. estimates were adjusted to get the central, high and low estimates for an average case of chronic bronchitis.[58]	Willingness to Pay (WTP)[59]

[52] Low, Central and High refer to low, central and high estimates used in uncertainty analysis, according to the weights which appear at bottom of table
[53] Page numbers refer to *AQVM Verson 3.0 (AQVM 3.0) Report 2: Methodology Final Report. Prepared by Stratus Consulting Inc. Sept. 3, 1999.*
[54] The WTP estimates reflect the perceived welfare effects of living with chronic bronchitis over the entire course of the illness, which can span many years.
[55] See reviewer and stakeholder comments regarding this approach.
[56] The purchasing power parity (PPP) index measures the relative value of currency based on the "purchasing power" of the currencies to convert U.S. values to their Canadian equivalent. All PPP values are from Statistics Canada National Income and Expenditure Accounts Annual Estimates 1981-1992 and 1984-1995.
[57] All Canadian consumer price index and medical cost index information comes from Statistics Canada 1996. U.S. price indices are from U.S. Bureau of Census (1994)
[58] This adjustment was done to better reflect the level of severity defined in the study by Alley et al. (1993) upon which the estimates of new cases of chronic bronchitis are based.
[59] WTP = Contingent Valuation WTP estimate.

Morbidity Effect	Estimate per Incident[60] (1996 $CDN)			Primary Sources	Description of Approach	Type of Estimate
	Low	Central	High			
Respiratory hospital admission *(p. 5-33 – 5-34)*	$3,300	$6,600	$9,800	Canadian Institute for Health Information (1994). Resource Intensity Weights: Summary of Methodology 1994/95. Burnett, R. et al. (1994). Effects of Low Ambient Levels of Ozone and Sulphates on the Frequency of Respiratory Admissions to Ontario Hospitals. *Environmental Research* 65: 172-194. Burnett R. et al. (1995). Associations between Ambient Particulate Sulphate and Admissions to Ontario Hospitals for Cardiac and Respiratory Diseases. *American Journal of Epidemiology*, 142 (1): 15-22.	The central estimate is calculated as follows: **Central $RHA=((average length of hospital stay X average daily wage[61]) + estimated cost of a hospital stay for treatment of respiratory disease in Canada ($1996) X WTP/COI ratio[62]** = (5.7 days x $117) + $2608 X 2 = $6,600 ± 50% for low and high estimate (all values rounded to the nearest $100) Hospitalization costs are estimated by multiplying Resource Intensity Weight of 1.1597 (RIW- an index of relative demand of hospital resources) by average cost of a unit of RIW which was $2,500 CDN in 1992. Cost is inflated to $2,608 (1996 $CDN using CDN medical care price index values of 136.5 for 1992 and 142.1 for 1996.[63]). For overall respiratory hospital admissions, an average across hospitalization costs was used for several respiratory illnesses related to PM10 and ozone exposure using admission rates reported in Burnett et al. (1994: 1995) as weights. A similar average across lengths of hospital stay reported for same illnesses was used to estimate foregone wages.	WTP estimates not available. Adjusted cost of illness (COI)[64] approach was used which requires data on hospitaliza-tion costs and foregone wages.

[60] Low, Central and High refer to low, central and high estimates used in uncertainty analysis, according to the weights which appear at bottom of table
[61] Statistics Canada reports average weekly earning of $586 for 1996. This is approx. $117/day. The average daily wage is used as a measure of the average opportunity cost of time for employed and not-employed individuals, on the presumption that those who are not employed value their leisure or household services at a level equal to the wage they forego in choosing not to pursue paid employment. See further discussion *p. 5-29*.
[62] A WTP/COI ratio of 2 is used to account for additional pain and suffering losses not reflected in the COI numbers (except for non-fatal cancers where 1.5 is used) . This ratio is based on studies addressing changes in incidence of asthma symptoms (Rowe et al., 1984; Rowe and Chestnut, 1986), increased frequency of angina symptoms (Chestnut et al. 1988) and risks of cataracts (Rowe and Neithercut, 1987) See further discussion *p. 5-7 - 5-9.*
[63] Estimate does not include fees for physician services.
[64] Adjusted COI = COI x 2 to approximate WTP

Morbidity Effect	Estimate per Incident[65] (1996 $CDN)			Primary Sources	Description of Approach	Type of Estimate
	Low	Central	High			
Cardiac Hospital Admission (*p. 5-34*)	$4,200	$8,400	$12,600	Canadian Institute for Health Information (1994). Resource Intensity Weights: Summary of Methodology 1994/95. Burnett, R. et al. (1994). Effects of Low Ambient Levels of Ozone and Sulphates on the Frequency of Respiratory Admissions to Ontario Hospitals. *Environmental Research* 65: 172-194. Burnett R. et al. (1995). Associations between Ambient Particulate Sulphate and Admissions to Ontario Hospitals for Cardiac and Respiratory Diseases. *American Journal of Epidemiology*, 142 (1): 15-22. Canadian hospital admissions data	Same method as above to calculate COI based estimate of value. **Central $CHA=((average length of hospital stay X average daily wage[66]) + estimated cost of a hospital stay for cardiac hospital admission in Canada ($1996) X WTP/COI ratio** Central $/Cardiac Hospital Admission = (5.6 days X $117) + $3533) X 2 = $8,400 ± 50% for low and high estimate	WTP estimates not available. Adjusted COI.
Emergency Room Visits (*p. 5-34*)	$290	$570	$860	U.S. EPA (1988). Regulatory Impact Analysis on the National Ambient Air Quality Standards for Sulfur Oxides (Sulphur Dioxide). Prepared by the Office of Air and Radiation, Research Triangle Part, NC.	U.S. average cost of emergency room visit of $85 (U.S. 1984 dollars) is converted to $168 1996 CDN dollars using the 1984 PPP index value of 1.25 and inflating using the Canadian medical care price index value of 89.9 for 1984 and 142.1 for 1996. Central $/Emergency Room Visit= (1 day[67] X $117) + $168) X 2 = $570 ± 50% for low and high estimate	WTP estimates not available. Adjusted COI.

[65] Low, Central and High refer to low, central and high estimates used in uncertainty analysis, according to the weights which appear at bottom of table
[66] Statistics Canada reports average weekly earning of $586 for 1996. This is approx. $117/day. The average daily wage is used as a measure of the average opportunity cost of time for employed and not-employed individuals, on the presumption that those who are not employed value their leisure or household services at a level equal to the wage they forego in choosing not to pursue paid employment. See further discussion *p. 5-29*.
[67] It is presumed that an emergency room visit is associated with an average of one work loss day.

Morbidity Effect	Estimate per Incident[68] (1996 $CDN)			Primary Sources	Description of Approach	Type of Estimate
	Low	Central	High			
Child Bronchitis (*p. 5-35*)	$150	$310	$460	Krupnick and Cropper (1989). Valuing Chronic Morbidity Damages: Medical Costs, Labor Market Effects, and Individual Valuations. Final Report to U.S. EPA, Office of Policy Analysis	U.S. average annual medical treatment costs of $42 (U.S. 1977 dollars) is converted to $153 1996 Canadian dollars by inflating to its 1983 U.S. dollar equivalent using the U.S. medical consumer price index values of 57.0 for 1977 and 100.6 for 1983. The 1983 dollar value is multiplied by the 1983 PPP index value of 1.24 and inflated using the Canadian medical care price index values of 85.1 for 1983 and 142.1 for 1996. Central $/Child Brochitis/year = $153 X 2 = $310 ± 50% for low and high estimate[69]	WTP estimates not available. Adjusted COI approach used.
Restricted Activity Days[70] (*p. 5-35*)	$37	$73	$110	Ostro (1987). Air Pollution and Morbidity Revisited: A Specification Test. *Journal of Environmental Economics and Management* 14:87-98.	Recent data from the Health Interview Survey[71] indicates that about 40% of all restricted activity days (RADs) are bed-disability days. Ostro (1987) suggested that RADs associated with air pollution exposure may be less severe on average than all RADs. An assumption that 20% of RADS due to air pollution exposure are bed-disability days was made. Productivity losses associated with bed-disability days are estimated as equivalent to the daily wage rate for employed individuals ($117).[72]Taking a weighted average of the value for bed-disability days and more minor RADs (see minor restricted activity days below) gives the average value for an air pollution induces RAD as follows: Central $/RAD = (.2 X $117 X 2) + (.8 X $33) = $73 ± 50% for high and low estimate	WTP is not available. Adjusted COI and WTP estimates for days with symptoms used.

[68] Low, Central and High refer to low, central and high estimates used in uncertainty analysis, according to the weights which appear at bottom of table
[69] These estimates do not reflect any value for lost productivity during the time the children are ill. Monetary estimates for lost productivity because of illness for children are not readily available. *(p. 5-35).*
[70] A restricted activity day (RAD) is a measure of illness defined by the Health Interview Survey (HIS) as a day on which illness prevents an individual from engaging in some or all of his or her usual activities. This includes days spent in bed, days missed from work, and days with minor activity restrictions because of illness.
[71] Reference not provided
[72] The same measure of lost productivity for not-employed individuals is applied on the presumption that it is a measure of average opportunity costs for all individuals. *(p. 5- 36*).

Morbidity Effect	Estimate per Incident[73] (1996 $CDN)			Primary Sources	Description of Approach	Type of Estimate
	Low	Central	High			
Asthma Symptom Days (*p. 5-36*)	$17	$46	$75	Rowe R.D., and L.G. Chestnut. (1986). Oxidants and Asthmatics in Los Angeles: A Benefits Analysis. Prepared by Energy and Resource Consultant, Inc. Report to the U.S. EPA, Office of Policy Analysis Washington, DC., March EPA 230-09-86-018.	WTP survey study that obtained asthmatics' estimates of WTP to prevent an increase in "bad asthma days" (BAD). Each respondent defined for himself a BAD on a 1 to 7 severity scale for asthma symptoms. WTP responses were positively associated with the baseline frequency of asthma symptoms and how an asthmatic defined a BAD (values increased with reported severity) A central estimate of $25 (1984 $U.S) was converted to 1996 $CDN of $46 for the central estimate, $17 for the low estimate and $75 for the high estimate by multiplying the original values by the 1984 PPP index value of 1.25 and then inflating using the Canadian consumer price index values of 92.4 for 1984 and 135.7 for 1996. [74]	WTP
Minor Restricted Activity Day *(p. 5-37)*	$20	$33	$57	Loehman et al. (1979). Distributional Analysis of Regional Benefits and Cost of Air Quality Control. *Journal of Environmental Economics and Management* 6:222-243. Tolley et al. (1986a). Valuation of Reductions in Human Health Symptoms and Risks. Prepared at the University of Chicago. Final Report for the U.S. EPA. Grant CR#-811053-01-0. January.	Survey respondents were asked how much they would be willing to pay to avoid a day with various specified symptoms such as serious or minor coughing. The focus was on respiratory symptoms that might be related to air pollution levels. Krupnick and Kopp's (1988) approach is followed which states that a MRAD must be more severe than a single symptom day (congestion, cough, etc.) and must be valued less than a work-loss day where one is entirely unable to work due to illness. The low estimate of $11 (1984 $U.S.) is based on the median estimate of Lehman's severe symptom day. Lehman's high value of $18 (1984 $U.S.) for a severe symptom day is selected for a central estimate. The high estimate of $31 (1984 $US) is based on Tolley's median estimate for a symptom combination. These values are converted to equivalent 1996 Canadian dollars ($20, $33, and $57) by multiplying by the PPP index value of 1.25 for 1984 and then inflating using the Canadian consumer price index values of 92.4 for 1984, and 135.7 for 1996.	WTP to avoid symptoms

[73] Low, Central and High refer to low, central and high estimates used in uncertainty analysis, according to the weights which appear at bottom of table
[74] These WTP estimates were also adopted by Krupnick and Kopp (1988). The Health and Agricultural Benefits from Reductions in Ambient Ozone in the United States. Resources for the Future. Washington, D.C. Discussion Paper QE-88-10.

Morbidity Effect	**Estimate per Incident**75 **(1996 $CDN)**			**Primary Sources**	**Description of Approach**	**Type of Estimate**
	Low	Central	High			
Acute Respiratory Symptom Days *(p. 5-37)*	$7	$15	$22	Loehman et al. (1979). Distributional Analysis of Regional Benefits and Cost of Air Quality Control. *Journal of Environmental Economics and Management* 6:222-243. Trolley et al. (1986a). Valuation of Reductions in Human Health Symptoms and Risks. Prepared at the University of Chicago. Final Report for the U.S. EPA. Grant CR#-811053-01-0. January.	The monetary valuation required for acute respiratory days is a value for the days on which symptoms are noticeable but do not restrict normal activities for that day. Median results of $4 to $12 per day (1984 U.S. dollars) from studies to estimate WTP to avoid a day with a single minor respiratory symptom such as head congestion or coughing were used.[76] $4, $8 and $12 were converted as the low, central and high values to equivalent 1996 Canadian dollars by multiplying the U.S. values by the 1984 PPP index value of 1.25 and then inflating using the Canadian consumer price index values of 92.4 for 1984 and 135.7 for 1996. The low, central and high values were $7, $15, and $22.	WTP
Probability weighting for all morbidity values	33%	34%	33%			

Low, Central and High refer to low, central and high estimates used in uncertainty analysis, according to the weights which appear at bottom of table

Median results from these studies were used because neither study did any adjusting for potentially inaccurate high WTP responses, resulting in reported mean WTP estimates that far exceed the median values.

Appendix C: Responses to Stakeholder Comments

Air Emissions

Level of accuracy of RDIS Emissions Inventory Data	Section 4.0 Section 4.1 Section 4.11 Section 4.12 Section 7.3.1 Section 7.3.2
Emission estimates are based exclusively on factors not actual measurement	Section 4.1 Section 4.12

Emission Reduction to Ambient Concentration Levels

Uncertainties in source-receptor relationships for PM and Ozone to properly link costs and benefits	Section 4.2 Section 4.4 Section 4.5 Section 4.8 Section 4.11 Section 4.12
Transboundary flow of pollutants from the U.S. is not considered	Section 3.2.5.1 Section 4.0 Section 4.8 Section 4.12
Simplistic methods used to simulate the relationship between PM and Ozone precursors and atmospheric levels	Section 4.1 Section 4.2 Section 4.3.1 Section 4.8 Section 4.10 Section 4.12
Present atmospheric levels of pollutants are only partially characterized	Section 4.1

Limited spatial and temporal data for PM, especially PM2.5	Section 4.1 Section 4.11 Section 4.12
Limited rural and background level data for PM and Ozone	Section 4.11 Section 4.12
Single site measurements may not be representative of regional air quality	Section 4.11 Section 4.12
Measurement error and inadequacies of ambient monitors	Section 4.11 Section 4.12

Approach to Estimation of Costs

Application of a blanket reduction scenarios of (25%, 50% 75)% is not directly comparable to benefits.	Section 7.3.1
Technologies are assigned to sources with no evidence to confirm their compatibility or functionality.	Section 7.3.1
U.S. cost estimation methodology has not been validated for use in Canada.	Section 7.3.1
U.S. cost model does not consider technology effectiveness nor capital and operating costs.	Section 7.3.1

Assessment of Health Effects

The statistical link between particulates and respiratory mortality is very weak. Dockery et al (1993) report statistically significant associations only when CV and respiratory diseases were grouped together or when deaths due to all causes were considered. The ACS study (Pope et al., 1995) reports a similar finding.	Section 5.2.1

Modest changes in CR relationships can have huge impacts on calculated benefits. Range of estimates for CRR in AQVM do not reflect true degree of complexity	Section 5.1 Section 5.2.1 Non-threshold dose-response assumption Section 5.4 Section 5.5.
Bias in the lack of presentation of epidemiological studies that report negative associations between ozone exposure and human health impacts (in the Ozone Science Assessment Document)	Section 5.1 Section 5.2.1 Causality Assumption Section 5.4
Confounding effects of co-occurring pollutants could result in overestimation of PM and ozone associated health impacts. Burnett's 1997 study concluded that the statistically significant positive association evident with fine particulate mass could "largely be explained by the gaseous air pollutants". Burnett et al. (1999) suggests that the proposed relationship between PM10 and cardiorespiratory hospitalizations (0.7% per 10 µg/m3 increase in PM10) may be significantly overstated	Section 5.1 Section 5.2.1 Causality Assumption Section 5.4 Section 5.5
Without better knowledge of who is affected and why, the public health significance of the findings are very uncertain	Section 5.2.1 Public Health Significance of Health Improvements
Default linear non-threshold dose-response may be false at the level of the individual.	Section 5.2.1 Non-threshold dose-response assumption
Burnett et al. 1995 study of PM10 associated hospital admissions used a univariate analysis based on sulphate exposures. There are methodological problems associated with approach used to convert sulphate to PM10 (ratio of sulfate to PM10 of 0.18)	Section 5.2
Studies by Schwartz et al. (1996, 1999) suggest that the coarse fraction of PM10 above 2.5 µm is not associated with mortality. The 1999 study concludes that coarse particles from windblown dust are not associated with mortality risks.	Section 5.2.1 Causality Assumption Section 5.3 Section 5.4 Section 5.5

Approach to Valuation of Health Benefits

VSL estimates appear to be very high relative to amounts that are spent on public programs to reduce risks to human life, or amounts that the public actually spends to reduce health risks. Studies of expenditures on public safety programs show that median costs per expected life saved are "low"	Section 8.5

($40,000) relative to VSL estimates.	
The use of VSL measures generates very large aggregate values that are difficult to accept given the sizes of other health related programs. For example, the aggregate value of reducing PM to background levels appears to be very large relative to the entire health care program in Canada.	Section 8.5
The economic valuation results for certain components of morbidity value appear to reflect a "worst case" scenario. For example, the estimates of Chronic Bronchitis used in the CWS process appears to be based on more severe cases than the dose response function is based on.	Section 8.5
QALY approach is a more appropriate approach given the age and compromised health status of those most affected.	Section 9.4 Appendix D
Willingness to pay approach is controversial. This controversy is not acknowledged or discussed and the justifications for selecting this as the preferred methods for CWS CBA is not provided.	Section 8.6 Section 11

Communication of Uncertainty

Issues and uncertainties associated with the methodologies are not communicated effectively. Sensitivity analyses requirement for key assumptions such as threshold assumption, discount rates etc. Sensitive bounds should be published and agreed upon with stakeholders Cumulative possible total of uncertainties needs to be part of the communication	Executive Summary Section 5.2.1 Section 7.3.2 Section 7.6 Section 11

Appendix D: Key Uncertainties in the Cost-Benefit Analysis

Table D1: Key Uncertainties in the Cost-Benefit Analysis of CWS for PM and Ozone

EMISSIONS ESTIMATION	
CWS APPROACH	Baseline emissions data from Environment Canada 1995 Residual Discharge Inventory System (RDIS) – fixed baseline No direct account taken of secondary aerosol production Transboundary (TB) sources not directly taken into account Natural emissions not directly included but indirectly included via subtraction of background levels Air Quality (AQ)– used several year average for ozone, TPM, PM_{10} and $PM_{2.5}$
PANEL CRITIQUE Key Limitations	RDIS – on a global basis NOx amounts probably accurate to about 20-30% based on fuel usage. PM sources are much more uncertain. Spatial emissions are also much more uncertain. Transboundary sources– small effect for Greater Vancouver Regional District (GVRD), 100% for Atlantic region, about 50% for the Windsor Quebec Corridor (WQC) Natural emissions – uncertain, but likely to vary from important to dominant away from urban centres, both for VOCs and $PM_{2.5}$ Open sources- potentially major contribution to PM10 but with large uncertainty Limited existing knowledge of composition of aerosols Monitoring – currently limited mostly to every 6 days for PM_{10}, $PM_{2.5}$, limited $PM_{2.5}$ data
RELATIVE UNCERTAINTIES (Probably Minor, Potentially Major)[77]	RDIS + natural sources + secondary sources – Potentially major uncertainties in spatial distribution of emissions and PM emissions in particular. Transboundary sources – potentially major for ozone and PM Natural sources– potentially major for $PM_{2.5,}$ away from urban centres; probably minor for ozone AQ monitoring probably minor for ozone while composition of aerosols is not well determined on a regular basis. This is of concern for estimation of health effects using epidemiological studies
DIRECTION OF BIAS[78]	Difficult to determine for ozone. In urban centres, will depend on whether or not in a non-linear regime. This will depend on the NOx/VOC ratio. If this is altered it could affect the linearity. PM is likely to be dominated by natural emissions away from urban centres; open sources remain uncertain and thus the cut backs applied to anthropogenic sources could sometimes be dominated by the unregulated sources.

[77] Likely Significance Relative to Key Uncertainties on Net Benefits Estimate: Probably minor (alternative assumption or approach could influence overall estimate by <20% difference), Potentially major (>20% difference). Adapted from US EPA study The Benefits and Costs of the Clean Air Act 1990 to 2010" Nov. 1990 study in which 5% difference was used see pg. 21, 33, 65, 79, 98.

[78] The U.S. EPA report "The Benefits and Costs of the Clean Air Act 1990 to 2010" Nov. 1999 used the following: Overestimate, Underestimate, Unable to determine based on current information

RECOMMENDATION/ ALTERNATIVE INPUTS, TOOLS, APPROACHES	Improvement of the emission data base on a year by year basis. Include forecast for baseline, projections/effects of other regulations coming on line. This would be inline with the GVRD. Improved spatial details for emissions. Transboundary – this would seem to be best handled by physically based (Eulerian) 3D modeling. Additional use of source receptor analysis would be very useful but will require upgrading and measuring Canadian source speciation. Need to improve estimates of natural emissions. Could improve year by year effect using remote sensing technology and measurements Correlation methods with proper source specification would improve the situation. Upgrade the current monitoring system to continuous monitoring. More rural monitoring to help assess open source/background emissions. More information on the composition of aerosols both for source identification and epidemiological studies.
TRANSLATING EMISSIONS CHANGES TO AIR QUALITY CHANGES	
CWS APPROACH	Reduction of ambient ozone and PM levels to match CWS – quasi linear for ozone and linear for $PM_{2.5}$ and PM_{10} reduction factor, R. Linear (scaled) application of R to emissions without (direct) consideration of long range transport or natural emissions.
PANEL CRITIQUE Key Limitations	Linearity would appear to be too limiting for ozone, perhaps also for $PM_{2.5}$ and PM_{10}. Data for correlation studies estimated from modeling studies that were (a) at limited horizontal resolution and (b) reductions applied in the model were across the board.
RELATIVE UNCERTAINTIES (Probably Minor, Potentially Major)[79]	Potentially major
DIRECTION OF BIAS[80]	Likely to overestimate changes in air quality for a given reduction in emissions. Could even get the direction of change wrong in certain cases.
RECOMMENDATION/ ALTERNATIVE INPUTS, TOOLS, APPROACHES	Use physical based modeling with improved emission inventory: this would address both limitations simultaneously. Develop Canadian emission data base, particularly for particle emissions, would allow for an improved assessment of effects by statistical methods. Use of integrated (3-D) Model with ozone and PM capabilities embedded in meteorological framework which is state of the art.

[79] Likely Significance Relative to Key Uncertainties on Net Benefits Estimate: Probably minor (alternative assumption or approach could influence overall estimate by <20% difference), Potentially major (>20% difference). Adapted from US EPA study The Benefits and Costs of the Clean Air Act 1990 to 2010" Nov. 1990 study in which 5% difference was used see pg. 21, 33, 65, 79, 98.

[80] The U.S. EPA report "The Benefits and Costs of the Clean Air Act 1990 to 2010" Nov. 1999 used the following: Overestimate, Underestimate, Unable to determine based on current information

ESTIMATION OF AVOIDED HUMAN HEALTH EFFECTS	
CWS APPROACH	AQVM is used to compute number of avoided health events using C-R functions drawn from the epidemiological literature (see Tables 4, 5 and 6) using a weight of evidence approach. To reflect uncertainties in the literature, low, central and high estimates are selected based on likely ranges and are assigned a probability weighting. Health endpoints for PM include: annual mortality, chronic bronchitis, respiratory hospital admissions, cardiac hospital admissions, emergency room visits, asthma symptom days, restricted activity days, acute respiratory symptom, child acute bronchitis. Health endpoints for ozone include: daily mortality risk, respiratory hospital admissions, emergency room visits, asthma symptom days, minor restricted activity days and acute respiratory symptoms. The Schwartz et al. (1996) time series study of daily mortality in 6 U.S. cities is used to develop the low C-R parameter for PM_{10} and $PM_{2.5}$. The Pope et al. (1995) prospective cross-sectional study of annual mortality rates is used for the high C-R parameter estimate. The central C-R parameter estimate is based on a two-thirds to one-third relative weighting of the Schwartz study (low parameter) and Pope et al. study (high parameter), respectively.
PANEL CRITIQUE Key Limitations	CWS gave greater weight (2/3) to mortality derived from daily time series data than to the mortality impact derived from cohort studies of annual mortality (1/3). The Pope et al. (1995) cohort study provides the firmest C-R parameter for the annual mortality impact because of the size of the cohort and the large number of North American communities. Annual mortality data should be used as the primary basis for determining the mortality impact because they include impact of peak daily exposures and cumulative effects attributable to baseline exposures over other time scales.
RELATIVE UNCERTAINTIES (Probably Minor, Potentially Major)[81]	Potentially major for estimation of reduction in mortality associated with PM and ozone reductions. Probably minor for other health endpoints.
DIRECTION OF BIAS	The effects of air pollution on health are likely underestimated because of random) errors in the accuracy of measuring exposure and outcome, and the use of daily time-series analyses which only captures acute effects. Further, the HEI reanalysis notes that C-R parameter from the Pope et al. cohort study of largely middle class volunteers is very likely an underestimate when applied to the overall population as the effect was larger for those with lesser educational attainment.

[81] Likely Significance Relative to Key Uncertainties on Net Benefits Estimate: Probably minor (alternative assumption or approach could influence overall estimate by <20% difference), Potentially major (>20% difference). Adapted from US EPA study The Benefits and Costs of the Clean Air Act 1990 to 2010" Nov. 1990 study in which 5% difference was used see pg. 21, 33, 65, 79, 98.

RECOMMENDATIONS /ALTERNATIVE INPUTS, TOOLS, APPROACHES	The central concentration response parameter should be based on the Pope et. al. 1995) study, the low from the Abbey et al. (1999) study and the high from the Dockery et al. (1993) study. The mortality benefits estimation should be more heavily weighted towards exposure-response relationships assessed for $PM_{2.5}$ rather than PM_{10}. More human chamber studies using realistic exposure conditions to explore cardiopulmonary response. These studies should be complemented with more field studies including individuals with greater susceptbility to health effects who could not participate, ethically in exposure chamber studies.
ESTIMATION OF AVOIDED NON-HEALTH EFFECTS	
CWS APPROACH	Household materials soiling was only non-health endpoint considered. Other endpoints were considered to be minor relative to health
PANEL CRITIQUE Key Limitations	Omits important endpoints relative to total of non-health endpoints such as visibility, greenhouse gases, agricultural yield, forestry, unmanaged ecosystems
RELATIVE UNCERTAINTIES (Probably Minor, Potentially Major)[82]	Potentially major from a distributional or sectoral standpoint. Ecosystem effects are highly uncertain but potentially major.
DIRECTION OF BIAS[83]	Underestimates benefits
RECOMMENDATION/ ALTERNATIVE INPUTS, TOOLS, APPROACHES	Include agricultural productivity at least[84] Use OME economic benefits, if AQVM cannot provide these numbers[85] Approach selection of non-health categories in a systematic fashion

[82] Likely Significance Relative to Key Uncertainties on Net Benefits Estimate: Probably minor (alternative assumption or approach could influence overall estimate by <20% difference), Potentially major (>20% difference). Adapted from US EPA study The Benefits and Costs of the Clean Air Act 1990 to 2010" Nov. 1990 study in which 5% difference was used see pg. 21, 33, 65, 79, 98.

[83] The U.S. EPA report "The Benefits and Costs of the Clean Air Act 1990 to 2010" Nov. 1999 used the following: Overestimate, Underestimate, Unable to determine based on current information

[84] http://www.gov.on.ca/omafra/stats/crops

[85] Impact of Ozone Exposure on Vegetation in Ontario (1989) Ontario Ministry of the Environment ARB-179-89-PHYTO, ISBN 0-7729-6386-X

BASELINE ASSUMPTIONS	
CWS APPROACH	Assumes no other existing or future air quality management policies, a static industrial structure, no economic growth, no existing abatement technologies in place, no future improvements in technology.
PANEL CRITIQUE Key Limitations	CWS does not attempt to define or quantify baselines
RELATIVE UNCERTAINTIES (Probably Minor, Potentially Major)[86]	Potentially major
DIRECTION OF BIAS[87]	Projected costs of meeting new regulations could be understated
RECOMMENDATION/ ALTERNATIVE INPUTS, TOOLS, APPROACHES	Definition of baseline is essential in a CBA study. Future CWS studies need resources to include proper estimates of: Impact of current and projected Canadian and U.S. regulatory policy Technological change Compliance baseline Projections of economic growth Demographic changes
COST OF EMISSION REDUCTION	
CWS APPROACH	Based on 1995 emissions Based on U.S. control cost data analyzed at process (SCC) level Smallest sources not included, costs less than $100/ton for NO_x controls and $150/ton for all other pollutants were eliminated Only considered the 15% least expensive sources Assumed that no control systems are currently in place Conversion of 1990 U.S. $/ton to 1995 CDN$/tonne assumed GDP deflator of 1.166029 and 15% reduction in relative cost of control technology inputs Costs are based on direct regulatory approaches without consideration of the potential for market instrument mechanisms
PANEL CRITIQUE Key Limitations	Assumes that all processes in a sector can be controlled by the same system, and that the cost will be independent of the size of the process Assumes similarity in cost and technology structure between the U.S. and Canada Assumes that costs are linear with emissions, this is only valid in certain cases

[86] Likely Significance Relative to Key Uncertainties on Net Benefits Estimate: Probably minor (alternative assumption or approach could influence overall estimate by <20% difference), Potentially major (>20% difference). Adapted from US EPA study The Benefits and Costs of the Clean Air Act 1990 to 2010" Nov. 1990 study in which 5% difference was used see pg. 21, 33, 65, 79, 98.

[87] The U.S. EPA report "The Benefits and Costs of the Clean Air Act 1990 to 2010" Nov. 1999 used the following: Overestimate, Underestimate, Unable to determine based on current information

	Costs are based on engineering costs that do not consider behaviour or market responses Tax interaction effect is not included Lack of consideration of baseline (technological change, current levels of abatement, regulatory change, economic growth) No evaluation of uncertainty Lack of transparency in implementation of model and interpretation of results Accuracy of Canadian emissions inventory data (RDIS) Impact of single control on multiple pollutants and interaction of controls aimed at separate pollutants not considered
RELATIVE UNCERTAINTIES (Probably Minor, Potentially Major)[88]	Some assumptions may have potentially major effects on cost estimation.
DIRECTION OF BIAS[89]	On balance it is likely that costs are underestimated if the tax interaction effects are as significant as they appear to be in the recent literature.
RECOMMENDATION/ ALTERNATIVE INPUTS, TOOLS, APPROACHES	Low Cost Improvements: Improved consideration of Canadian industry and source emission categories (SIC and SCC combined) and treatment options Ground truthing of control costs to the plant level Assessment of existing emission control implementation Consideration of non-technical approaches to emissions reduction (fuel switching) Consideration of co-benefits or multiple pollutant reductions with individual technologies Development of the baseline including consideration of alternative regulatory approaches (incentive approaches to emission reduction) Increase transparency in modeling of direct costs Assess degree of uncertainty in costs estimates Higher cost Improvements: Improve RDIS General equilibrium methods should be applied to regulatory policy Assess costs under incentive based regulatory schemes Research on tax-interaction effect in a Canadian context Continued development of alternative decision-making frameworks as methods to triangulate with traditional CBA Investment in human capital to improve CBA of environmental regulation

[88] Likely Significance Relative to Key Uncertainties on Net Benefits Estimate: Probably minor (alternative assumption or approach could influence overall estimate by <20% difference), Potentially major (>20% difference). Adapted from US EPA study The Benefits and Costs of the Clean Air Act 1990 to 2010" Nov. 1990 study in which 5% difference was used see pg. 21, 33, 65, 79, 98.

[89] The U.S. EPA report "The Benefits and Costs of the Clean Air Act 1990 to 2010" Nov. 1999 used the following: Overestimate, Underestimate, Unable to determine based on current information

VALUATION OF HEALTH BENEFITS	
CWS APPROACH	Use of AQVM; discount rate = 2%, 5%, 7.5%
PANEL CRITIQUE Key Limitations	No major limitations. At the time, represented consensus among economists on appropriate interpretation and treatment of literature except that almost all benefit measures are transfers from the US.
RELATIVE UNCERTAINTIES (Probably Minor, Potentially Major)[90]	Major uncertainties about the VSL because of benefits transfers involving the hedonic wage and accidental death studies to the air pollution context.
DIRECTION OF BIAS[91]	Probably biased upwards on net, but biases run in opposite directions.
RECOMMENDATION/ ALTERNATIVE INPUTS, TOOLS, APPROACHES	Maintain reliance on willingness to pay approach. AQVM needs to be updated regularly as new literature is produced and accepted. Alternative approaches could be used in sensitivity analyses.
VALUATION OF NON-HEALTH BENEFITS	
CWS APPROACH	Household soiling only non-health endpoint assessed using AQVM.
PANEL CRITIQUE Key Limitations	Estimates for household soiling are based on dated research Unclear process for identifying which non-health benefit categories to include in CWS CBA Almost all benefit measures are transfers from the U.S. Limited Canadian information.
RELATIVE UNCERTAINTIES (Probably Minor, Potentially Major)	Ecosystem effects and values are highly uncertain and potentially large
DIRECTION OF BIAS	Underestimate
RECOMMENDATION/ ALTERNATIVE INPUTS, TOOLS, APPROACHES	Update and improve AQVM with non-health benefits Include non-health benefits in a systematic fashion. Research to improve Canadian components of valuation database and ecosystem valuation estimates.

[90] Likely Significance Relative to Key Uncertainties on Net Benefits Estimate: Probably minor (alternative assumption or approach could influence overall estimate by <20% difference), Potentially major (>20% difference). Adapted from US EPA study The Benefits and Costs of the Clean Air Act 1990 to 2010" Nov. 1990 study in which 5% difference was used see pg. 21, 33, 65, 79, 98.

[91] The U.S. EPA report "The Benefits and Costs of the Clean Air Act 1990 to 2010" Nov. 1999 used the following: Overestimate, Underestimate, Unable to determine based on current information

Appendix E: Complete Text of Terms of Reference

The Royal Society of Canada Expert Panel to Review the Socio-Economic Models and Related Components Supporting the Development of Canada-Wide Standards for Particulate Matter and Ozone

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TERMS OF REFERENCE

FOR THE FORMATION AND OPERATION OF AN EXPERT PANEL TO REVIEW THE SOCIO-ECONOMIC MODELS AND RELATED COMPONENTS SUPPORTING THE DEVELOPMENT OF CANADA-WIDE STANDARDS (CWS) FOR PARTICULATE MATTER (PM) AND OZONE

Underlying Premise

An independent experts' review of the inputs, methodologies and results of models and other components related to the socio-economic analyses supporting the selection of PM and Ozone CWS would serve as a valuable input to the existing federal/provincial/territorial process. It would assist in the review of standards following the fall '99 meeting of CCME Ministers. Specifically, it would:

- Help all parties develop a better understanding and appreciation of the uncertainties associated with the analyses;
- Add further credibility to the process by broadening the openness, objectivity and transparency of the analyses; and
- Thereby improve the prospects for consensus building among all stakeholders.

Recent health and environmental assessments providing the underlying science for the selection of CWSs for PM and Ozone are already available from the WGAQOG[92] process. The experts' review will not include re-examination of the basic health and environmental science.

Panel Terms of Reference

[92] WGAQOG = Working Group on Air Quality Objectives and Guidelines.

The objective of the Expert Panel process will be to provide an independent, expert review and critique of the socio-economic analyses conducted in developing the Canada-wide standards on PM and ozone. Through a review of the models and associated data and assumptions used in the analyses, the Panel will produce a report addressing the following questions:

d. What are the strengths, merits, limitations, gaps and the degree of uncertainties of the proposed approaches, models, and their inputs and outputs?

e. By what means could the models and analytical approaches be improved, so as to minimize uncertainties and maximize the relevance, reliability and utility of outputs?

f. What other approaches and/or tools could be used to conduct these analyses?

Proposed Process

The process will be consistent with the Expert Panel Manual of Procedural Guidelines developed by the Royal Society of Canada (RSC). In the event of a conflict between this process and the RSC Guidelines, the provisions of the RSC Guidelines will prevail. The following three groups will have specified roles and responsibilities, in keeping with these guidelines:

A **Sponsors' Committee**, consisting of representatives of the CWS Development Committee (DC) for PM and Ozone and stakeholder groups (industry, environmental and health NGOs, others) will:

- select and instruct a Technical Secretariat that will in turn support the process.
- jointly with the Technical Secretariat, prepare a prospectus, consistent with these Terms of Reference and the RSC Guidelines for submission to RSC, for the work of the Expert Panel.
- approve statement of work for the Expert Panel.
- provide adequate funding and support to the Expert Panel
- approve final terms with the Expert Panel.
- receive a report outlining the results of the Expert Panel findings and recommendations; and
- ensure that the Expert Panel findings are made available to the DC and stakeholders for consideration in the CWS process.

A **Technical Secretariat**, engaged by and reporting and accountable to the Sponsors' Committee, will be comprised of CRESTech[93], NERAM[94] and the Royal Society of Canada (RSC) and will, among other tasks:

[93] Centre for Research in Earth and Space Technology
[94] Network for Risk Assessment and Management

- jointly with the Sponsors' Committee, prepare a prospectus, consistent with these Terms of Reference and the RSC Guidelines for submission to RSC, for the work of the Expert Panel.
- recruit Expert Panel members; the RSC will screen and make the final selection.
- assist the Sponsors' Committee and the Expert Panel to gain agreement on the Terms of Reference.
- RSC will advertise the Expert Panel process.
- CRESTech will provide logistical, administrative, and contractual support to the Expert Panel process.

The **Expert Panel** will:

- follow the statement of work to which it has agreed.
- hold a public meeting at the beginning of its deliberations.
- deliver its observations and suggestions as per the established
- terms and timeframe.
- present and discuss its results with the DC and stakeholders
- to assist them in their interpretation of their content and implications.

Commitment to Success

Recognizing that the Expert Panel review process constitutes a new and valued input to the CWS process that is intended to be of mutual benefit to all engaged, the DC and stakeholders commit to:

- engage actively in the process.
- share relevant information required for the process.
- provide adequate funding and support for the process.
- share results of the process freely with the Canadian public and consider the results of the Expert Panel observations and suggestions in the review of the Canada-wide Standards on PM and ozone.
- respect the established scope, objectives, schedule and budgets agreed upon for the Expert Panel.

Recognizing that the integrity and credibility of the Expert Panel, its composition, terms of reference and processes, are vital to the success of the expert review, the Sponsors' Committee, and the technical secretariat are committed to ensuring that the Expert Panel and process be:

- objective and independent.
- unbiased and free of conflicts of interest, real or perceived
- supported by proper structures and procedures consistent with established standards for such processes

This is what the RCS process is designed to accomplish.

It should be recognized that the CWS process takes, as input, peer reviewed scientific information and a range of other analytical, technical and socio-economic information, and has a well-developed mechanism for considering broad stakeholder input. The DC in consultation with the Core Advisory Group (CAG) and other stakeholders will make judgments about how the results of the socio-economic analysis are to be used in developing the CWS. The Expert Panel will focus on the quality, robustness and uncertainties related to the inputs, methodologies, models and output of the socio-economic analyses, recognizing that the scope of the analyses at this stage is at the broad macro level to assist in selection of ambient target levels and timelines and is not intended to provide the basis for detailed design of all emission reduction measures that may be needed to meet the target levels.

Components Related to Socio-Economic Analysis

Components of the socio-economic analyses to be reviewed by the Panel may include but will not be limited to the following:

A. Identifying Emission Sources and Estimating Air Quality Improvements

<u>Inputs:</u>

- identification of sources potentially implicated regionally and nationally and assumed/calculated emission reductions.

<u>Methodologies and assumptions</u>:

- assumptions on linkages of emission reductions to ambient level reductions.

<u>Outputs:</u>

- estimates of how much air quality improves with various emission reduction scenarios.

B. Estimating Costs

<u>Inputs:</u>

- emission inventories, process information, discount rates, labour rates, etc.

<u>Methodologies and assumptions:</u>

- technologies, other reduction measures and efficiencies, applied
- trigger mechanisms which determine which technology is applied
- cost algorithms and methods of calculating costs
- methodologies to determine regional and cross-pollutant impacts

Outputs:

Abatement costs, direct and indirect, to reduce pollutant emissions, aggregated in various ways (e.g., for specific source sectors, and nationally and regionally across Canada).

C. Estimating Benefits

Inputs:

- changes in ambient data on a geographic basis and links to population.

Methodologies and assumptions

- the AQVM model includes:
    - dose/response relationships
    - analyses methodologies
    - monetization assumptions

Outputs:

- estimated health and environmental impacts avoided and monetized benefits at a national and regional level.
- estimated co-benefits for other areas (e.g., climate change, acid rain).

D. Comparing Costs and Benefits

- the costs compared to the monetized benefits.
- the costs compared to the environmental resources at risk and human health impacts avoided, including any estimated co-benefits.

Members of the Panel

Vic Adamowicz, PhD (Agricultural and Applied Economics), MSc (Agricultural Economics) BSc (Agriculture), Canada Research Chair and Professor, Department of Rural Economy, University of Alberta; Program Leader, Sustainable Forest Management Network of Centres of Excellence. Panel specialty: Socio-Economics.

Robert Dales, MD, FRCP (Respiratory Medicine), FRCP(C) (Internal Medicine), CSPQ (Respiratory Medicine), CSPQ (Internal Medicine), MSc (Epidemiology & Biostatistics). Professor, Department of Medicine, Head , Respirology, University of Ottawa; Clinician & Division Head, Respirology , Ottawa Hospital. Panel Specialty: Respiratory Epidemiology

Beverley Anne Hale, PhD (Biology), MSc (Botany), BSc (Biology). Associate Professor, Department of Land Resource Science, University of Guelph. Panel specialty: Environmental Impacts.

Steve E. Hrudey, *Panel Chair*, PhD (Public Health Engineering), MSc (Public Health Engineering), BSc (Mechanical Engineering). Professor, Department Environmental Health Sciences and Associate Chair, Public Health Sciences, University of Alberta; Administrative Law Judge, Alberta Environmental Appeal Board. Panel specialty: Risk Management.

Alan Krupnick, PhD (Economics), MA (Economics) BSc (Finance). Director, Quality of the Environment Division and Senior Fellow, Resources for the Future, Washington, DC. Panel specialty: Socio-Economics and Risk Assessment.

Morton Lippman, PhD (Environmental Health Science), SM (Industrial Hygiene), BChE (The Cooper Union). Director, Human Exposure and Health Effects Program, Nelson Institute of Environmental Medicine, New York University Medical Center; Director, Aerosol & Inhalation Research Laboratory and Professor, NYU School of Medicine. Panel specialty: Environmental Health and Risk Management

John McConnell, PhD (Quantum Mechanics), BSc (Applied Mathematics), FRSC. Professor, Atmospheric Physics, Department of Earth and Atmospheric Science, York University; Co-investigator, MOPITT EOS experiment, ODIN satellite mission, and the MSC/CFCAS/NSERC Global Chemistry for Climate Project. Panel specialty: Atmospheric Science.

Paolo Renzi, MD, FRCP(C) (Internal Medicine), FRCP(C) (Pulmonary Medicine). Professor of Research, Université de Montréal; Pulmonary Physician, Notre Dame Hospital, Université de Montréal; Research Director, Meakins-Christie Laboratories, McGill University; Director, Asthma and COPD Clinic, Notre Dame Hospital. Panel specialty: Health Impacts